

**EVALUATION OF SPRAY IRRIGATION AS A METHODOLOGY  
FOR ON-SITE WASTEWATER TREATMENT AND DISPOSAL ON  
MARGINAL SOILS**

by

Gregory Thomas Monnett

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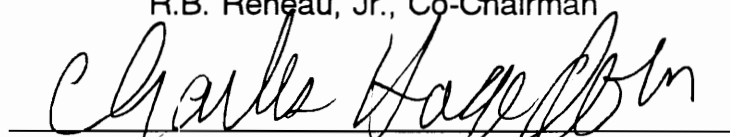
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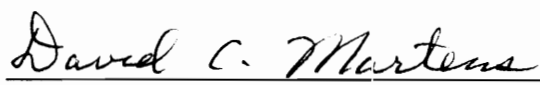
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Crop and Soil Environmental Science


APPROVED:

  
\_\_\_\_\_  
R.B. Reneau, Jr., Co-Chairman

  
\_\_\_\_\_  
C. Hagedorn, Co-Chairman

  
\_\_\_\_\_  
D.C. Martens

  
\_\_\_\_\_  
T.A. Dillaha

  
\_\_\_\_\_  
C.W. Randall

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# **Evaluation of Spray Irrigation As A Methodology For On-Site Wastewater Treatment and Disposal on Marginal Soils**

**by**

**Gregory Thomas Monnett**

**Committee Chairman: R.B. Reneau, Jr.  
C. Hagedorn  
Crop and Soil Environmental Sciences**

## **ABSTRACT**

Two field sites with soil limitations, supported by column and laboratory studies, were used to evaluate spray irrigation as an alternative methodology for onsite wastewater treatment and disposal. The first site was located in Montgomery Co., VA on a Chilhowie soil (very-fine, mixed, mesic Typic Hapludalf) and the second was located in Alleghany Co., VA on a variant of the Monongahela series (fine-loamy, mixed, mesic Typic Fragiudult). Secondary pretreatment and disinfection of the effluent was performed before application through a slow rate spray irrigation system to the soil-plant system. Effluent application rates were based on the hydraulic and N assimilative capacities of the soil-plant system. Effluent applications of 1.25 and 2.5 cm wk<sup>-1</sup> were examined at both sites.

After 20 months of effluent application at the Montgomery Co. site and 12 months of application at the Alleghany Co. site, adequate wastewater renovation was occurring in both sites. Water analyses at the 60 cm depth showed no potential groundwater contamination problems. Significant increases in chloride concentrations and EC measurements in the subsurface waters at both sites along

with little or no increases in nitrate, ammonium, or phosphorus levels indicated that the effluent was being renovated as it percolated vertically through the soil profile. Runoff water quality measurements from the spray irrigation sites indicated no serious threat to nearby surface waters. Proper filtration of the sheet flow along with the relatively low loading rates, contributed to the good quality of runoff waters in both winter and summer seasons. Plant tissue analysis indicated that plant uptake of N was a large N sink during the growing season.

Denitrification column studies examined the effect of different effluent application frequencies. There were no differences between irrigation frequencies in the amount of nitrous oxide accumulated after 24 h except in columns that were suspected to contain pockets of carbon particulates which contributed to high denitrification activity. Single daily effluent applications produced extended periods of low N<sub>2</sub>O emissions, while more frequent effluent applications produced short periods (1-3 h) of rapid N<sub>2</sub>O emission rates immediately after effluent applications. The denitrification capacity of the soils was shown to be limited by both C and NO<sub>3</sub><sup>-</sup>-N. Results from the denitrification column studies suggested that there is potential for optimizing N loss in spray irrigation systems by maintaining effluent in the microbially active topsoil through proper effluent application frequencies.

Both the field and column studies demonstrated that acceptable renovation of surface applied effluent by slow rate spray irrigation systems on sites with soil limitations can be obtained.

## DEDICATION

To my love, Margaret, who renewed my life.

## **ACKNOWLEDGEMENTS**

Special thanks goes to my advisor and friend Ray Reneau, Jr. who was a constant source of advice, insight and encouragement and who helped to build my character. My committee members, C. Hagedorn, D.C. Martens, T. Dillaha and C. Randall are acknowledged for their advice and willingness to serve on my committee. Great appreciation goes to Mike Saluta for his insights and technical expertise and to Marcia Degen, who gave much assistance and advice. The Virginia Department of Health is acknowledged for funding of this research.

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## INTRODUCTION

This research addresses the potential for use of slow rate spray irrigation (spray irrigation) for on-site wastewater treatment and disposal systems (OSWTDS) in soils that have insufficient depth or unsatisfactory hydraulic conductivity for conventional OSWTDS. Use of OSWTDS is receiving more attention as the number of residences not served by public sewage increases and the amount of land suitable for satisfactory disposal of home sewage with conventional OSWTDS decreases. The conventional OSWTDS is the most widely used system in the U.S. and consist of a septic tank and a subsurface absorption field and uses gravity to distribute effluent. Canter and Knox (1985) estimated that more than 70 million people in the U.S. utilize conventional OSWTDS. The U.S. EPA (1980) reported that approximately 25% of all housing units dispose of their wastewater by OSWTDS, and estimated that only 32% of the total land area in the U.S. has soils suitable for OSWTDS.

In Virginia, on-site treatment and disposal of domestic waste in a sanitary manner is a critical problem because of the large areas of marginally suited soils and the need to preserve agricultural land and to maintain ground and surface water quality. Use of alternative OSWTDS technologies such as spray irrigation may provide more sanitary disposal of effluent from existing homes and allow development of communities in areas with soils that do not meet the requirements for conventional OSWTDS.

Spray irrigation of treated and chlorinated effluents offers potential as an alternative to conventional OSWTDS because the effluent is applied uniformly to the soil surface and effectively utilizes the entire available renovating capacity of the soil column. Spray irrigation may be an appropriate technology in areas where a direct discharge to a surface water is not feasible and soils are inadequate for conventional or alternative OSWTDS,

This research is comprised of three parts. Part I includes the evaluation of spray irrigation as an OSWTDS in two field studies at individual homesites. Part II utilizes column and laboratory studies to evaluate denitrification rates under different effluent application frequencies and to determine the denitrification capacity of soils under different effluent applications. Part III summarizes the N balances for spray irrigation systems utilizing values from both the field and column studies.



# **PART I FIELD STUDIES**

## **INTRODUCTION**

Research using spray irrigation as an OSWTDS at homesites is limited. Experience gained from land treatment of municipal wastewater using slow rate spray irrigation systems was applied in this research for the design of the spray irrigation systems. There are concerns associated with transferring spray irrigation technology from municipal systems to single family home systems that are related to wastewater renovation capacity, homeowner maintenance of the system, storage facilities and procedures, aerosols and runoff quality. Information was gathered from neighboring states on their policies concerning spray irrigation as an OSWTDS, and regulations in those states that do allow spray irrigation of treated domestic wastewater were used as guidelines to formulate criteria for the experimental systems used in this research.

Two experimental field sites were employed to assess the potential for using spray irrigation as an alternative to conventional OSWTDS. Each site was a private residence and was chosen based on design criteria and soil characteristics. Soils used for the experimental spray irrigation systems had limitations that prohibit installation of a conventional OSWTDS, however, certain minimum soil conditions must be available before spray irrigation is considered. By designing the system on worst case conditions and by applying a high quality effluent so that zero runoff

would occur, health and environmental risks were minimized. The spray irrigation systems were evaluated by monitoring leachate water quality, quality of runoff water, soil and plant tissue analysis, denitrification measurements and soil moisture status. The primary objectives in this research were to determine the feasibility of this alternative technology for use on marginal soils and to quantify the nitrogen (N) fractions in these systems. Nitrogen is the nutrient with the greatest potential for ground and surface water contamination from OSWTDS. Determining the fractionation of N in these systems can help determine where the primary N losses occur and what N transformations have taken place. The effect of loading rates on N movements and transformations as well as other properties important to system performance were assessed.

## REVIEW OF LITERATURE

### RELATED RESEARCH PROJECTS

Research specifically designed to evaluate spray irrigation as an alternative OSWTDS is limited. However, there are well documented experiences with large scale land treatment projects applying municipal wastewaters. Two of the best known municipal irrigation projects include those at Penn State University (PSU) (Sopper and Kardos, 1973) and Unicoi State Park (Nutter and Red, 1985). Both projects operated without any winter seasonal storage and operated during freezing conditions. The PSU system has been applying secondary treated effluent in forest irrigation system for over 25 years. There has been no long term negative environmental impact at the PSU system. Nitrate ( $\text{NO}_3^-$ -N) levels have remained below  $10 \text{ mg L}^{-1}$  in samples collected from shallow wells (Schutz, 1991).

Spray irrigation of wastewater at Unicoi State Park, near Helen, GA, was initiated in 1973 and was designed to apply 283500 Lpd to 2.4 ha of forested land. Wastewater was irrigated on slopes of 15 to 30% and lengths of 90 to 110 m. The soils were clayey, oxidic, mesic, Typic Hapludult. A total of 7.5 cm of wastewater was applied each week throughout the year at a rate of  $0.81 \text{ cm h}^{-1}$ . The equipment consisted of impact sprinklers on 1.5 meter high risers and the operating pressure was 45 psi. Nutter and Red (1985) concluded that the infiltration capacity of the forest soil exceeded the wastewater application and

rainfall intensity. Although there was rapid and extensive down slope lateral subsurface flow through the A and B horizons following infiltration, the researchers concluded that there was satisfactory wastewater renovation with both high rainfall and high wastewater hydraulic loading. It was concluded that most of the N applied was loss through denitrification which was enhanced by high carbon (C) amounts in the forest soil (Nutter and Red, 1985).

Proper design is imperative for the success of slow-rate spray irrigation systems. Loading rates must be designed to the site's specific hydraulic properties. The soil permeability is a controlling factor in the degree of renovation that can occur. An example in the importance of site specificity is the work of Hook and Kardos (1978), who examined the long-term effects of spray irrigation at two different sites in Pennsylvania. The first site was on a sandy loam soil with a sandy clay loam subsoil that had rapid permeability and a hardwood forest cover. The second site was a silt loam topsoil with a silty clay to clay subsoil that had moderate permeability and a white spruce cover. Both sites were irrigated with chlorinated secondary sewage effluent year round at a rate of 5 cm wk<sup>-1</sup> with no plant material removed from the sites. The mean annual concentration of total N in the applied effluent ranged between 22.7 - 27.1 mg L<sup>-1</sup> for the nine year study. The first site proved to be ineffective after 12 months when a steady state flow developed that produced over 10 mg L<sup>-1</sup> of NO<sub>3</sub><sup>-</sup>-N in soil solution at the 120 cm depth. The second site with the clay subsoil proved to be effective for the entire term with the NO<sub>3</sub><sup>-</sup>-N concentrations rarely exceeding 10 mg L<sup>-1</sup> at the 120 cm

depth. This research demonstrates how reactions involving waste renovation in the soil are strongly influenced by soil type.

Cold weather spray irrigation has also been demonstrated. A water quality management project in East Lansing, Michigan, established both design and operation of cold-weather spray irrigation (Leland et al., 1979). Two and a half centimeters of wastewater were applied twice weekly on a site where 70% of the soils were either a loam, silt loam, or silt glacial till. The soils had a large degree of spatial variability with numerous clay and sand pockets present. The overall water balance for the study showed that 14% of the output was runoff, 67% was infiltrated into the soil and 19% was loss to evapotranspiration. The water balance for the winter (Dec-Mar) showed 29% was runoff, the remainder was infiltrated, and there was no evapotranspiration. Overall nutrient balances showed most of the N and phosphorus (P) was taken up by the soil-plant system with 90% of the input P being retained on the site.

Problems associated with winter application in the East Lansing site included frozen pipes and uneven spray distribution because of spray nozzle ice buildup. Field observations revealed unfrozen conditions under the ice pack that formed on the ground and as a result, water was able to infiltrate. Runoff water quality was the poorest during the winter, runoff volumes were small, and most of the wastewater infiltrated the soil. Frost penetration into the soil was apparently prevented by beginning irrigation early in the winter. A protective ice pack was formed on the soil which allowed for significant infiltration. Nitrate buildup was

observed in the groundwater but remained below 10 mg L<sup>-1</sup>. Leland et al. (1979) concluded that cold season wastewater irrigation is a viable option in Michigan when using low N (10 mg L<sup>-1</sup>) wastewater (Leland et al., 1979).

The Lower Colorado River Authority studied spray irrigation of wastewater from a single family residence in Llano County, Texas (Lamon, 1986). The system included a package aerobic treatment unit and applied 283 Lpd on 280 m<sup>2</sup> which is approximately 0.63 cm wk<sup>-1</sup>. Results collected from one year of application showed little correlation with effluent concentrations and constituent accumulation levels in the soil. This non-correlation was attributed to the low application rates. Only one sample was collected of a rainfall runoff event during an irrigation event. The sample results showed NO<sub>3</sub><sup>-</sup>-N levels between 1.16 - 1.93 mg L<sup>-1</sup> and P levels ranging from 1.75 - 3.48 mg L<sup>-1</sup>, while the control plot runoff had N and P concentrations of 0.42 and 0.05 mg L<sup>-1</sup>, respectively. Lamon (1986) suggested that irrigation should not be applied during rainfall events.

Scherer (1982) examined onsite household wastewater disposal by lawn sprinkling using several sand filter designs. The system recommended by Scherer (1982) consisted of a sand filter with 3 different sand media and ventilation tubes inserted vertically into the filter. Scherer (1982) concluded that disposal by lawn sprinkling as a tertiary treatment was satisfactory with no effect on the quality of runoff waters from the site.

Rebuck (1967) conducted a study in Pennsylvania to determine how a sequence of increasing irrigation amounts affected runoff and soil infiltration. The

site had a vegetative cover of red clover and the soils generally were silt loam topsoil with a clay loam or clay subsoil. The experiment consisted of three irrigation periods; Oct-Nov, Mar-Apr and June-July. Wastewater was applied via irrigation with an application rate of  $0.63 \text{ cm hour}^{-1}$ . Each period started with  $2.5 \text{ cm wk}^{-1}$  application and was increased  $2.5 \text{ cm}$  each week until  $15 \text{ cm wk}^{-1}$  was applied. Rebeck (1967) measured interflow which was defined as water which infiltrates, moves laterally, and then reappears on the surface at the base of a slope. In the first period (Oct-Nov) none of the irrigated water appeared as interflow at the  $2.5 \text{ cm wk}^{-1}$  rate. The Mar-Apr period had 9% of the effluent appear as interflow at the  $2.5 \text{ cm wk}^{-1}$  rate. The June-July period showed no interflow from effluent application at either the  $2.5 \text{ cm wk}^{-1}$  or the  $5 \text{ cm wk}^{-1}$  rate. These results demonstrate that seasonal changes can influence the hydraulic properties of an irrigation system. The Mar-Apr applications had effluent appear as interflow because there was high rainfall and relatively low evapotranspiration. This research demonstrates the importance in considering precipitation and evapotranspiration as well as water lost below the root zone. Effluent application rates must be conservative enough so that added water from precipitation will not exceed the soils infiltration rate and thus result in runoff or interflow.

## ENVIRONMENT AND HEALTH CONCERNS

A major concern with onsite spray irrigation is the aerosolization of enteric microorganisms. There are numerous reports of downwind microbial aerosol drift of untreated sewage from municipal spray irrigation fields. (Katzenelson et al., 1977.; Sorber et al., 1976.; Bausum et al., 1982.) Although aerosol bacteria are reported to have short survival times, survival can be greatly altered by factors such as relative humidity, solar radiation and temperature. Extended transport of aerosol bacteria can result in a potential health hazard. Katzenelson et al. (1977) worked with municipal effluent which was not chlorinated, and found coliforms at a maximum distance of 30 m. Bausum et al. (1983) used a municipal spray irrigation facility of approximately 2.4 ha in Ohio to look at microbial aerosol strength downwind of the spray field. The effluent used was from a lagoon and was chlorinated (2 mg L<sup>-1</sup> residual chlorine). Although there was wide variability between samplers for dye and bacteria studies, downwind bacterial aerosol concentrations at distances less than 50 m were greater than upwind concentrations. They also noted that it is difficult to define the travel of aerosol drift beyond 50 m. Aerosol bacteria counts 200 m downwind were close to background levels. Lamon (1986) showed positive counts for fecal coliform bacteria collected by deadfall sampling that were within 15 m of the sprinkler, and concluded that a 45 m buffer was appropriate for spray irrigation. Sorber et al. (1976) investigated aerosol drift from two different chlorinated effluents of municipal



wastewater. The mean aerosol reduction at 40 m downwind from the source was 96.8% for total aerobic bacteria and 98.5% for coliform-like bacteria. Chlorination has proven to be very effective in eliminating any potential threat from aerosol exposure to enteric microorganisms. Bausum et al. (1982) compared aerosol from chlorinated and unchlorinated secondary treatment wastewater. Chlorination reduced wastewater bacteria levels by 99.9% and coliphage levels were reduced by 95.4%. Proper chlorination and buffer zones are essential for safe operation of spray irrigation at homesites. Buffer zones must be established that insure safety for the owner and neighboring residents and for protection of the environment. Proper buffer zones are also needed to ensure that wastewater filters through enough of the plant-soil system for renovation to occur so that it does not significantly degrade surface and ground waters.

Contamination of groundwaters by nutrients, microbes and viruses from septic systems is a major concern. Many states have programs to ensure that systems are designed and installed properly to prevent contamination. Of these contaminants, N may potentially be the greatest groundwater pollutant. Infants who ingest waters contaminated with high concentrations of  $\text{NO}_3^-$ -N risk contracting methemoglobinemia that depletes the blood's oxygen ( $\text{O}_2$ ) supply. Nitrate contamination of groundwater also causes environmental concerns. If the groundwaters surface, the high  $\text{NO}_3^-$ -N may result in accelerated eutrophication with subsequent algal blooms and ( $\text{O}_2$ ) depletion in the water (Novotny and Chesters, 1981). U.S Public Health Services regulations have adopted  $10 \text{ mg L}^{-1}$

$\text{NO}_3^-$ -N as the drinking water level standard (U.S. EPA, 1981) Proper purification of the waste effluent by the soil is essential to prevent groundwater contamination from OSWTDS. Physical and chemical properties of the soil govern the extent of purification.

## **NITROGEN CYCLE**

Goldstein and Moberg (1973) reported that 3.3 to 5.6 kg of available N are present in each hectare-meter of typical sewage effluent. Leaching of mobile nutrients such as  $\text{NO}_3^-$ -N are potential hazards associated with land treatment of sewage effluents. Nitrogen can undergo many processes in the soil under a slow rate irrigation system. The major N transformations in the soil include: mineralization, nitrification, denitrification, ammonium fixation and absorption, immobilization, adsorption of ammonia by organic matter and plant uptake. Proper design of effluent loading rates must consider N transformations within the soil-plant system. The effluent loading rate is based on determining the load limiting factor. Estimations of the N balance and the transformations that occur in a spray irrigation system is needed to help determine the limiting factor for wastewater application. In fact, for many municipal systems that utilize spray irrigation to crops as a renovation method, N and not water assimilative capacity is normally the factor that limits effluent application. Following is a discussion of the N processes that need to be considered in designing loading rates.

## Denitrification

Denitrification is a desirable mechanism for reducing N applied in wastewater since the end product, dinitrogen gas ( $N_2$ ), is environmentally safe. Denitrification is regulated by soil moisture ( $O_2$  content), soil structure,  $NO_3^-$ -N concentration, available carbon (C) and temperature. The threshold  $O_2$  concentration which inhibits denitrification has been shown to be very low,  $10 \mu\text{mol L}^{-1}$  (Tiedje, 1988). The  $O_2$  content is directly related to the redox potential in soils. Meek et al. (1969) reported the disappearance of  $NO_3^-$ -N in columns to be related to a decrease in redox potential measured in those columns. Reddy and Patrick, (1975) incubated silt loam soils in flasks under different aerobic and anaerobic cycles. Soils under long term anaerobic conditions maintained a redox potential of -300 mV. When long aerobic periods preceded anaerobic periods the redox potential decreased at a slower rate, going from 600 mV to 200 mV and then decreasing to -300 mV. The researchers concluded that the stabilization of the potential at 200 mV was a result of denitrification. Other researchers have confirmed redox potentials between 200 and 350 mV during denitrification (Meek et al., 1969; Magdoff et al., 1974; Smith et al., 1983).

Generally finer textured soils such as silt loams, clay loams and clays are more prone to promote denitrification because drainage in these soils is commonly slow and anaerobic conditions are likely to occur. Pilot and Patrick (1972) noted that finer textured soils required less water filled porosity for denitrification to occur than in coarser textured soils. Favorable conditions in soils for denitrification of

applied wastewater include: high levels of organic matter (available C source), high cation exchange capacity for the adsorption of  $\text{NH}_4^+$ -N, neutral to alkaline soil pH, alternating saturated and unsaturated soil moisture conditions, and warm temperatures (U.S.EPA, 1981). Rolston and Cervelli (1978) indicated that determining absolute measurements of denitrification are difficult because of very complex interactions that affect the microbial populations and the process. In a conventional agricultural system N losses from denitrification can commonly be in the range of 20 to 30% as long as favorable conditions exist (Firestone, 1982). Spray irrigation of effluent has the potential to promote conditions advantageous for high denitrification rates. The irrigation cycle of alternating wetting and drying enhances the alternating aerobic and anaerobic environments. Secondary treatment of the effluent increases the amount of  $\text{NO}_3^-$ -N available for microbial reduction. And since the effluent is surface applied, available C in the topsoil can be utilized.

### **Plant Uptake**

Plant uptake is normally the primary N removal mechanism for spray irrigation systems (U.S.EPA, 1981). The uptake of N from wastewater by plants varies greatly depending on the plant, N loading rates, management conditions employed and amount of water applied. Forage grasses have relatively high N demand which increases when irrigated and harvested regularly. Some reports (Palazzo, 1981) show that forage grasses can remove over  $400 \text{ kg ha}^{-1}$  per yr of

N. Studies with industrial wastewater applications in Virginia indicate that tall fescue (*Festuca arundinacea*) has the capacity to remove from 296 to 412 kg ha<sup>-1</sup> yr of N (Boardman et al., 1984.). Forest sites do, however, present problems with nutrient loading since vegetation is not normally removed on an annual basis. Sopper and Kardos (1973) noted that nutrient renovation efficiencies of wastewater applied in forested systems are much less than renovation efficiencies in agronomic crops. The importance of vegetative cover in spray irrigation systems will be discussed later.

### **Mineralization and Immobilization**

Mineralization and immobilization are important processes involved with the soil organic N fraction. Immobilization is the incorporation of N into microbial tissue. This process may lead to the assimilation of applied N into stable humus forms in the soil. The conversion of organic N to inorganic forms of N is termed mineralization. Ammonification is the conversion of organic N to ammonia (NH<sub>3</sub>) or ammonium (NH<sub>4</sub>). Mineralization and immobilization function in opposite directions and is collectively known as mineralization immobilization turnover (MIT). Consequently, measurements of mineral N in the soil is an indication of the differences between the two processes. Under high C levels, N immobilization will increase in the soil which may erroneously be considered N loss (Stevenson, 1982). However, with time this N will be liberated to the inorganic pool.

## **Nitrification**

The biological transformation of  $\text{NH}_4^+\text{-N}$  to nitrite ( $\text{NO}_2^-$ ) and  $\text{NO}_3^-$  is nitrification. The process occurs in two steps by two different aerobic bacteria. The most common bacteria that oxidizes  $\text{NH}_4$  to  $\text{NO}_2^-$  is *Nitrosomas*. *Nitrobacter* is the most common bacteria involved in oxidizing  $\text{NO}_2^-$  to  $\text{NO}_3^-$  (Alexander, 1977). Nitrification of the wastewater is expected to occur in an aerobic pretreatment process (U.S.EPA, 1980). Applying nitrified wastewater should enhance denitrification rates in the soil since the majority of the applied N is in the  $\text{NO}_3^-$ -N form. The nitrification of waste applied  $\text{NH}_4^+\text{-N}$  is affected by soil environmental conditions, particularly temperature, moisture, pH, bacteria populations and aeration (Schmidt, 1982). Nitrification of soil applied  $\text{NH}_4^+\text{-N}$  can be fairly rapid under favorable conditions. The temperature coefficient ( $Q_{10}$ ) is 2 over a range of 5 to 35° C (Tisdale et al., 1985). The nitrification of wastewater applied  $\text{NH}_4^+\text{-N}$  can be adversely affected by the soil pH. Soil pH levels below 4.5 and above 8.5 impedes nitrification (Fuller and Warrick, 1985; Schmidt, 1982) .

## **Ammonium and Ammonia Processes**

Ammonium is a cation which can satisfy the negative charge associated with clay particles through ion exchange. Adsorbed  $\text{NH}_4^+\text{-N}$  can be displaced by other cations such as Ca, Mg and Na. Adsorption of  $\text{NH}_4^+\text{-N}$  is not affected by aerobic or anaerobic conditions, thus it occurs on soil particles in aquifers. Ammonium adsorption is temporary and can be readily oxidized to  $\text{NO}_3^-$ -N by

bacteria (Nommik and Vahtras, 1982). A soils capacity for  $\text{NH}_4^+$ -N retention depends on the soils cation exchange capacity (CEC). In most finer textured soils, exceeding the ion exchange capacity of the soil with  $\text{NH}_4^+$ -N is unlikely. Ammonium adsorption might be of importance in spray irrigation as an OSWTDS. Lance and Whisler (1972) studied N transformations in the soil using different flooding cycles in soil columns. They observed that even though  $\text{NH}_4^+$ -N absorption by the soil CEC is not an important permanent removal process, it is a benefit because it results in the retention of  $\text{NH}_4^+$ -N in the soil until a dry period when it can be oxidized to  $\text{NO}_3^-$ . They also noted that the  $\text{NO}_3^-$  can subsequently be denitrified upon another wetting period.

Ammonium can also be fixed in the interlayer space of the crystal lattice of certain clays. The ions in this state do not readily diffuse to soil solution. And they are not easily displaced by other cations. Ammonium in this state is not accessible to nitrifying bacteria. But over the long term, the effect of  $\text{NH}_4^+$ -N fixation on the N budget is minimum (Stevenson, 1982).

With increased pH,  $\text{NH}_4^+$ -N from wastewater can react with excess hydroxyl groups (OH) and convert to  $\text{NH}_3$  which can subsequently escape into the atmosphere. The volatilized  $\text{NH}_3$  can also be chemically adsorbed on organic matter and mineral fractions, which can occur under both aerobic and anaerobic conditions (Stevenson, 1982). Ammonia retention by mineral fractions is also proportional to the soils' CEC (Nommik and Vahtras, 1982). Nitrogen losses from  $\text{NH}_3$  volatilization may be increased when wastewater is irrigated. Sprinkler

systems increase evaporation and with it the quantity of  $\text{NH}_3$  volatilized. Henderson et al. (1961) reported that  $\text{NH}_3$  losses from sprinklers were near 10% between pH 7 and 8 but rose sharply above pH 8.

## PHOSPHORUS

Phosphorus can be a potential pollutant causing accelerated eutrophication of surface waters. Fortunately P is highly immobile in most soil systems because of adsorption and precipitation on soil surfaces. Since the effluent is applied on the soil surface in a spray irrigation system, biological immobilization and plant uptake can also contribute to P removal. Phosphorus immobilization can form stable organic C-P compounds. Typically sewage effluent contains considerably more of a crop's P than N requirement. However P leaching is generally not a problem since adsorption/precipitation processes in the soil effectively limit P mobility (Hsu, 1965; Enfield and Bledsoe, 1975). Plant uptake of P by forage grasses range between 29 to 56 kg ha<sup>-1</sup> per yr of P (Palazzo, 1981). Hydrous metal oxides of Fe and Al have high capacities to adsorb P. These compounds exist in the soil as discrete particles or as coatings or films on other soil particles or as amorphous compounds (Hsu, 1965). Soil mineralogy and texture are two of the most important factors affecting P adsorption. Enfield and Bledsoe (1975) noted that wastewater  $\text{PO}_4$  adsorption in the soil depended on the amounts of clay, types of clay, Al, Fe, calcium (Ca) compounds and soil pH. Phosphorus also



forms insoluble precipitates with a large number of elements. Factors such as pH, concentrations of P, Fe, Al, Ca, and competing anions affect P precipitation reactions with effluent and soils. Sawhney and Hill (1975) indicated that P adsorption sites in soils used for OSWTDS can be regenerated with time.

## **VEGETATIVE COVER**

Vegetation is an essential part of a slow rate spray irrigation system used for land treatment and renovation of domestic wastewater. Crop selection is determined by the crop's compatibility with the spray irrigation systems. Desirable plant characteristics for wastewater treatment include high N uptake, high transpiration, high tolerance for wet soils, low sensitivity to wastewater constituents, and minimum management requirements. Using vegetation for wastewater treatment offers many benefits. Vegetation can increase the permeability of soil within the root zone. Prolific root systems can increase O<sub>2</sub> and water diffusion in the soil. The plant-soil system is complex and requires careful consideration in determining wastewater loading rates. Certain wastewater constituents can adversely affect the vegetation and consequently cause system failure. Wastewater constituents that can affect plant growth include: water, P, sulfur (S), oil and grease, specific organics, salts, acids and bases, anions, metals and N (Overcash and Pal, 1979). Vegetation is one of the most effective monitoring systems of waste assimilation by a soil-plant system. Monitoring the

plants' chemical composition, dry matter production, and overall appearance can give a quick indication of system performance.

## **Forages**

Forages are the highest removers of nutrients. However harvesting the vegetative cover is necessary for nutrient removal. Among the grasses which are adaptable to Virginia, bermuda (*Cynodon dactylon*), canary grass (*Phalaris canariensis*), rye grass (*Lolium perenne*) and tall fescue are most commonly used in wastewater irrigation systems (U.S.EPA, 1981). The advantages of perennial forages was best described by (Canadian EPS, 1984). Advantages of forages include: competitive with natural vegetation, good utilization of nutrients, tolerance to wet conditions, relatively successful on poor soils, excellent erosion control, longer periods of active growth, improves soil structure and aeration, ability to withstand traffic, and cold weather hardiness. Once a selection is made it is recommended that the vegetative cover be established before construction of the spray system. It is desirable to be able to harvest the cover if nutrient removal is important.

Tall fescue is a common forage used in Virginia that can grow in a wide range of climatic conditions. It is a cool season grass that will persist through hot summers. It can also grow when the mean weekly temperature is 4.4° C. Tolerances to wet conditions, alkalinity and salinity are also advantages of fescue for wastewater treatment. Fescue is also a deep rooted crop, this offers

advantages for soil aeration and permeability. (Heath et al., 1980). Tall fescue grows well in all parts of Virginia and Kentucky 31 is the best suited cultivar (Wolf, 1988).

Reed canarygrass is a cool season perennial that is commonly used in municipal sewage irrigation sites because of its high nutrient uptake (U.S.EPA, 1981). Reed canarygrass is also very tolerant of wet conditions but does not adapt to saline soils. Another advantage is that it is one of the highest yielding perennial grasses used for grazing. Cutting of reed canarygrass is best at 10 to 13 cm. Reed canarygrass is not as useful as tall fescue in the winter since tall fescue can grow after a frost and reed canarygrass usually can not. Establishment of reed canarygrass is a problem since germination is slow and irregular (Heath et al., 1988). Reed canarygrass can be established in any region of Virginia (Wolf, 1988). Cool season perennial ryegrass can also be utilized for effluent irrigation. The use of ryegrass for effluent irrigation in Virginia is marginal because of problems with plant diseases. It is best adapted to the Appalachian and the Blue Ridge regions (Wolf, 1988). Its advantages included adaptation to wet areas and a wide range of soils. Ryegrass also has a high nutrient uptake. It is considered a bunch grass and offers a good cover for erosion control (Heath et al., 1980).

Bermudagrass is limited for effluent irrigation to the coastal plain and the southern piedmont regions of Virginia. It is a warm season perennial that grows best when mean daily temperatures are about 24° C. There is very little growth when temperatures reach 9° C. Bermudagrass grows well in both acid and basic

soils and has very high nutrient uptake. It is tolerant to flooding but not as tolerant as tall fescue and reed canarygrass. Midland is the best suited cultivar for Virginia (Heath et al., 1980).

## **Forest**

Generally soils with forest cover have excellent hydraulic properties. The absence of heavy traffic and thick surface litter enhances infiltration rates in forest soils (Sopper and Kardos, 1973). The high soil infiltration rate presents benefits in that higher hydraulic loading rates can be used and that sites with steeper slopes can be utilized since runoff is reduced in these soils. Although studies have showed that effluent irrigation does present benefits to a forest system, nutrient removal from forested systems is low.

Sopper and Kardos (1973) demonstrated increased forest yields with effluent irrigation. Plots of red pine were irrigated with municipal wastewater for ten years at 2.5 cm wk<sup>-1</sup> and 5 cm wk<sup>-1</sup> rates. The 2.5 cm wk<sup>-1</sup> rate increased the average annual height and diameter growth of the red pines as compared to a control plot with no irrigation. However the 5 cm wk<sup>-1</sup> rate proved to be detrimental to tree growth. On plots with white spruce, irrigation with 5 cm wk<sup>-1</sup> of municipal wastewater increased tree growth 360% over a 10 year period compared to a control. The researchers also showed higher productivity of hardwood forests from effluent irrigation, although only slight increases were reported in the older hardwoods (50 - 70 yrs). Younger trees (30 - 50 yrs) have shown 69% and 40%

increases in growth with effluent irrigation.

## **SOIL AND SITE CONDITIONS**

Spray irrigation systems should only be utilized on sites that have limitations that prohibit installation of conventional OSWTDS, however, certain minimum soil conditions are still needed for proper renovation of the effluent. Inadequate soil conditions can result in minimal wastewater renovation and contamination of ground and surface waters. Soil texture, hydraulic properties of the soil pedon, depth to restrictive horizon, slope, vegetative cover and buffer zones all need to be considered in criteria for site selection.

Hydraulic properties of the whole soil pedon need to be determined. Factors that affect hydraulic properties include: soil texture, chemistry, mineralogy, physical characteristics such as depth to fraigpan, depth to bedrock, stoniness, depth to watertable, compaction, slope and landscape position (Steele et al., 1986). The permeability of the most limited horizon is usually the controlling factor in determining maximum effluent loading rates. The depth of the limiting horizon is also of importance. If the horizon is too shallow proper renovation might not occur. If the soil becomes waterlogged, anaerobic conditions will develop and the level of wastewater renovation will be decreased as it travels through the soil.

It is well known that clayey soils have higher water holding capacities and generally slower infiltration rates than sandy or loamy soils and that soil texture

influences wastewater application rates. A study by Tare and Bokil (1982) investigated the relationship between wastewater application and particle size distribution. They observed that when the finer soil fraction (<75 mm) is greater than 40%, the percolation rate is drastically reduced. Effluent renovation greatly improves as the finer fraction in the soil increases. Soils such as sandy or loamy soils which have higher hydraulic conductivities may have reduced effluent treatment by the soil if effluent is applied at rates that encourage saturated flow. Plant uptake, biochemical and chemical reactions may be limited when the wastewater moves too rapidly through the soil. Adjusting the application rate to match the soils hydraulic properties are needed for proper renovation of the wastewater.

In many sections of Virginia slope will be a factor that limits site selection for spray irrigation. Kentucky has indicated that slopes in excess of 6% not be used for spray irrigation (Stanley, 1987). The North Carolina Dept. of Natural Resources (1987) suggested that slopes less than 5% are desirable, but slopes up to 10% can be utilized if berms are employed. Pennsylvania (PA Dept of Env Resources, 1972) has limited slopes for spray irrigation to 8% on suitable open grassed or wooded areas. Steeper slopes (up to 35%) have been used in forested systems, because steep grades generally do not limit the high infiltration rates in forest soils (U.S.EPA, 1981).

Kentucky (Stanley, 1987), North Carolina (NC Div. Nat. Res., 1987), and Pennsylvania (PA Dept. Env. Res., 1972) all have programs that allow spray

irrigation of treated wastewater from a single residence after biological treatment processes. These states also have different buffer zone requirements with respect to the irrigated areas. The Department of Environmental Protection in Kentucky states that it generally requires 60 m to any dwelling or neighboring lots for a loading rate of  $2.5 \text{ cm wk}^{-1}$  (Stanley, 1987). Guidelines for North Carolina buffer zones set by North Carolina Div. Nat. Res. (1987) for a loading rate of  $2.5 \text{ cm wk}^{-1}$  are as follows: 45 m to property lines, 15 m to public roads, 30 m to streams, 15 m to wet weather drainage areas, and 60 m to the owners residence. Pennsylvania which uses very conservative loading rates ( $0.5$  to  $1.25 \text{ cm wk}^{-1}$ ) compared to other states also has buffer zones that are comparably smaller. The PA Department of Environmental Resources (1972) design guidelines for spray irrigation buffer zones are as follows: 8 m to property boundaries, 8 m to roads/driveways, 30 m to occupied dwellings, 8 m to unoccupied buildings, 8 m to streams/watercourses/ponds and 15 m to wells.

## **EFFLUENT QUALITY**

The quality of the effluent applied through spray irrigation systems is comparable to the quality of the effluent now discharged from single family homes to surface waters. Pretreatment methods used for spray irrigation will normally consist of either a package aerobic unit or a septic tank and sand filter in series. However, constructed wetland technology may also provide adequate pretreatment

prior to spray irrigation. In any case, the treated effluent should be chlorinated before being sprayed.

There are several different types of aerobic units on the market, but the most common systems are composed of three compartments. The first compartment provides primary treatment where the solids are allowed to settle and anaerobic digestion occurs. The primary treatment chamber also functions as an equalization basin to prevent shock loads of toxic materials from entering the second compartment. Air is pumped into the second chamber or agitation is used to promote aerobic digestion and mixing of the wastewater. The third chamber promotes settling, or clarification, of the effluent.

Ammonium is the principal form of N in effluent from septic tanks. Most N from septic tank effluent is 75%  $\text{NH}_4^+\text{-N}$  and 25% organic N (Walker et al, 1973). With the aerobic digestion in the aeration chamber the majority of the N is expected to be nitrified to the  $\text{NO}_3^-\text{-N}$  form. Disinfection of the effluent usually follows the clarification step by using a solid tablet chlorination system. A comparison was made by (Otis and Boyle, 1976) of effluent characteristics from aerobic units and septic tanks. They noted that nearly complete nitrification is achieved through the aerobic units while ammonium ( $\text{NH}_4^+\text{-N}$ ) and organic N are the dominate forms in septic tank effluent. They also observed some denitrification in the aerobic units.

Effluent from an aerobic unit or from a sand filter is generally low in BOD, high in  $\text{NO}_3^-$ , and low in suspended solids (SS). One test conducted by the



National Sanitation Foundation Testing Laboratory (1974) evaluated the performance of an aerobic unit from the Jet Co. The results of over 100 samples collected periodically from the unit showed a 89% and 87% reduction from the influent in BOD and SS, respectively. Environmental Management Inc. (1978) conducted tests on an aerobic units used widely in Kentucky. These systems produced effluents with a BOD of 31 mg L<sup>-1</sup>, a dissolved oxygen (DO) of 4.5 mg L<sup>-1</sup>, SS of 36 mg L<sup>-1</sup>, and pH of 6.9.

An intermittent sand filter preceded by a septic tank will also produce a suitable effluent for spray irrigation if followed by chlorination (U.S.EPA, 1981). In a sand filter, the wastewater is purified by a combination of physical straining and biochemical reactions that produce a high quality effluent. The two types of sand filters that are normally employed for OSWTDS are either a buried or a free access filter. Approximately 60 to 70% of the BOD and TSS can be removed by sand filters (Bouma, 1979). Effluent N concentrations from sand filters can be estimated from literature values to be approximately 30 -40 mg L<sup>-1</sup> (Cashell et al., 1986).

One of the objectives of wastewater treatment is to filter out the suspended solids. In most cases, the filtering action of the soil removes virtually all of the suspended solids from a common household effluent. Although large quantities of suspended solids in the wastewater can lead to clogging of the soil pores. Reductions in hydraulic conductivity after prolonged infiltration of sewage effluent have been attributed to biological exudation and biomass accumulation. Vinten et

al. (1983) studied the effect of suspended solids in wastewater on the soil's hydraulic conductivity. They concluded that the reduction in hydraulic conductivity is mainly due to accumulation of the coarse suspended solids at the soil surface. They also demonstrated that filtration of the wastewater reduced the effect on the hydraulic conductance. Efficient pretreatment as with a sand filter or an aerobic unit should lessen the problems associated with pore clogging and soil permeability and thus results in a higher long term acceptance rate.

Spray irrigation as an OSWTDS are fragile systems when compared to municipal treatment plants. Small irrigation systems are designed to process normal household waste. Some household products can be classified as hazardous waste and care must be observed in their disposal. If hazardous waste is poured down the drain it can cause harmful effects to the OSWTDS and may adversely effect the environment. These chemicals, upon entering the treatment system, can kill the organisms that degrade the waste. Certain toxic solvents can also pass through the system unchanged and accumulate in the environment. Hazardous waste in effluent applied to soils through a spray irrigation system poses more of a threat of exposure (since the wastewater is exposed to the atmosphere) than a conventional subsurface treatment system. Household hazardous waste include: used motor oil, gasoline, paints, varnishes, pesticides, fertilizers, organic solvents, glues, and certain cleaners such as floor care products, metal polish and furniture polish (Water Poll. Control Fed., 1987).

## STORAGE REQUIREMENTS

A storage system for slow rate spray irrigation system is generally made up of three components: emergency storage, water balance storage, and operational storage. Most municipal systems are designed for several months storage so that effluent does not have to be applied during the winter months. Large storage facilities would not seem practicable for OSWTDS spray irrigation. A pragmatic design for an OSWTDS irrigation system applies wastewater year around with storage consisting of only that volume needed for operational and emergency storage. The Dept. of Env. Protection in Kentucky (Stanley, 1987) recommended that storage facilities of 1900 to 3700 L be used, this generally is enough for two - three days flow for an average household. In North Carolina where there are a number of home systems operating, there is a requirement of five days storage. One particular site in NC used two 3700 L septic tanks and connected the top and bottom of the tanks to act as a single tank to give a total storage of 7500 L (NC Div. Nat. Res., 1987). The storage requirements in the Pennsylvania system are for adverse weather conditions such as heavy rainfall, extreme cold, high winds, and deep snow. The Bureau of Water Quality Management in Pennsylvania simply states that storage must be designed to handle the maximum reasonable variation in flow.

## RUNOFF WATER

The quality of runoff waters from an effluent spray irrigation site is a critical concern. The degree of slope at a potential spray irrigation site is considered in site selection due to runoff concerns. Normally application rates are conservative and use only 4 - 10% of the soil infiltration rate and are designed so that application rates will produce no runoff. The occurrence of stormwater runoff is dependent upon the steepness and length of a slope, the soils' permeability and erodibility, plant cover, moisture conditions and the intensity and duration of the rainfall (Novotny and Chesters, 1981).

Results by several researchers working with spray irrigation sites have shown that stormwater runoff quality is generally acceptable. Of course results are site specific, but runoff quality can improve as it runs across large filter strip buffer areas. Overman and Schanze (1985) monitored runoff quality from a municipal irrigation site using bermudagrass as a cover. They used a loading rate of 3.3 cm wk<sup>-1</sup> on a 2% slope. They reported that large concentration reductions of suspended solids, BOD and total N occurred when the runoff went across the field. Suspended solids were reduced by 81%, BOD by 78% and total N by 67%. They concluded that runoff from the spray field had negligible impact on the quality of the receiving stream.

Scherer (1982) considered stormwater runoff quality in his dissertation "Onsite Household Wastewater Treatment for Disposal by Lawn Sprinkling". The

research compared runoff quality from plots receiving sandfilter effluent, tap water and control plots receiving no irrigation. The application rates were 0.88 cm wk<sup>-1</sup> and 0.12 cm wk<sup>-1</sup> on 9.3 m<sup>2</sup> plots. The results showed that total coliform amounts in runoff between the control and effluent sites to be different by only 2.6%. The differences in fecal coliforms were insignificant. Ammonia, BOD and total organic carbon (TOC) amounts were also insignificant between control and effluent runoff plots. Scherer concluded that runoff chemical and biological characteristics had large variation but on the average land application of sand filter effluent had no appreciable effect on the quality of runoff.

## **MATERIALS AND METHODS**

### **FIELD SITES DESCRIPTIONS**

Two experimental field systems were installed to assess the potential for use of spray irrigation as an alternative OSWTDS. The first site was located in Montgomery County, Virginia and included a single family residence that was denied a permit for a conventional OSWTDS. The site was not suited for a conventional or low pressure system due to shallow depth to shale bedrock and slowly permeable clays. The experimental site was approximately 20 acres of mostly open pasture, 5-7% slope, and southeast exposure to the sun. There were several neighboring residences close to the property boundaries. The buffer zones used for the experimental area were as follows : 60 m to home owners residence, 90 m to neighboring residence, 30 m to a stream, and 30 m to a well. The neighboring residences and nearby surface waters were beyond the buffer zones established. The Montgomery Co site had a valid NPDES permit for discharge in a nearby stream as a backup system in case the spray system experienced problems.

The second spray site was located in Alleghany County north of Covington, Virginia. This site was also a single family residence, however the site had an approved permit for conventional OSWTDS on one section of the property. The homeowner expressed interest in alternative OSWTD systems and allowed the placement of an experimental system in a pasture. The residence property was

approximately 20 acres of open pasture with slopes of 3-5%. The same buffer zones used in the Montgomery Co. site were employed at the Alleghany Co. site. The site was in an isolated area and there were no neighboring residences on the property borders. The Alleghany Co. site also had a valid NPDES permit for land application and utilized a conventional OSWTDS as a backup system.

### **SOIL CHARACTERISTICS**

The Montgomery Co. site was on a Chilhowie silt loam (very-fine, mixed, mesic Typic Hapludalf). A soil profile description along with physical, chemical and mineralogical properties are listed in Table 1. Procedures for soil characterization are described in the soil analysis section. The soil profile was typically 60-70 cm deep, however depth varied over the site. The upper limit of the shale was highly fractured. Gray mottles were present in the clay directly above the shale between 60 and 70 cm, and were few and faint in the shallower depths. The permeability of the topsoil was very rapid ( $46 \text{ cm hr}^{-1}$ ), however permeability decreased with increased depth (Table 1). Subsoil textures ranged from silty clay to clay (Table 2). The heavy clay in the Bt horizon contained 20% beidellite/vermiculite clay, and the permeability in this layer was greatly reduced. The CEC of the subsoil was relatively high due to the presence of the 2:1 clays. The profile as a whole contained a large amount of clay which increased greatly with depth.

The Alleghany Co. site was located on a variant of a Monongahela silt loam

(fine-loamy, mixed, mesic Typic Fragiudult). The soil profile description along with physical and chemical properties of the profile are given in Table 3. Many gray mottles in the 70 to 90 cm depths indicated a seasonal high water table. Some light gray mottles were present at 50 cm depth and the water table was observed to be within 50 cm of the surface on several occasions. The seasonal water table was due to the nearby shale ridges releasing water during the winter months. A weak discontinuous fragipan was located under part of the site at a depth near 1 m which was evident by the increased bulk density. The Ap and B1 horizons were mostly silt loam with increased clay content with depth (Table 4). Subsoil textures ranged from silt loam to silty clay and permeability was the slowest in the 30-45 cm depth. The upper limit of the bedrock was deeper than 150 cm however, in certain areas, alluvial deposited cobbles were found at depths of 60 to 100 cm. The soil pedon was a terrace capping over a redder residual soil.



Table 1. Soil profile description and physical characteristics of the Montgomery Co. site.

Type: Chilhowie Series

Very-fine, mixed, mesic Typic Hapludalfs

Profile Description:

A1- 0 to 5 cm, very dark grayish brown (10YR 4/3), silt loam, strong very fine granular structure, 10% shale fragments

Bt- 5 to 50 cm, yellowish brown (10YR 5/4), clay, few medium faint olive brown (2.5Y 4/4) mottles, moderate very fine and fine angular blocky structure, medium clay films on faces of peds, 10% shale fragments

C- 50 to 76 cm, olive brown (2.5Y 4/4) very shaly clay, firm, sticky, 70% shale fragments

<u>Depth</u> cm	<u>Bulk Density</u> g cm <sup>-3</sup>	<u>Hydraulic Conductivity</u> cm hr <sup>-1</sup>
0-5	0.80	45.9
15-20	1.28	3.71
30-45	1.31	6.59
45-50	1.18	0.42

Chemical Analysis:

		<u>A horizon</u>	<u>B horizon</u>
Organic Matter	%	5.5	1.0
pH		6.5	5.0
CEC	(cmol/kg)	20	38
Available P	(mg L <sup>-1</sup> )	10	3.5
Exchangeable K	(cmol/kg)	0.26	0.22
Exchangeable Ca	(cmol/kg)	14	13
Exchangeable Mg	(cmol/kg)	1.8	0.75
Exchangeable H <sup>+</sup>	(cmol/kg)	3.3	22.9

Subsoil Mineralogy:

illite	40%
beidellite/vermiculite	20%
montmorillonite	7%
quartz	11%
kalonite	5%
DCB iron	4.5%
other	12.5%

Table 2. Particle Size Analysis in the Montgomery Co. site.

Treat ment	Horz	Depth cm	BD g/ cm <sup>3</sup>	Sand mm	VC mm	C mm	M mm	F mm	VF mm	Silt um	CSI um	MSI um	FSI um	Clay um	Textural Class
-----%															
0	Ap	0-15	0.78	12.8	1.2	3.4	4.2	2.8	1.2	57.1	18.6	21.8	16.6	30.1	SiCL
	Bl	15-20	1.33	10.9	1.8	2.5	2.7	2.5	1.2	55.0	15.7	15.3	23.9	34.2	SiCL
	Bl	30-35	1.20	2.8	0.5	0.8	0.5	0.4	0.5	28.6	5.9	8.2	14.6	68.6	C
	Bl	45-50	1.29	4.1	1.1	1.4	0.6	0.6	0.3	30.7	6.0	10.9	13.8	65.2	C
1.25	Ap	0-15	0.67	11.2	1.1	2.9	4.0	2.2	1.0	59.0	17.8	22.4	18.7	29.8	SiCL
	Bl	15-20	1.35	9.2	1.8	2.0	2.1	2.0	1.2	52.6	6.3	18.6	27.7	38.2	SiCL
	Bl	30-35	1.36	8.1	1.3	2.2	1.9	1.6	0.9	47.7	4.6	17.9	25.2	44.2	SIC
	Bl	45-50	1.31	7.1	1.3	1.7	1.8	1.2	0.9	37.1	4.1	14.0	19.0	55.8	C
2.5	Ap	0-15	0.89	17.1	1.4	5.2	5.4	3.5	1.6	58.8	19.9	19.4	19.4	24.1	SIL
	Bl	15-20	1.31	16.3	1.4	4.5	5.0	3.7	1.7	56.8	9.3	22.2	25.3	26.9	SIL
	Bl	30-35	1.42	8.0	0.8	2.3	2.5	1.6	0.7	54.2	7.9	22.5	23.7	37.8	SiCL
	Bl	45-50	1.28	9.6	4.5	2.9	1.1	0.6	0.4	27.7	5.0	6.9	15.8	62.7	C

SiL = silt loam

SiCL = silty clay loam

SIC = silty clay

C = clay

**Table 3. Soil profile description and physical characteristics of Alleghany Co. site.**

Type: Monongahela silt loam (Variant)  
 fine-loamy, mixed, mesic Typic Fragiudult  
 Profile Description:

Ap- 0 to 20 cm, grayish brown (10YR 5/2) silt loam, weak fine granular structure.

B1- 20 to 33 cm, yellowish brown (10YR 5/4) silt loam, weak fine subangular blocky structure.

B21t- 33 to 56 cm, yellowish brown (10YR 5/6) silt loam, weak to moderate fine subangular blocky structure.

B22t - 56 to 84 cm, yellowish brown, silty clay loam, many light brownish gray (2.5Y 6/2) and yellowish brown (10YR 5/8) mottles.

B23t- 84 to 117 cm, light yellowish brown (10YR 6/4) silty clay loam, very distinct yellowish brown (10YR 5/8) and light brownish gray (2.5Y 6/2) mottles; has fragic characteristics.

C- 117 to 150 cm, gray (10YR 6/1) silty clay

<u>Depth</u> cm	<u>Bulk Density</u> g cm <sup>-3</sup>	<u>Hydraulic Conductivity</u> cm h <sup>-1</sup>
0-5	0.99	12.9
15-20	1.29	14.4
30-45	1.47	5.2
45-50	1.43	7.7

**Chemical Analysis:**

		<u>A horizon</u>	<u>B horizon</u>
Organic Matter	%	1.2	0.4
pH		5.6	4.9
CEC	(cmol/kg)	5.5	8.6
Exchangeable K	(cmol/kg)	0.1	0.1
Exchangeable Ca	(cmol/kg)	2.1	1.0
Exchangeable Mg	(cmol/kg)	0.3	0.6
Exchangeable H	(cmol/kg)	3.0	4.7

Table 4. Particle Size Analysis in the Alleghany Co. site.

Treatme nt	Horiz	Depth cm	BD g/ cm <sup>3</sup>	Sand mm	VC mm	C mm	M mm	F mm	VF mm	Silt um	CSI um	MSI um	FSI um	Clay um	Textured Class
----- % -----															
0	Ap	0-15	0.96	19.1	3.5	3.7	3.4	5.3	3.1	70.6	26.3	25.1	19.2	10.2	SIL
	B1	15-20	1.32	18.0	3.6	3.0	1.6	5.9	3.8	65.9	12.1	37.7	16.1	16.1	SIL
	B1	30-35	1.45	13.2	2.1	1.6	2.0	4.1	3.2	61.2	8.7	34.3	18.3	25.6	SIL
	B1	45-50	1.39	5.7	0.1	0.6	0.8	2.0	2.1	52.5	8.6	25.6	18.3	41.8	SIC
1.25	Ap	0-15	1.01	15.6	1.9	1.8	2.8	5.6	3.4	71.1	20.7	30.4	20.0	13.4	SIL
	B1	15-20	1.35	16.3	2.0	1.7	2.7	5.4	4.4	70.4	18.3	31.6	20.5	13.2	SIL
	B1	30-35	1.51	7.8	0.6	0.8	1.2	2.8	2.3	60.2	9.4	29.5	21.2	32.0	SICL
	B1	45-50	1.38	5.4	0.1	0.5	0.8	2.3	1.7	48.9	7.5	22.7	18.8	45.7	SIC
2.5	Ap	0-15	0.90	19.3	2.5	3.7	3.9	5.9	3.3	71.1	29.8	23.3	18.0	9.6	SIL
	B1	15-20	1.31	21.8	3.7	3.2	3.2	6.7	4.9	66.4	13.5	32.4	20.5	11.8	SIL
	B1	30-35	1.54	19.0	4.1	3.1	2.9	5.0	3.9	64.8	16.0	33.3	15.6	16.2	SIL
	B1	45-50	1.51	13.1	1.8	1.6	2.2	4.2	3.2	65.1	12.4	31.4	21.3	21.8	SIL

SIL = silt loam  
 SICL = silty clay loam  
 SIC = silty clay

## DESIGN OF EFFLUENT LOADING

### Hydraulic Assimilative Capacity

Since slow rate spray irrigation utilizes the soil, it must be managed in harmony with the complex biological community present in the "living filter". Hydraulic loading is an important factor in this management scheme. If the soil becomes waterlogged, anaerobic conditions will develop and the level of wastewater renovation will be reduced as waste travels through the soil. The hydraulic assimilative capacity was generally determined using the methods outlined in the U.S.EPA Process Design Manual for Land Treatment of Municipal Wastewaters (1981). The design of the effluent application rates considered such factors as the permeability of the most limiting soil horizon, depth to a restricting layer, and the month with the most restricted potential for wastewater application based on the water balance (normally January). The general water balance equation used by U.S.EPA (1981) to determine hydraulic loading rates for zero runoff from slow rate spray irrigation systems was employed to determine hydraulic assimilative capacity. The water balance equation for the slow rate spray irrigation system is shown below:

$$L_w = Et - P_r + P_w \quad (1)$$

where:

$L_w$  = wastewater hydraulic loading rate, cm/unit time

$Et$  = evapotranspiration rate, cm/unit time

$P_r$  = precipitation rate, cm/unit time

$P_w$  = percolation rate, cm/unit time

The standard time unit interval for assessing the hydraulic assimilative capacity of the soil is monthly. Since runoff of applied wastewater was designed to be 0, this component is not present in the water balance equation. The hydraulic loading rate ( $L_w$ ) was based on the percolation rate ( $P_w$ ) of the most limiting soil horizon. Evapotranspiration rates were determined using the Blaney-Cridle method (Doorenbos and Pruitt, 1977) which utilizes precipitation and temperature means estimated from routine meteorological data (HISARS, 1988). Winter evapotranspiration rates are sometimes not readily available and other data sources may have to be used. Percolation rates ( $P_w$ ) were determined from:

$$P_w = P_s * H * DDPR * DAYS \quad (2)$$

where:

- $P_w$  = Allowable percolation rate (cm month<sup>-1</sup>)
- $P_s$  = Permeability of most restrictive soil horizon (cm hr<sup>-1</sup>)
- H = hours day<sup>-1</sup>
- DDPR = daily designed percolation rate (%)
- DAYS = days month<sup>-1</sup>

The soil permeability was estimated from Soil Survey permeability estimates for the most restrictive soil horizon or based on soil properties at the site. The DDPR was chosen to be 7% of the most limiting soil horizon, which was based on guidelines given by the U.S.EPA (1981). January is generally the most limited month for applying wastewater in the hydraulic budget due to the low evapotranspiration. Calculations of the hydraulic budget for the most limited month (January) for both sites are presented in Table 5. The weekly hydraulic loading in January was 3.5 and 9.7 cm for the Montgomery Co. and Alleghany Co. sites,

respectively. The difference in allowable loading rate reflects the less restrictive permeability and lower average rainfall in Alleghany Co. site. The procedure was repeated for each month so a yearly hydraulic budget could be constructed. The yearly hydraulic budgets for both sites are presented in Tables 6 and 7.

Table 5. Calculations of the Water Balance Equation for the most limited month (January) at the Alleghany and Montgomery County sites.

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Montgomery Co. site:

$$L_w = Et - P_r + P_w$$

$$L_w = 0.2 - 7.1 + (0.4 * 24 * 0.07 * 31)$$

$$L_w = 13.9 \text{ cm month}^{-1}$$

$$L_w = \mathbf{3.5 \text{ cm wk}^{-1}}$$

Where:

$L_w$  = wastewater hydraulic loading rate, cm/unit time  
 $P_r$  = January rainfall = 7.1 cm (HISARS, 1988)  
 $Et$  = January evapotranspiration = 0.2 cm (Blaney-Criddle method)  
 $P_w = P_s * 24 \text{ hours day}^{-1} * \text{DDPR} * \text{DAYS}$   
 $P_s$  = Permeability = slow = 0.40 cm h<sup>-1</sup>  
 DDPR = daily design percolation rate = 7%  
 DAYS = number of days in month = 31

Alleghany Co. site:

$$L_w = Et - P_r + P_w$$

$$L_w = 0.8 - 5.5 + (0.835 * 24 * 0.07 * 31)$$

$$L_w = 38.7 \text{ cm month}^{-1}$$

$$L_w = \mathbf{9.7 \text{ cm wk}^{-1}}$$

Where:

$L_w$  = wastewater hydraulic loading rate, cm/unit time  
 $P_r$  = January rainfall = 5.5 cm (HISARS, 1988)  
 $Et$  = January evapotranspiration = 0.8 cm (Blaney-Criddle method)  
 $P_w = P_s * 24 \text{ hours day}^{-1} * \text{DDPR} * \text{DAYS}$   
 $P_s$  = Permeability = slow = 0.835 cm h<sup>-1</sup>  
 DDPR = daily design percolation rate = 7%  
 DAYS = number of days in month = 31

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Table 6. Annual Hydraulic Budget for Alleghany Co. site

Month	P <sub>r</sub> cm	Et cm	Excess cm	P <sub>w</sub> cm	Max Hydraulic Loading/month cm
Jan	5.5	0.8	-4.7	43.4	38.7
Feb	6.5	1.3	-5.2	39.2	38.2
Mar	6.7	4.7	-2.0	43.4	41.4
Apr	6.0	8.6	2.6	42.1	44.7
May	7.6	12.2	4.6	43.4	48.0
Jun	9.0	18.6	9.6	42.1	51.7
July	9.9	19.2	9.3	43.4	52.7
Aug	7.8	16.7	8.9	43.4	52.3
Sept	5.9	13.5	7.6	42.1	49.7
Oct	7.6	6.6	-1.0	43.4	42.4
Nov	6.3	3.2	-3.1	42.1	39.0
Dec	6.3	1.1	-5.2	43.4	38.2
<b>Total</b>	<b>79.1</b>	<b>97.9</b>	<b>21.4</b>	<b>511.4</b>	<b>502</b>

P<sub>r</sub> - (Precipitation) Average monthly rainfall data, Covington, Va 1936-86. (HISARS, 1988)

Et - (Evapotranspiration) Calculated by the Blaney-Criddle Method

Excess - (Et-P<sub>r</sub>)

P<sub>w</sub> - (Percolation) Based on a Monongehela soil, 7% of the most limiting horizon

Table 7. Annual hydraulic budget for Montgomery Co. site.

Month	P <sub>r</sub> cm	Et cm	Excess cm	P <sub>w</sub> cm	Max Hydraulic Loading/month cm
Jan	7.1	0.2	-6.9	20.8	13.9
Feb	7.9	0.9	-7.0	18.7	11.7
Mar	9.7	2.5	-7.2	20.8	13.6
Apr	8.6	4.8	-4.8	20.1	15.3
May	9.4	9.9	0.5	20.8	21.3
Jun	8.9	14.0	5.1	20.1	25.2
July	9.1	16.1	7.0	20.8	27.8
Aug	8.9	14.0	5.1	20.8	25.9
Sept	9.7	10.8	1.1	20.1	21.2
Oct	8.4	4.5	-3.9	20.8	16.9
Nov	6.6	3.1	-3.5	20.1	16.6
Dec	7.9	0.2	-7.7	20.8	13.1
Total	102	81	-22.2	244.7	222.5

P<sub>r</sub> - (Precipitation) Average monthly rainfall data, Blacksburg, Va 1953-86. (HISARS, 1988)

Et - (Evapotranspiration) Calculated by the Blaney-Criddle Method

Excess - (Et-P<sub>r</sub>)

P<sub>w</sub> - (Percolation) Based on a Chilhowie soil, 7% of the most limiting horizon

## Nutrient Assimilative Capacity

In many instances, application of wastewater via slow rate spray irrigation systems may be limited by N addition. A conservative approach to system design is to assume that wastewater percolate should contain less than 10 mg L<sup>-1</sup> before it mingles with in situ groundwater. A N balance equation was used to determine if N was a load limiting factor. An equation listed in the U.S.EPA design manual for Land Treatment of Municipal Wastewater (1981) calculates the allowable annual hydraulic loading based on N limits as follows:

$$L_n = \frac{C_p(P_r - Et) + U(10)}{(1-f)(C_n) - C_p} \quad (3)$$

where:

$L_n$  = mass loading of N, cm yr<sup>-1</sup>

$C_p$  = percolate N concentration, mg L<sup>-1</sup>

$P_r$  = precipitation rate, cm yr<sup>-1</sup>

$Et$  = evapotranspiration cm yr<sup>-1</sup>

$U$  = crop N uptake rate, kg ha<sup>-1</sup> per year

$C_n$  = N concentration in applied wastewater

$f$  = fraction of applied N removed by denitrification and volatilization

Equation 3 is a combination of the water balance and the N balance equations. From this equation the annual hydraulic loading based on N limits was developed. In using the equation for designing the loading rates in the experimental systems, 10 mg L<sup>-1</sup> N was used as maximum allowable percolate concentration ( $C_p$ ). The estimated crop uptake of N ( $U$ ) used in designing the experimental loading rates was based on data from a similar situation where effluent was spray irrigated over fescue (Boardman et al., 1984). It is assumed that approximately 25% of the N present in this effluent will be lost by NH<sub>3</sub> volatilization and denitrification.

Estimates for the fraction of applied N removed by denitrification and volatilization ( $f$ ) was based on U.S.EPA (1981) guidelines for secondary treatment effluent. The annual N assimilative capacity was calculated for both experimental sites and is shown in Table 8. The allowable weekly loading rate based on N in the equation was 4.6 and 4.3 cm in the Montgomery Co. and Alleghany Co. sites, respectively. When comparing the allowable loading rates for the hydraulic assimilative capacity and the N assimilative capacity (Tables 5 and 8) it was observed that the Montgomery Co. site was limited by the hydraulic assimilative capacity and the Alleghany Co. site was limited by the N assimilative capacity during the most limited month of January. The different limiting factors between the sites was caused by the more favorable hydraulic conditions at the Alleghany Co. site. Based on information from other states that utilize irrigation as an OSWTDS, rates of 1.25 and 2.54 cm wk<sup>-1</sup> were selected for the experimental systems. These rates were below the limiting rate for each site based on hydraulic and N assimilative capacity.

Table 8. Calculations for equation (3) in determining N load limits for both sites

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Montgomery County Site:

$$L_n = \frac{(C_p)(P_r - Et) + U(10)}{(1-f)(C_n) - C_p}$$

$$L_n = \frac{(10)(102-81) + 400(10)}{(1-.25)(37) - 10}$$

$$L_n = 237 \text{ cm yr}^{-1}$$

$$L_n = 4.6 \text{ cm wk}^{-1}$$

where:

$L_n$  = mass loading of N,  $\text{cm yr}^{-1}$

$C_p$  = percolate N concentration, ( $10 \text{ mg L}^{-1}$ )

$P_r$  = precipitation rate, ( $102 \text{ cm yr}^{-1}$ ) (Table 7)

$Et$  = evapotranspiration ( $81 \text{ cm year}^{-1}$ ) (Table 7)

$U$  = crop N uptake rate, ( $400 \text{ kg ha}^{-1}$  per yr) (Boardman et al, 1984)

$C_n$  = N concentration in applied wastewater ( $37 \text{ mg L}^{-1}$ ) (Otis and Boyle, 1976)

$f$  = fraction of applied N removed by denitrification and volatilization  
(25%) (U.S.EPA, 1981)

Alleghany County Site:

$$L_n = \frac{(C_p)(P_r - Et) + U(10)}{(1-f)(C_n) - C_p}$$

$$L_n = \frac{(10)(79.1-97.9) + 400(10)}{(1-.25)(37) - 10}$$

$$L_n = 224 \text{ cm yr}^{-1}$$

$$L_n = 4.3 \text{ cm wk}^{-1}$$

where:

$L_n$  = mass loading of N,  $\text{cm yr}^{-1}$

$C_p$  = percolate N concentration, ( $10 \text{ mg L}^{-1}$ )

$P_r$  = precipitation rate, ( $79.1 \text{ cm yr}^{-1}$ ) (Table 6)

$Et$  = evapotranspiration ( $97.9 \text{ cm yr}^{-1}$ ) (Table 6)

$U$  = crop N uptake rate, ( $400 \text{ kg ha}^{-1}$  per yr) (Boardman et al, 1984)

$C_n$  = N concentration in applied wastewater ( $37 \text{ mg L}^{-1}$ ) (Cashell et al, 1986)

$f$  = fraction of applied N removed by denitrification and volatilization  
(25%) (U.S.EPA, 1981)

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## SPRAY SYSTEM DESIGN

### Pretreatment

Spray irrigation as an OSWTDS requires proper pretreatment and disinfection of the effluent for safe operation. The pretreatment employed at the Montgomery Co. site was a NORWECO model 820 extended aeration package treatment plant with a tablet chlorinator. The unit consists of three chambers; settling, aeration and clarification (Figure 1). The settling chamber acts as a small septic tank where solids settle and undergo anaerobic treatment. The volume of the first chamber was 1700 L. The aeration chamber consists of an electric air pump which agitates and aerates the effluent for aerobic digestion. In this chamber solids are broken down and nitrification occurs. The volume of the aeration chamber was 2268 L and the detention time under a constant flow of 12.8 L min<sup>-1</sup> was approximately 2.86 h (Kellam, 1992). Effluent flows into the 945 L clarification chamber where solids are settled then recirculated into the aeration chamber and the effluent then flows through a microfilter before heading to the chlorination chamber. The retention time in the clarification chamber under a constant flow rate of 12.8 L min<sup>-1</sup> was approximately 3.78 h (Kellam, 1992). The unit was immediately followed by a tablet type chlorinator. Effluent was pumped through the tablet chlorinator with a small pump located in the chlorine contact chamber. After the chlorine contact chamber, more effluent retention time was achieved within an approximately 10 m length 15 cm diameter PVC pipe between the contact chamber and the pump chamber.

The pretreatment system at the Alleghany County site consisted of a septic tank followed by a buried gravity sand filter with a tablet chlorinator. The waste from the house first entered a 4800 L septic tank. The overflow from the septic tank was gravity distributed over the top of a sand filter. The sand filter had a surface area of 55.7 m<sup>2</sup> (600ft<sup>2</sup>), was lined with 30 mL plastic, and accommodated 2400 L day<sup>-1</sup> at an approximate loading rate of 3.7 Lpd m<sup>-2</sup> (1 gpd ft<sup>-2</sup>). The sand layer consisted of washed sand with a uniformity coefficient less than 3.5 and effective size of 0.5 to 1 mm. The distribution system on top of the sand consisted of 10 cm perforated plastic drainage pipe. The treated effluent was collected at the base of the sand filter and drained by a perforated 10 cm schedule 40 PVC pipe imbedded in the gravel layer, which delivered the effluent to the chlorinator. A cross section diagram of the sand filter is shown in Figure 2. The chlorinator was made from 10 cm diameter PVC which had perforations on the bottom of the pipe. Chlorine tablets were stacked in the pipe and effluent flowed by gravity through the pipe. From the chlorinator, the effluent traveled to a storage chamber. The spray irrigation system utilized effluent from the sand filter after chlorination. The effluent was collected in a storage chamber before application to the spray field. The focus of this research was not on the performance of the pretreatment method, therefore no suggestions for improving the pretreatment systems are given.

### **Spray System Storage**

After pretreatment, effluent flowed to a pump chamber for storage of effluent before being applied to the spray field at regular intervals. No storage of effluent, other than for equipment failure, was provided at each site. It was considered to be impractical for the systems to have water balance storage. It was believed that the average homeowner should not assume the responsibility of checking to see if daily weather conditions were favorable for wastewater application. The effluent storage tank at the Montgomery Co. site was a 6000 L concrete tank which held 1.5 days of average effluent flow. The Alleghany County site storage tank was a 8000 L tank which held 1.4 days of average effluent for emergency storage in the event of a pump failure. The storage tanks also served as a pump chamber. Both sites utilized a 1.5 HP effluent pump capable of pumping 45 Lpm against 30 m of head that supplied effluent to the spray irrigation system.

### **Spray Irrigation System**

The configuration of the spray irrigation system was similar at both sites. A general schematic of the spray irrigations systems is presented in Figure 3. The effluent application rates in the field were constant year around. The system was designed so that the daily effluent was spray irrigated each night. The anticipated daily wastewater flow at the Montgomery Co and Alleghany Co. sites was 1200 L day<sup>-1</sup> and 1800 L day<sup>-1</sup>, respectively. The effluent application was split into two 30 minute spray periods per day to encourage infiltration. Effluent was spray irrigated



at 1 am and again at 3 am. Late night application was utilized since there is low human activity during this period. The irrigation pump timer control was located in the homeowners residence. Mercury "on and off" float switches in the storage tank overrode the timed irrigation events when the effluent level in the storage tank was too high or too low. A high water alarm in the storage tank alerted the homeowner if the pump was not operating.

The supply line from the storage tank and the lateral lines to the sprinklers were both installed by using a trenching machine to excavate a 15 cm wide trench. Lines were placed below the frost line of the soil (0.75 - 1 m). The supply line consists of 5 cm, schedule 40 PVC. The lateral lines consisted of 2.5 cm, schedule 40 PVC. Pressure activated drain valves were placed at the base of each sprinkler riser, below the frost line. The drain valves emptied the risers after a spray irrigation event. This prevented the risers and sprinklers from freezing. The drain valves were placed in a small bed of gravel in order to help infiltrate wastewater that drains from the risers (Figure 4).

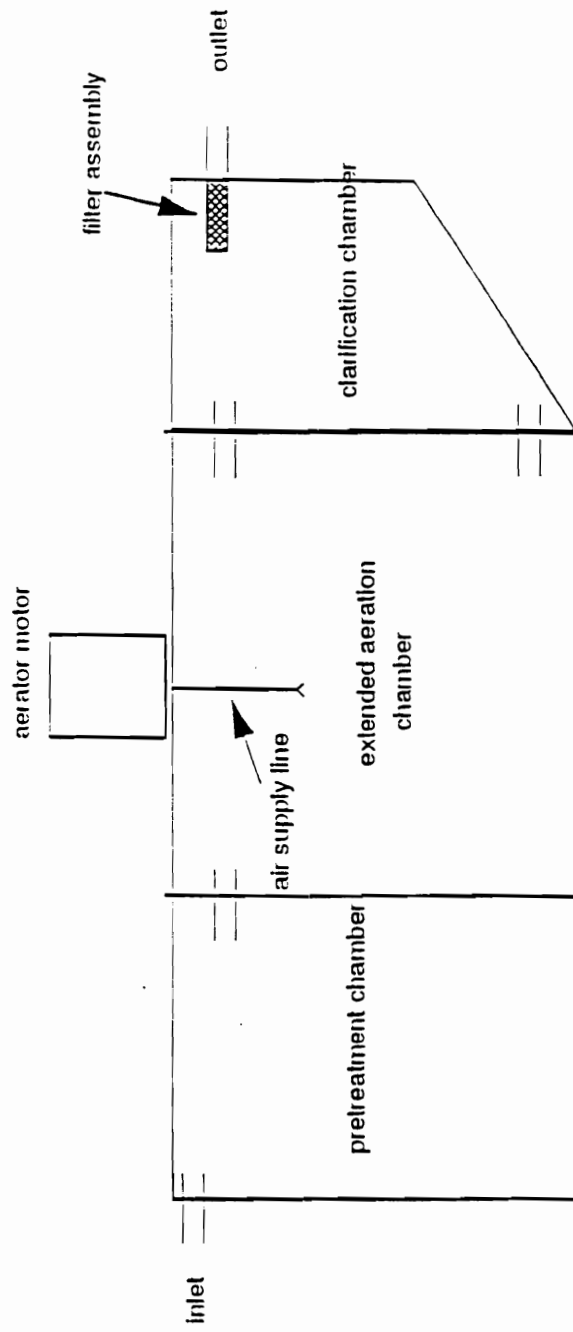


Figure 1. General schematic of the extended aeration package treatment plant.

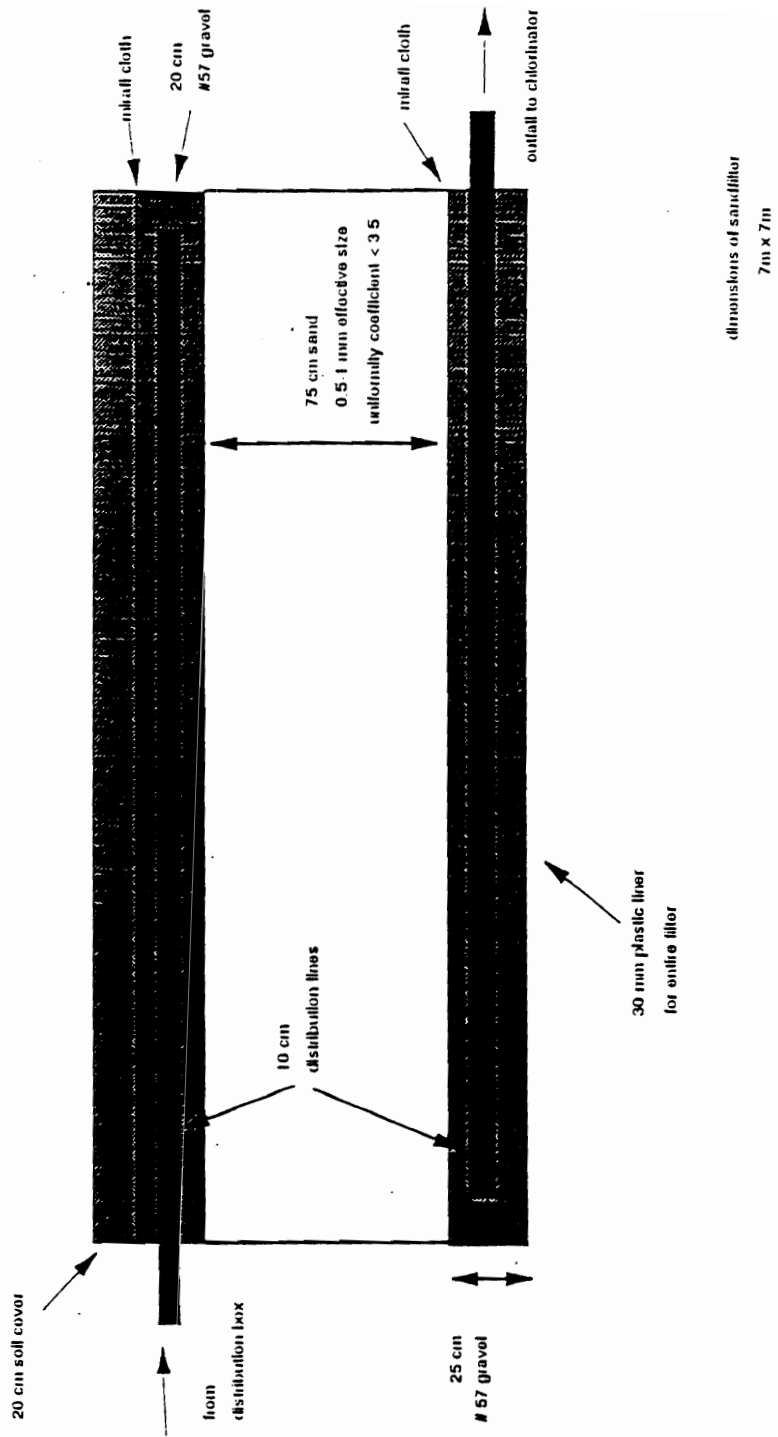


Figure 2. Cross section diagram of the sandfilter utilized in the Allegheny Co. site.

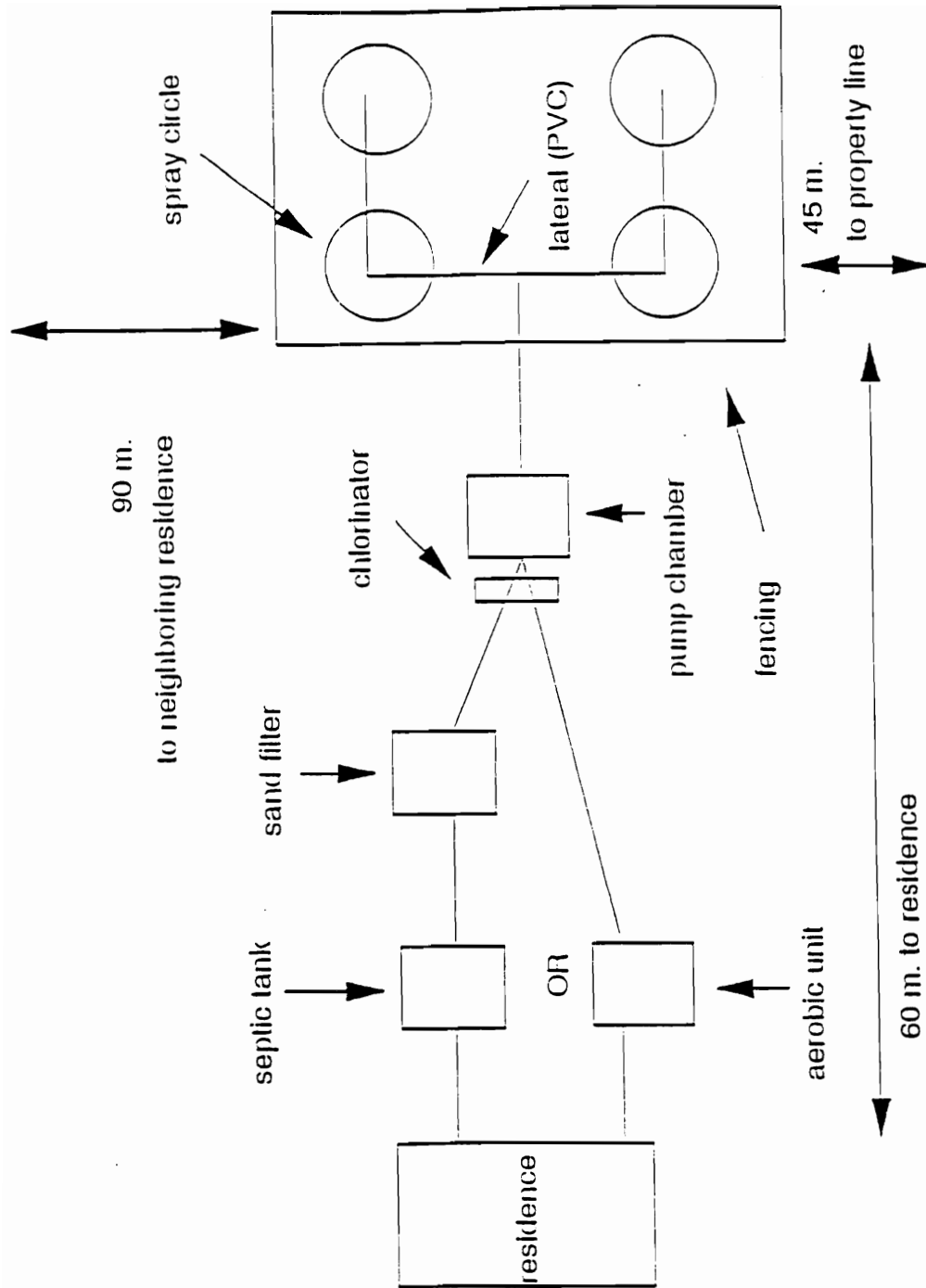


Figure 3. General schematic of spray irrigation systems.

Four risers were used to distribute effluent at both sites. The design application rates were  $1.25 \text{ cm wk}^{-1}$  (low loading rate) and  $2.5 \text{ cm wk}^{-1}$  (high loading rate). Two sprinkler heads were used to apply each rate at each site. The spray circles were 12.1 m in diameter and placed on 21 m centers to allow for potential upgrade of the system to 18 m diameter spray circles if necessary. There was no overlapping of the spray circles. The application areas were enclosed by a barbed wire fence. A 15 m buffer zone between the wetted perimeter and the fence line was also maintained.

The sprinklers used were Rainbird Model SLL20H-PM impact type sprinklers. The model used was made of stainless steel and was designed for effluent application. The sprinklers also had a frost proof design for winter application. A low trajectory ( $7^\circ$ ) and large drop size was utilized to minimize aerosol drift. A variety of nozzle sizes are available for the sprinklers which allows for flexibility in adjusting the loading rates. A 2.7 mm (7/64 inch) nozzle at 30 PSI which delivered 7.2 Lpm (1.9 gpm) was used for the  $1.25 \text{ cm wk}^{-1}$  rate and a 3.6 mm (9/64 inch) nozzle at 30 PSI which delivered 12.8 Lpm (3.4 gpm) was used for the  $2.5 \text{ cm wk}^{-1}$  rate.

To measure the uniformity of the irrigation distribution, rain gauges were placed across the spray circles at the Montgomery Co. site. In each spray circle, 12 rain gauges were placed at 3, 4, 4.9, 5.8, 6.7 and 9.1 m away from the sprinklers. Effluent was applied for 15 min and the volume collected in the gauges recorded.

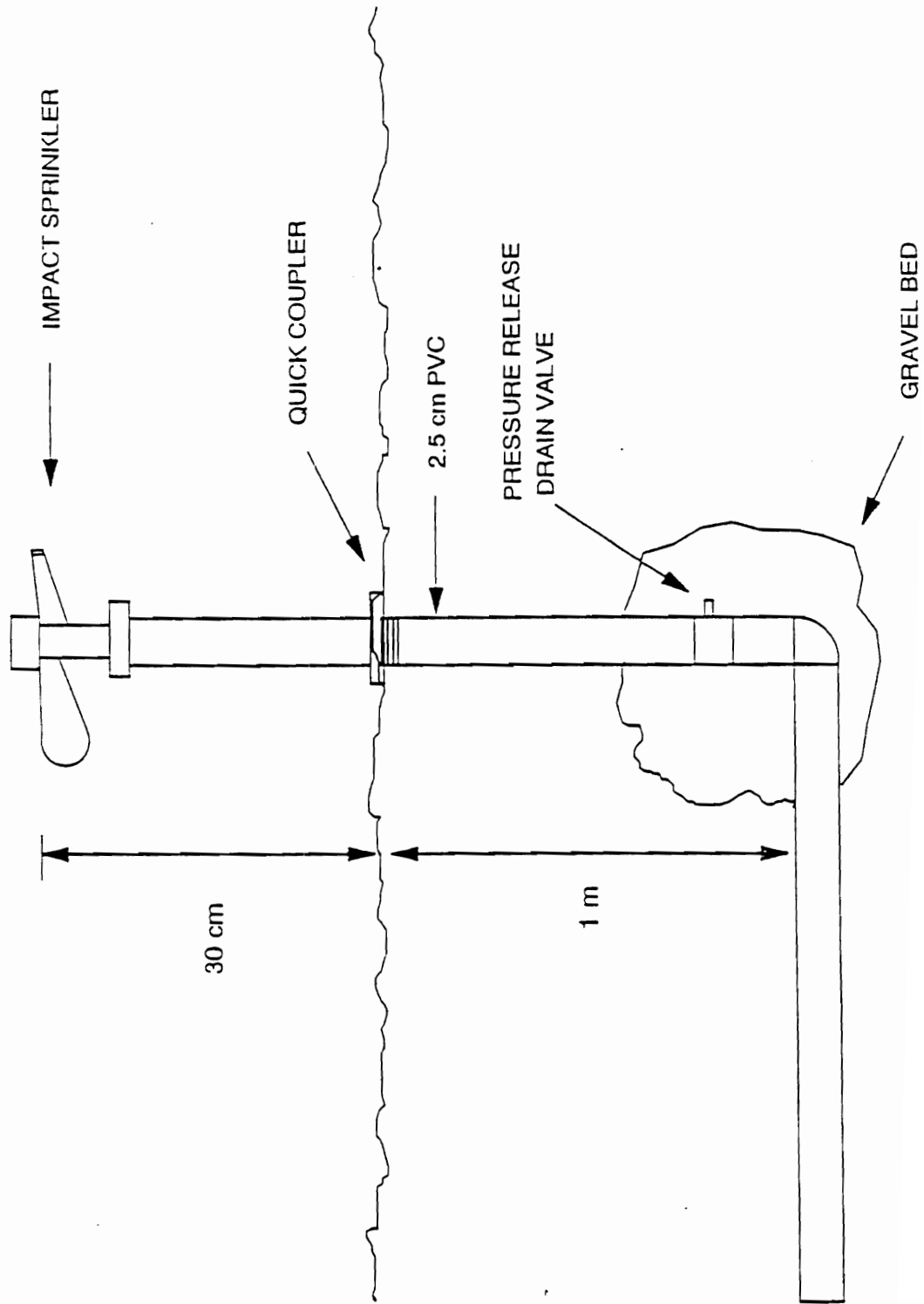


Figure 4. Design schematic of sprinkler riser and drain valve.

## EFFLUENT QUALITY

### Montgomery Co.

The aerobic unit which served as the pretreatment device prior to spray irrigation at the Montgomery Co. site, malfunctioned on several occasions. The aeration motor had to be replaced 4 times during the first year of the unit's operation. The  $\text{NO}_3^-$ -N concentrations in the unit effluent (Table 9) were below  $2 \text{ mg L}^{-1}$  which is indicative of low nitrification. Higher concentrations of  $\text{NO}_3^-$ -N are expected in the effluent from a properly functioning aeration unit. Otis and Boyle (1976) reported that an aerobic unit produced effluent with  $30.1 \text{ mg NO}_3^- \text{ N L}^{-1}$  compared with an average of  $0.56 \text{ mg NO}_3^- \text{ N L}^{-1}$  in septic tank effluent. A large reduction in  $\text{NH}_4^+$ -N concentrations from the first chamber to the unit effluent was expected due to nitrification during aerobic digestion. There was no difference between  $\text{NH}_4^+$ -N concentrations in the first chamber ( $14.6 \text{ mg L}^{-1}$ ) of the unit and the effluent ( $11.9 \text{ mg L}^{-1}$ ). The levels of TKN in the 1st chamber ( $15.4 \text{ mg L}^{-1}$ ) were also much lower than expected (Otis and Boyle, 1976). Total nitrogen levels measured in wastewater are highly dependent on the amount of organic solids present in the sample. Wastewater samples collected from the first chamber were removed only from the liquid layer near the surface of the tank. Removing liquid near the surface was not representative of the solids content near the bottom of the chamber. Chloride concentrations were high due to the use of a water softener by the homeowner.

Fluctuations in the concentrations of the wastewater components measured

is evident by the range of concentrations shown in Table 9. Fluctuations in component concentrations with time in domestic wastewater are common since flow is dependent on intermittent homeowner activity (U.S.EPA., 1980).

Proper disinfection of the effluent was dependent on the performance of a tablet chlorinator. On several occasions the chlorine tablets were swollen and stuck in the tablet holder which prevented sufficient contact with the effluent. Fecal coliform counts were determined on the chlorinated effluent on several occasions, and positive results, which indicate presence of fecal bacteria, were recorded for samples collected from the pump chamber during periods when the chlorinator did not function properly. A design improvement is needed to more effectively chlorinate the effluent from the aerobic unit. Since there was a possibility that humans might come into contact with the spray irrigated effluent, it was important that the effluent be properly disinfected before application to the soil-plant system.

### **Alleghany Co.**

Effluent quality from the septic tank - sandfilter following chlorination is shown in Table 10. Nitrification in the sand filter appeared to improve with time (Appendix Table A-1). The  $\text{NO}_3^-$ -N concentration mean of the sandfilter effluent ( $28.3 \text{ mg L}^{-1}$ ) was much higher than in the aerobic unit effluent and agrees with data reported by Otis and Boyle (1976) for sand filter performance. Ammonium concentrations were similar to values from the aerobic unit effluent.

Chloride concentrations were lower in the sandfilter effluent at the Alleghany



Co. site than in the Montgomery Co. site due to the absence of a water softener. Phosphate concentrations were similar to those present in the aerobic unit effluent at the Montgomery Co. site. However,  $\text{PO}_4\text{-P}$  concentrations from both sites were below the typical range (6-24  $\text{mg L}^{-1}$ ) for residential wastewater (U.S.EPA, 1980). The chlorinator at the Alleghany Co. site adequately disinfected the effluent from the sandfilter. Bacteria analysis of the effluent showed few or no fecal coliforms.

### **VOLUME OF EFFLUENT APPLIED**

In determining the irrigation distribution of the spray circles, the majority of the effluent was collected on the outer edge (6 m) of the spray circles (Figure 5). The "doughnut" pattern is typical of nozzles which produce large drop size. More effluent was collected in rain gauges which were on the down slope side of the sprinklers.

A flow meter was installed at the Montgomery Co. site in June 1990 in the spray field for a more accurate reading of the volume of effluent applied as compared to the pump dose counter. Means for the flow meter and pump dose counter during different time intervals are listed in Table 11. Household effluent production averaged  $971 \text{ L day}^{-1}$  which is 85.6% of the designed flow of  $1134 \text{ L day}^{-1}$ . This reduced loading rate produced actual field application rates of 1.7 and  $0.85 \text{ cm wk}^{-1}$ . The designed flow was based on VA Dept. of Health Standards which is 570 Lpd per bedroom. This estimate is generally higher than the average

water use. The counter, which records pump doses, averaged 1.7 times a day which is 85% of the designed 2 doses day<sup>-1</sup>. Since the average flow is lower than the designed application, the second dose was applying a lower quantity of effluent.

A water meter was installed in the Alleghany Co. site in May 1990, for a more accurate measure of the household effluent production. Effluent flow averaged 738 L day<sup>-1</sup> which was 43% of the 1701 L day<sup>-1</sup> design flow (Table 12). The actual field applications averaged 1.47 and 0.73 cm wk<sup>-1</sup>. The average volume of effluent applied between August and October 1990 was 597 L day<sup>-1</sup> and was only 35% of the design flow. Design flow was based on 3 bedrooms in the household. Water use by the homeowners in the Alleghany County site was much lower than Montgomery Co. site. The agreement in water usage between the flow meter and pump dose counter was not as close at the Alleghany site. The pump dose counter malfunctioned from February to May, 1990, which reduced the number of readings taken.

Table 9. Effluent quality from package aerobic unit, Montgomery Co. site

Variable	unit	N†	Mean	Min	Max	SE‡
<b>1st Chamber■</b>						
pH		3	7.0	6.6	7.7	0.3
EC	S m <sup>-1</sup>	3	0.100	0.084	0.011	0.008
NO <sub>3</sub> <sup>-</sup> -N	mg L <sup>-1</sup>	5	0.4	0.1	0.6	0.1
NH <sub>4</sub> <sup>+</sup> -N	mg L <sup>-1</sup>	5	14.6	9.3	18.7	2.0
TKN	mg L <sup>-1</sup>	3	15.4	9.5	19.2	3.0
PO <sub>4</sub> -P	mg L <sup>-1</sup>	2	3.6	2.9	4.3	0.7
Cl <sup>-</sup>	mg L <sup>-1</sup>	3	58	53	66	4
<b>Unit Effluent</b>						
pH		7	7.6	6.3	8.4	0.3
EC	S m <sup>-1</sup>	3	0.108	0.080	0.122	0.013
NO <sub>3</sub> <sup>-</sup> -N	mg L <sup>-1</sup>	19	1.9	0.1	9.8	0.7
NH <sub>4</sub> <sup>+</sup> -N	mg L <sup>-1</sup>	18	11.9	2.4	19.3	1.3
TKN	mg L <sup>-1</sup>	12	20.6	3.9	54.3	3.6
PO <sub>4</sub> -P	mg L <sup>-1</sup>	11	4.1	3.4	5.0	0.2
Total P	mg L <sup>-1</sup>	6	7.4	6.1	8.1	0.3
Cl <sup>-</sup>	mg L <sup>-1</sup>	8	54	44	88	5

† N indicates number of samples.

‡ SE indicates standard error.

■ 1st chamber in the aerobic unit where anaerobic conditions and settling exist.

Table 10. Chlorinated effluent quality from a gravity dosed sand filter, Alleghany Co site.

Variable	unit	N†	Mean	Min	Max	SE‡
pH		4	7.1	6.1	7.6	0.3
EC	S m <sup>-1</sup>	2	0.070	0.056	0.085	0.014
NO <sub>3</sub> <sup>-</sup> -N	mg L <sup>-1</sup>	6	28.3	0.5	80.0	11.6
NH <sub>4</sub> <sup>+</sup> -N	mg L <sup>-1</sup>	6	15.4	9.1	35.7	4.2
PO <sub>4</sub> -P	mg L <sup>-1</sup>	3	2.7	0.7	4.8	1.2
Cl <sup>-</sup>	mg L <sup>-1</sup>	2	13	12	14.1	12.2

† N indicates number of samples.

‡ SE indicates standard error.

Table 11. Volume of effluent applied at the Montgomery Co. site.

Date	Days between measurements	Dose Counter	Water Meter
		dose day <sup>-1</sup>	L day <sup>-1</sup>
7/27/89	55	1.6	
8/14/89	18	1.9	
9/13/89	30	2.0	
9/28/89	15	1.7	
1/11/90	65	2.0	
3/7/90	55	1.9	
5/17/90	71	2.0	
7/13/90	21		828
7/16/90	60	2.8	
8/7/90	25		877
9/25/90	49		971
12/11/90	148	1.8	
2/4/91	55	1.8	
2/4/91	132		1011
Total Mean		1.7	971
% of Design Flow		85.0	85.6

Table 12. Volume of effluent applied at the Alleghany Co. site.

Date	Days between measurements	Dose Counter dose day <sup>-1</sup>	Water Meter L day <sup>-1</sup>
6/19/90	41	1.8	847
7/11/90	22		684
8/29/90	49		646
8/29/90	71	1.3	
10/30/90	62		597
10/30/90	62	1.1	
1/14/91	76		865
Total Mean		1.3	738
% of Design Flow		65.0	43.4

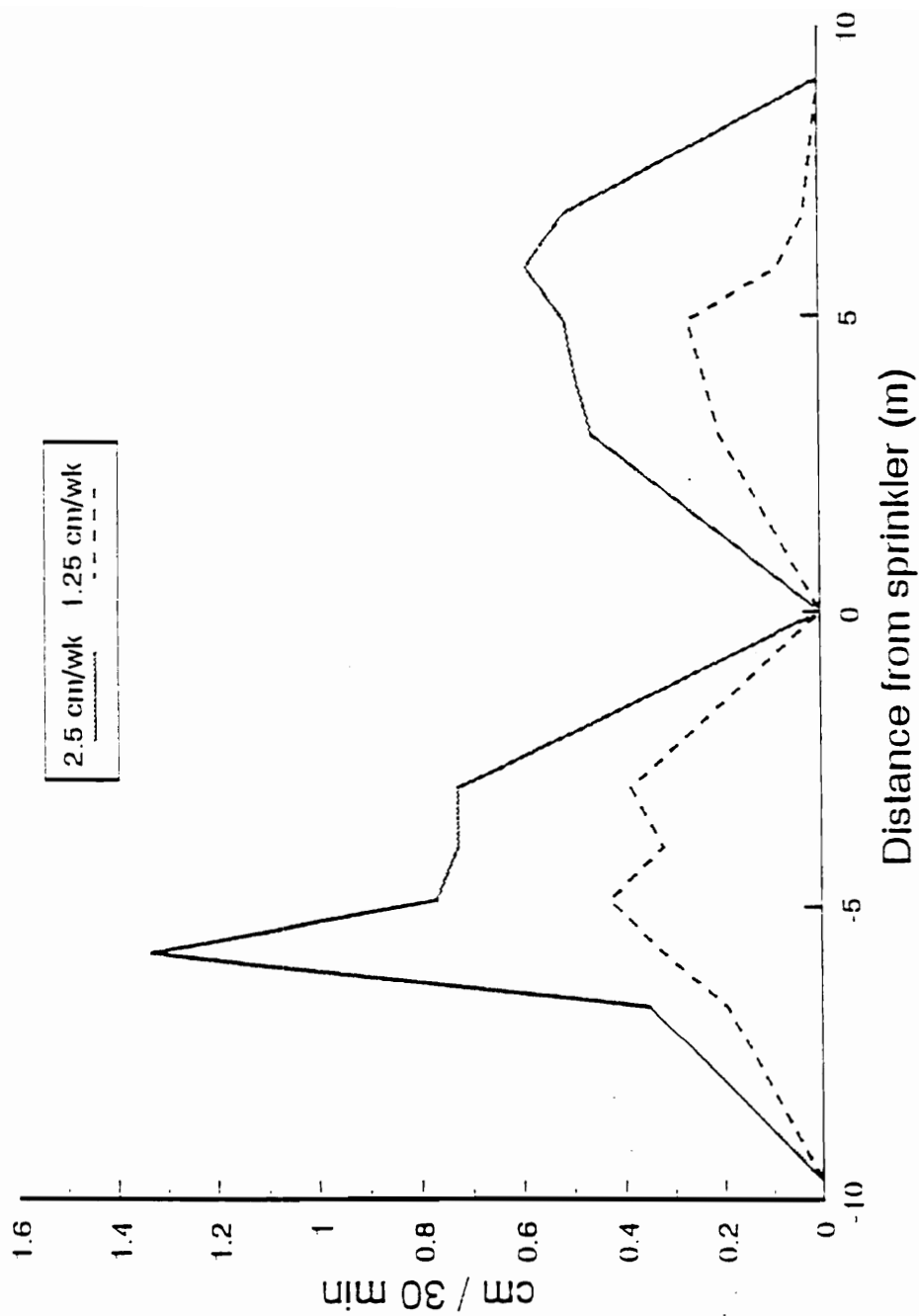


Figure 5. Irrigation distribution curves for both loading rates at the Montgomery Co. site

## **SYSTEM ANALYSIS**

The placement of monitoring equipment at the Montgomery Co. and Alleghany Co. sites are shown in Figures 6 and 7, respectively. Tensiometers, suction lysimeters, sample wells at two depths, access tubes for neutron moisture meter readings, and runoff collection were all included within a sampling area. Soil moisture content was determined with depth at several selected times over the course of the experiment. Samples of runoff water, soil, denitrification gases and plant tissue were also collected periodically throughout the study. The details on each monitoring procedure are discussed in more detail in following sections.

### **Subsurface Water Collection**

Subsurface water samples were collected to determine the groundwater and soil solution quality at different depths in the soil profile. Subsurface water samples from both sites were collected from suction lysimeters and shallow wells. Monthly sampling of subsurface water was conducted (weather permitting) at both sites. Water samples were taken from the Montgomery Co. site during a 20 month period and during a 12 month period in the Alleghany Co. site.

### ***Suction Lysimeters***

All suction lysimeters were placed at a depth of 60 cm. Suction lysimeters were installed in order to obtain soil solution samples during unsaturated soil



conditions. Lysimeters were placed within the runoff sampling strips in four different locations across the spray circle diameter. In each site 36 solution samplers were installed. In the Montgomery County site, the first lysimeter was located on the edge or top of the spray circle. The second, third and fourth lysimeters were located 4.6, 9.2 and 13.8 m down slope from the top of the spray circle, respectively. In the Alleghany Co. site, the first lysimeter was located 3 m from the top of the spray circles and the second, third and fourth lysimeters were located 7.6, 12.2 and 16.8 m from the top of the spray circles respectively.

The suction lysimeters were constructed by fitting a 100 kPa high flow ceramic cup to a 5 cm diameter length of Fasco tubing. The ceramic cups were fitted using epoxy adhesive to insure an airtight seal. To install the sampler, a 6 cm hole was augured to the desired depth. A slurry of silica flour was poured into the hole so that the ceramic cup would be covered when the sampler was inserted. The flour ensures good contact with the surrounding soil. After the sampler was in place, soil was packed around the sampler to within 2-6 cm of the surface. Cement was poured around the samplers to insure sealing. Bentonite was placed around the surface of the sampler as an added protection against water short cutting down the side of the sampler. A section of PVC pipe was used as a sleeve over the top 10 cm of the lysimeter which was cut at the soil surface. The sleeve was also cemented in the ground and a cap was placed over the sleeve to prevent debris falling into the lysimeter. The lysimeters were cut at the soil surface in order to make it easier to mow over the plots. Although samples could

be obtained from the suction lysimeters during saturated conditions, well samples were preferred. A general design schematic of a solution lysimeter in the field is shown in Figure 8.

In order to obtain a water sample, residual liquid was first discarded. A stopper which had plastic tubing attached through the center was used to seal the lysimeter in order to create a vacuum within the lysimeter. The vacuum was created by using a hand pump. A vacuum was left on the lysimeter for approximately 0.5 hours which normally was enough time for water to collect in the lysimeter. Samples were then drawn by use of a hand pump.

### ***Sample Wells***

Sample wells were installed at 60 and 150 cm depth below the soil surface at both sites. Wells at the 60 cm depth were placed along side the suction lysimeters in all the sampling areas as shown in Figures 6 and 7. Two additional sample wells at 150 cm depth were placed in each sampling area. There was a total of 54 wells at each field site. These wells were placed at the top and bottom edges of the spray circles. Sampling wells were constructed of 3.8 to 5 cm, schedule 40 PVC pipe with 0.63 cm holes perforating two opposite sides of the lower 15 cm of the pipe. The holes were covered with fiberglass screening and a PVC solid cap was attached to the bottom which created a small reservoir of approximately 50 mL. A 6 cm hole was augured to the desired depth and the well placed in it. Coarse sand was added around the bottom of the sampler to cover

the holes. Soil was then packed around the well to within 2-6 cm of the surface. Cement was poured around the samplers to insure sealing. Bentonite was used to seal around the top of the sampler. Sleeves made from PVC were placed at the top of the wells such as they were for the suction lysimeters. Sample wells were also cut flush to the soil surface in order to aid in mowing the grass cover. A general design schematic of a well sampler in the field is shown in Figure 8. Water samples were obtained by first discarding any residual water and then allowing the well to refill. Samples were then taken by using a hand pump.

### **Soil Moisture Measurements**

The soil moisture status as affected by spray irrigation of effluent was measured by tensiometers and a neutron moisture meter. Tensiometer readings which measures the soil matric potential, were taken monthly at both sites. Soil moisture content measurements were taken periodically using the neutron moisture meter.

### ***Tensiometers***

Tensiometers were placed at a depth of 30 cm adjacent to suction lysimeters and sample wells within the sampling areas as demonstrated in Figures 6 and 7. The tensiometers were constructed from lengths of 1.2 cm electrical conduit (PVC) fitted with a 100 kPa standard ceramic cup on one end and with an approximately 8 cm length of clear acrylic tube on the other end. The tensiometers were filled

with a 50/50 mix of water and ethylene glycol in order to ensure against freezing. A rubber septum was inserted into the acrylic tube to create a sealed system. A screw auger was used to open a hole to the desired depth. A slurry of silica flour was added to the hole in an amount sufficient to cover the ceramic cup. Soil was used to back fill the hole to the surface and the surface was sealed with bentonite. The tensiometer was kept flush with the soil surface in order to simplify mowing the grass cover. A sleeve made from 5 cm PVC with a length of 7 cm was placed over the top of the tensiometers to protect it from routine activities. The sleeve was cemented into the ground and then capped. The moisture tension was measured with a Soil Measurement Systems brand tensiometer. Thirty-six tensiometers were utilized at each field site.

### ***Neutron Moisture Meter***

A neutron probe moisture meter was utilized to obtain soil moisture measurements to a depth of 150 cm in the soil profile. The meter used was a Troxler Electronic Laboratories model 2601 which consists of a scaler module and a depth moisture gauge (model 1255). The moisture gauge contains a source of fast neutrons which are emitted from Americium-Beryllium at a rate of 100 millicuries ( $2.5 \times 10^5$  neutrons per second). Access tubes were constructed from aluminum irrigation pipe with a 5 cm inside diameter. The access tubes were placed by auguring holes to a depth of 150 cm with a Giddins soil coring machine. The tubes were sealed with rubber stoppers on both ends. The tops of the tubes

were cut flush to the soil surface and approximately 5 cm of bentonite was applied around the top of the tube to prevent water short cutting down the side of the tube.

Four access tubes were placed in each sampling area as shown in Figures 6 and 7. A total of 36 access tubes were installed at each site. Moisture measurements were made at the 60, 90, 120, and 150 cm depths in each tube. Readings from the scaler were then converted to gravimetric units ( $\text{g H}_2\text{O g}^{-1}$  soil) based on the soils' bulk density. Readings were taken on 9/5/90, 11/7/90 and 2/5/91 at the Montgomery Co. site and on 11/15/90 at the Alleghany Co. site.

### **Runoff Water Collection**

Determining the quality of runoff from the spray areas was necessary in evaluating the performance of slow rate spray irrigation systems. The system was designed so that runoff from an irrigation event was not expected. Low runoff was accomplished by establishing loading rates based on the most limited month from the hydraulic budget. The percolation component within the hydraulic budget was chosen to be a conservative value (7% of hydraulic conductivity or permeability of the most impermeable soil horizon) in order to prevent runoff. The low slopes (<5%) at the site also limit runoff. However runoff across the spray areas might occur during a storm event. Storm water runoff samples were collected from the sheet flow across the spray field. Each spray circle contains one or two runoff collecting strips which served to assess the water quality which ran across the surface through the irrigated areas (Figures 6 and 7). Strips of aluminum flashing

(10 cm width) were inserted into the topsoil to form a barrier to contain runoff in a rectangular area which measures 1.5 m by 17 m and started at the top of the spray circle and then collected the samples 4.6 m down slope of the spray circle perimeter.

An edging machine was used to place a 5 cm deep slit into the top soil. The flashing was placed edgewise into the slit and granular bentonite was poured on both sides of the flashing in order to make a seal that would prevent up slope sheet flow from entering the runoff strip. The flashing at the bottom of the strips tapered off into a cone shape in order to direct the water to a collection bowl. The collection bowl was made from concrete and contained a drain of PVC pipe which lead to a collection barrel. A 60 L plastic barrel was placed below the soil surface at the bottom of each runoff strip for water collection. Runoff samples were taken after storm water or snow melt events. Subsamples were taken and the volume collected measured from 10 runoff events over a 20 month period at the Montgomery Co. site. Runoff samples were not collected from the Alleghany Co. site since quick sample preservation immediately after a runoff event was hampered by the travel distance to the site.

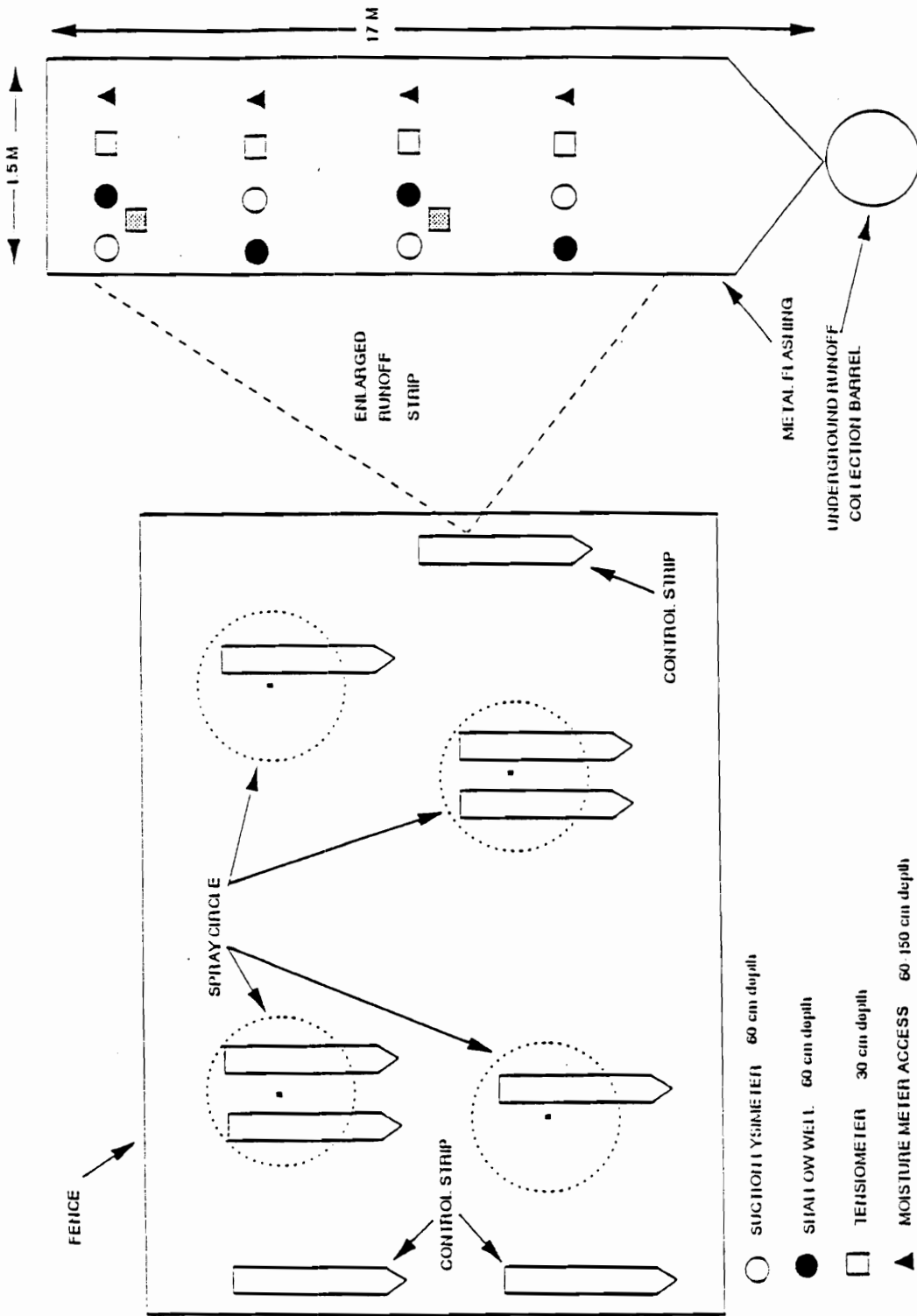


Figure 6. Monitoring equipment layout in the Montgomery Co. site.

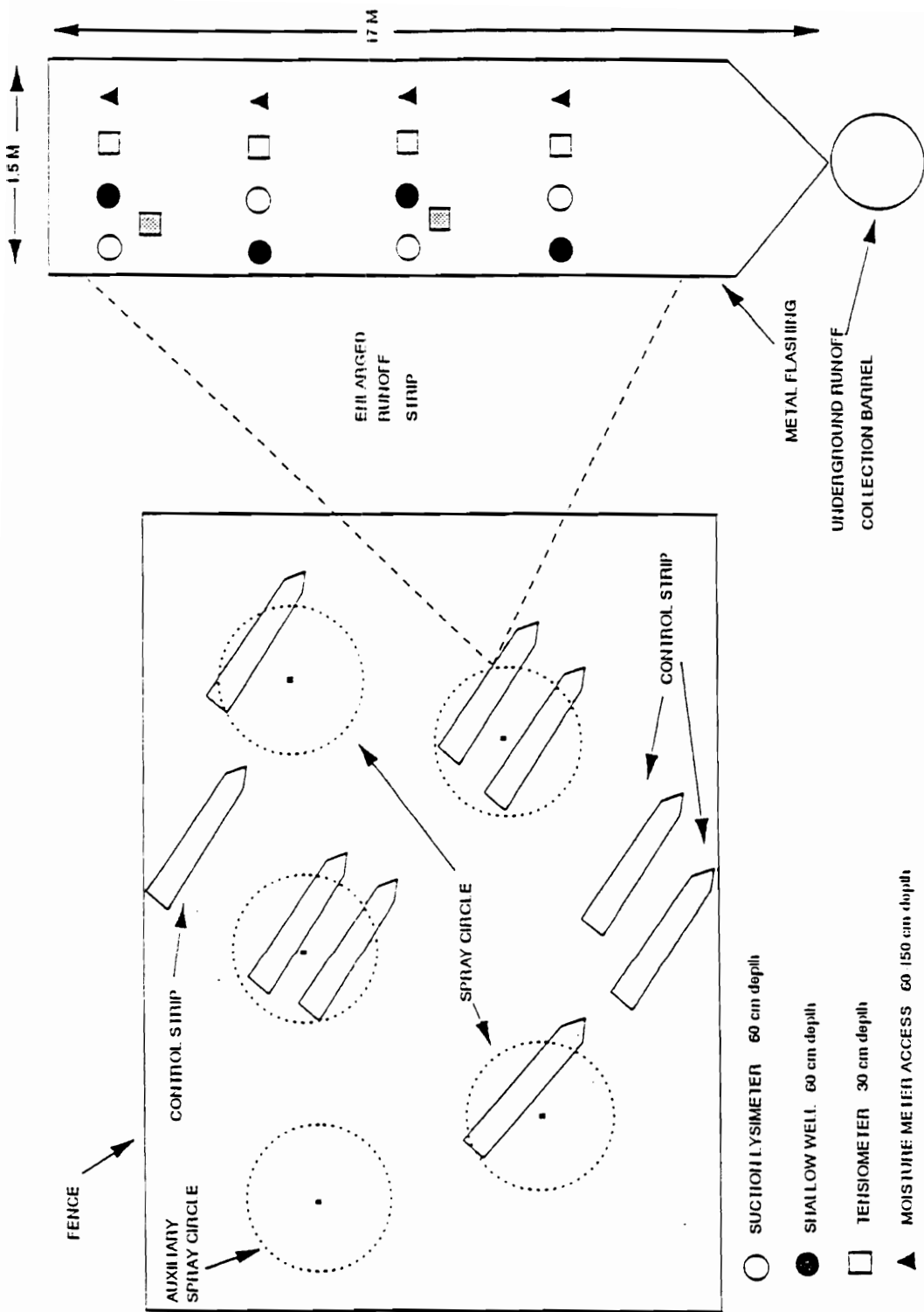


Figure 7. Monitoring equipment layout at the Allegheny Co. site.



### **Plant Tissue Collection**

A tall fescue cover was maintained at both sites. Both sites received one application of 2,4-D herbicide to control broadleaf weeds before effluent was initiated. Tissue samples of the tall fescue cover were collected at both sites within the sampling areas by using a mower with a bag attachment. Samples were collected 7 times at the Montgomery Co site and 5 times at the Alleghany Co. site. Sample collection frequency was based on growth of the vegetative cover. Approximately 4 m<sup>2</sup> was harvested from each runoff strip for determination of yield and N composition. Weight measurements for yield and subsamples for tissue analysis and moisture content were collected.

### **Soil Sample Collection**

Soil samples were collected from the top 15 cm of the profile periodically over a 10 month span at the Montgomery Co. site. Soil samples were taken in conjunction with denitrification measurements. Core samples were taken using a 3 cm diameter hand soil probe. At each date 6 cores were taken within each spray circle and 3 cores were taken from each control plot. The cores were taken at the outer edge of the spray circles since the majority of the effluent was applied there. After denitrification measurements were performed, the samples were measured for moisture, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>+</sup>-N content.

In November 1990, soil samples were collected through the entire profile at both sites. The samples exemplify 18 and 10 months of effluent application at the

Montgomery and Alleghany Co. sites, respectively. Five cm diameter cores were collected using of a tractor attached hydraulic press (Giddings soil coring machine). Four cores were taken within each sampling strip giving a total of 36 cores in each site for chemical analysis. Cores were then divided into increments of 0-15, 15-30, 30-45 and 45-60 cm depths. Eighteen additional cores, of the same depth increments used for chemical analysis, were taken in each site for measurements of hydraulic conductivity and bulk density.

### **Field Denitrification Measurements**

Denitrification measurements were based on the acetylene blocking technique (Tiedje, 1982). Acetylene blocks the final transformation of  $N_2O$  to  $N_2$  by inhibiting the enzyme nitrous oxide reductase. This procedure provides an estimate of the actual field denitrification rates. Denitrification measurements were taken monthly at the Montgomery County site between February, 1990 and May, 1990. A method similar to that used by Myrold (1988) was employed to measure  $N_2O$  emission. Soil cores from the top 15 cm of the profile were taken with a 3 cm diameter tube sampler and were placed in 50 mL test tubes and an air-tight rubber septum was inserted into the opening of each test tube. On each date, 6 cores were taken within each spray circle and 3 cores were taken from each control plot. The cores were taken at the outer edge of the spray circles since the majority of the effluent was applied there. Ten milliliters of gas were removed with needle and syringe and replaced with 10 mL of acetylene gas ( $C_2H_2$ ). The test tubes were

then placed back into the soil for 24 hrs in order to equilibrate to soil temperatures in the field. The soil temperature was recorded with a max-min thermometer. The samples were subsequently brought to the lab for immediate analysis. One-half milliliter gas samples were removed from each test tube and analyzed for the presence of  $N_2O$ . The analysis was made on a Varian 3700 Gas Chromatograph equipped with an electron capture detector. A glass 2.0 m column pre-packed with Porapak-Q was used with an inlet temperature of  $60^\circ C$ , a column/oven temperature of  $50^\circ C$ , and a detector temperature of  $340^\circ C$ . Argon gas (Ag) with 10% methane ( $CH_4$ ) was used as the carrier. The peaks were integrated using a 3390A Hewlett Packard Integrator. Good separation of the  $N_2O$  peak had been observed under these conditions with a retention time of approximately 1.3 minutes.

After denitrification analysis the head space within the test tubes was determined in order to quantify total  $N_2O$  emissions from the soil. The pressure inside the test tubes was first equalized with atmospheric pressure. Ten mL of air was then injected into the test tubes with a needle and syringe. The pressure change within the test tubes were then measured with a pressure transducer.

The amount of head space was then determined from an equation applying the ideal gas law:

$$P_1 * V_1 = P_2 * V_2 \quad (4)$$

where  $P_1$  = atmospheric pressure  
 $V_1$  = volume of headspace plus injected volume  
 $P_2$  = pressure after injection  
 $V_2$  = volume of head space

Subsamples were taken from each sample for moisture content determinations. The soil weight in each sample was also recorded. Procedures for the moisture determinations are the same as referred to in the soil analysis material and methods. Calculations of the denitrification rate required the determination of total soil weight, the amount of  $N_2O$  solubized in the soil water and test tube head space. Calculations were based on the following equation.

$$N_2O-N = G * (H + [W * \alpha]) \quad (5)$$

where :

$N_2O-N$  ( $\mu\text{g g}^{-1} \text{ day}^{-1}$ )  
 $G$  =  $N_2O-N$  measured ( $\mu\text{g mL}^{-1}$ )  
 $H$  = head space in test tube (mL)  
 $W$  = water volume (mL)  
 $\alpha$  = bunsen coefficient for  $N_2O$  adsorption in  $H_2O$

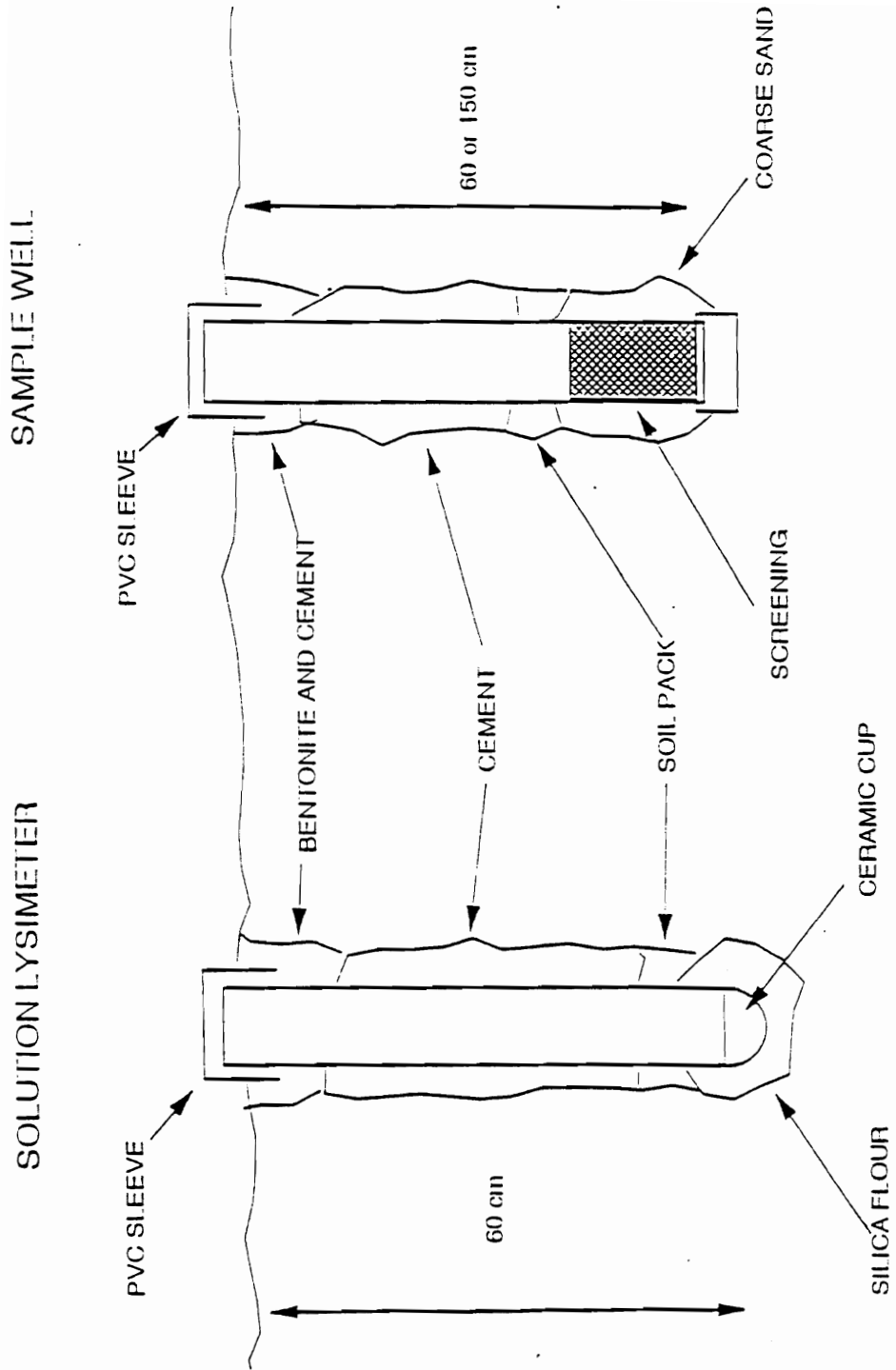


Figure 8. Design schematic of suction lysimeters and samples wells in the field.

## WATER ANALYSIS

### Nitrate and Ammonium

Water samples were first filtered with a vacuum micro-pore filter using cellulosic, 0.45 micron filter paper. Three mL aliquots were then pipetted to plastic sample cups for analysis. Analysis was done colorimetrically with an Orion Scientific dual channel auto-analyzer. Nitrate-N and  $\text{NH}_4^+$ -N were determined simultaneously on the analyzer. The  $\text{NO}_3^-$ -N procedure utilized a copper-cadmium reduction column to reduce  $\text{NO}_3^-$ -N to  $\text{NO}_2^-$ -N. The  $\text{NO}_2^-$ -N ions reacted with sulfanilamide under acid conditions to form a diazo compound. The diazo compound then reacted with N-1-naphthylethylenediamine dihydrochloride to form a reddish-purple azo dye (APHA, 1985). Two reagents were mixed for the  $\text{NO}_3^-$ -N analysis procedure. The first reagent was composed of 85 g  $\text{NH}_4\text{Cl}$  diluted to 1 L. The second reagent contained 10 g sulfanilamide, 100 mL concentrated  $\text{H}_3\text{PO}_4$ , and 1 g N-1-naphthylethylenediamine, all diluted to 1 L. The  $\text{NH}_4^+$ -N procedure involved alkaline sodium salicylate and hypochlorite reacting with  $\text{NH}_4^+$ -N to form an indo-blue color dye (APHA, 1985). The blue color was intensified by sodium-nitroprusside. Two reagents were also mixed for the  $\text{NH}_4^+$ -N procedure. The first reagent consisted of 80 g sodium salicylate, 60 mL of 50% sodium hydroxide, and 0.3 g sodium nitroprusside, all diluted to 1 L. The second reagent was comprised of 23 mL sodium hypochlorite, and 6.5 g sodium hydroxide, all diluted to 1 L. A standard curve of 0.2 to 1.0  $\text{mg L}^{-1}$  was used for both  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N. A check of known concentration from the U.S.EPA Quality Assurance Research

Division was also used. Samples measuring outside the range of standards were diluted by an automatic dilutor.

### **Ortho-phosphates**

Ortho-phosphates in solution were determined using an ascorbic acid colorimetric procedure. An antimonyphospho-molybdate complex is formed in the sample which turns a blue color with the addition of ascorbic acid (U.S.EPA, 1979). A spectrophotometer is used to measure the intensity of blue color in the samples. Water samples were first filtered through a vacuum millipore filter (0.45 microns) and then placed in 50 mL test tubes. Dilutions were made on samples which were expected to have high concentrations. A reagent mix was prepared immediately before analysis. A 100 mL volume of reagent mix consisted of 50 mL of 5N H<sub>2</sub>SO<sub>4</sub>, 5 mL potassium antimonyl tartarate solution, 15 mL ammonium molybdate solution and 30 mL ascorbic acid. Eight mL of reagent mix was added to 50 mL of sample and mixed thoroughly. Measurements on the spectrophotometer were read within 25 minutes. A standard curve was prepared from a solution of potassium phosphate in concentrations of 0.2 to 1.0 mg L<sup>-1</sup>. A check of known concentration from the U.S.EPA Quality Assurance Research Division was also used.

### **Chlorides**

Water samples were first filtered by a vacuum millipore filter with 0.45

micron filter paper. The analysis procedure utilized potentiometric titration performed with a digital chloridometer (APHA, 1985). A diluted sample was combined with an acid solution and a gelatin reagent before analysis. The acid solution consisted of 6.4 mL concentrated nitric acid and 100 mL glacial acetic acid and diluted to 1 L. The gelatin reagent consisted of 60 g Knox unflavored gelatin No 1., 1 g thymol blue and 1 g thymol diluted to 1 L. The reagent was then heated to form a gel. Three mL of diluted sample was combined with 1 mL acid solution and 4 drops gelatin reagent into a glass vial for automatic titration by the chloridometer. A standard curve was prepared from a known standard supplied by the manufacture of the chloridometer.

### **Total Kjeldahl Nitrogen**

Samples for TKN analysis first undergo digestion. Where organic components of N are broken down to  $\text{NH}_3$  for analysis. For digestion 10 mL of shaken sample are placed in digestion test tubes with 3 mL of concentrated  $\text{H}_2\text{SO}_4$  and 1 g catalyst mixture. Digestion test tubes were placed in a heating block at  $380^\circ\text{C}$  for 3 h. The samples were allowed to cool and then diluted to 50 mL with distilled  $\text{H}_2\text{O}$ . The samples were vortexed and mixed thoroughly. Aliquots were taken for  $\text{NH}_3$  analysis. Ammonia analysis was performed by either an automated colorimetric procedure or by steam distillation and titration. The automated colorimetric procedure uses the same reaction in the  $\text{NH}_4^+\text{-N}$  water analysis except samples are run in an acid matrix and a working buffer of sodium



phosphate and sodium hydroxide. The steam distillation-titration procedure utilizes a rapid distillation apparatus. An aliquot of the digested sample is heated with a strong alkali solution (50% NaOH) to volatilize  $\text{NH}_3$  which is then condensed into a boric acid indicator solution. The boric acid - indicator solution consisted of 16 g boric acid with 16 ml of bromcresol green and methyl red indicators diluted to 1 L. The receiving solution was then titrated with either 0.005 or 0.01 N  $\text{H}_2\text{SO}_4$  depending on N content (U.S. EPA, 1979, Method 351.2).

### **pH**

Water pH was measured by a glass electrode paired with a reference electrode. Standard buffer solutions of pH 4.0 and 7.0 were used to calibrate the pH meter (U.S. EPA, 1979, Method 150.1).

### **Electric Conductivity**

Electric Conductivity was measured by a conductivity bridge. Samples were measured against a standard solution of 0.01 N KCl (U.S. EPA, 1979, Method 120.1).

## **SOIL ANALYSIS**

### **Profile Characterization**

The soil profile characterizations at both sites were based on cores collected from a hand auger and from a Giddins soil coring machine. Nine profile cores were collected from each effluent application rate. Within each spray circle,

cores were collected from the top, middle and bottom of the circles. Descriptions of horizon color, depth and structure were performed on each core. Subsamples from depth of increments of 0-15, 15-30, 30-45 and 45-60 were collected from each soil core. Particle size analysis of the subsamples were performed using the pipette method modified from Day (1965).

### **Nitrate and Ammonium**

Nitrate-N and  $\text{NH}_4^+$ -N were extracted from soil samples with 2 N KCl. Approximately 15 g of soil was weighed and placed into plastic shaker bottles with 150 mL of 2 N KCl. The soil and extractant were shaken for 1 hour on an automatic shaker. Samples were allowed to settle and an aliquot of supernatant was collected with a pipette. Nitrate-N and  $\text{NH}_4^+$ -N was determined by a colorimetric autoanalyzer using the same procedure as in water analysis except in a KCl matrix. Concentrations were corrected for water content in the sample and based on a soil dry weight basis (U.S. EPA, 1979, Methods 353.2 and 350.1).

### **Chloride**

Chloride was extracted from soil samples with 0.5 N ammonium nitrate ( $\text{NH}_4\text{NO}_3$ ). Approximately 5 g of soil was weighed and placed in a shaker bottle with 50 mL of  $\text{NH}_4\text{NO}_3$  solution is added. The samples were shaken for 1 hr. An aliquot of supernatant was pipetted out of the shaker bottles for  $\text{Cl}^-$  analysis.

Chloride analysis was performed on a digital chloridometer using the same procedure used for water  $\text{Cl}^-$  analysis modified from Adriano and Doner (1982).

### **Total Kjeldahl Nitrogen**

Soil TKN was determined using the same procedure as water TKN except using 0.5 g of soil instead of water samples.

### **pH**

Soil pH was determined in a 1:1 soil to water ratio with a glass electrode (McLean, 1982).

### **Carbon**

Soil C was determined by a combustion method using a Leco Carbon Determinator. Approximately 0.5 g of soil was placed on an auto-balance within the instrument and then in a combustion oven. Volatilized  $\text{CO}_2$  was measured by a detector and reported as %C (Nelson and Sommers, 1982).

### **Phosphorus and Cations**

Soil samples were extracted with Mehlich No. 1 extractant (0.05 N HCl in 0.025 N  $\text{H}_2\text{SO}_4$ ). Two grams of soil were combined in shaker bottles with 20 mL of extractant. The samples were shaken for 10 minutes and filtered through a Whatman No. 1 filter paper. All elements (P, K, Mg, Ca, Mn and Zn) were

analyzed in the same extractant by ICP (Inductively Coupled Plasma spectrometer) by the VPI&SU Soil Testing Lab (Olsen and Dean, 1965; Mehlich, 1953).

### **Hydraulic Conductivity**

Saturated hydraulic conductivity was measured in the vertical direction using a constant head method (Klute and Dirksen, 1986). The bottom of each core was wrapped in cheesecloth and then saturated from the bottom up to remove trapped air. A inverted 500 mL volumetric flask was used as a source of constant head above each core. Volumes of water that percolated through the cores were recorded for each core until the water volumes were constant.

## **PLANT TISSUE ANALYSIS**

Samples were ground to a fine size (1.0 mm) by a plant tissue grinder. Subsamples of 0.20 g were first digested (390 ° C) with 36 N H<sub>2</sub>SO<sub>4</sub> and Kjeldahl catalyst for 3 hours. After cooling samples were diluted to 50 mL with D.D H<sub>2</sub>O. Nitrogen determinations were made with either automated phenol blue procedure (U.S.EPA, 1979) or the steam distillation procedure (U.S.EPA, 1979).

Nitrogen analysis was performed after kjeldahl digestion by standard procedures and determined by either a Micro-Kjeldahl distilling apparatus followed by titration of the boric acid indicator with dilute sulfuric acid or by the colorimetric indophenol method (U.S. EPA, 1979).

## STATISTICS AND CALCULATIONS

Both experimental field sites were analyzed as completely random design experiments. Three treatments were compared in each analysis. Treatments included two effluent application rates and a control which received no effluent application. Each treatment in the field contained three replications. Duncan's multiple range test ( $P \geq 0.05$ ) was used when applicable for testing between means.

Calculations for the determination of effluent dilution in surface and subsurface waters were based on the composition of  $\text{Cl}^-$  in these waters. Equations considering the fraction of the sample volume which is native groundwater and a second equation considering the concentrations of  $\text{Cl}^-$  in applied effluent and in water samples were solved simultaneously (Stewart and Reneau, 1988). The equations solved simultaneously are:

$$X + Y = 1 \quad (6)$$

$$C_1 X + C_2 Y = C_3 \quad (7)$$

where:

X is the fraction of sample volume that is composed of native groundwater,  $C_1$  is the surface or ground water  $\text{Cl}^-$  concentration, Y is the fraction of the sample volume that is applied effluent,  $C_2$  is the applied effluent  $\text{Cl}^-$  concentration and  $C_3$  is the  $\text{Cl}^-$  concentration of the sample. Predictions of N and P concentrations in

surface and subsurface waters were based on the effluent dilutions determined by solving Eqs 6 and 7.

## RESULTS AND DISCUSSION

### MOISTURE STATUS

Changes in soil matric potential ( $\Psi_m$ ) at the 30 cm depth were observed at the two sites throughout the study (Figures 9 and 10). At the Montgomery Co. site, the soil  $\Psi_m$  was lower in the control than in the irrigated plots in May 1990. This was due to low rainfall at this site. Soil  $\Psi_m$  measured during the other periods showed only small differences in the soil  $\Psi_m$  between the control and irrigated plots. The control plots had lower  $\Psi_m$  than the irrigated plots when all measurements were considered (Table 13). These data also indicate a trend of higher  $\Psi_m$  with increased irrigation. At the Alleghany Co. site, there was no difference in  $\Psi_m$  between the control and irrigated plots when all measurements were averaged (Table 13). However, there is a trend toward higher  $\Psi_m$  with increased effluent irrigation. As in the Montgomery Co. site, the largest differences in  $\Psi_m$  between irrigated and non-irrigated rates occurred in the summer (June, 1990) when the soil was dry (Figure 9). However, unlike the Montgomery Co. site, only the high irrigation rate had a lower  $\Psi_m$  than the control rate.

It is important to note that the time of the year had a larger effect on the  $\Psi_m$  than did effluent irrigation. In the dry months during the summer,  $\Psi_m$  decreased in both the irrigated and control plots, while in the winter,  $\Psi_m$  in both irrigated and non irrigated plots were close to zero (saturated soil conditions). Even though the

soil was near saturation at the 30 cm depth in the winter, no surface ponding of effluent was observed at either site. Overall effluent irrigation had little apparent effect on moisture conditions at the 30 cm soil depth. This would be expected since the loading rates were low and the upper part of the soil profile would be more susceptible to changes in the  $\Psi_m$  due to climate effects.

The moisture status deeper in the soil profile was measured with a neutron moisture meter (Table 14). Moisture contents at the Montgomery Co. site are means from measurements made in September and November 1990 and February 1991 (Figure 11). The soil moisture content increased, to the 90 cm depth, with increased effluent application. At depths greater than 90 cm, there was no trend between moisture content and effluent applications. The shale layer was located at the 80 cm depth and the percolation of the effluent below this layer may have been restricted. This probably accounts for the lower soil moisture contents present in both the control and the 1.25 cm wk<sup>-1</sup> irrigation rates. There was a trend of higher moisture contents below the shale layer at the higher irrigation rate.

Soil moisture content under the high, low and control rates were similar to the 90 cm depth at the Alleghany Co. site (Figure 12). Deeper in the soil profile, at the 120 and 150 cm depths, there were increases in soil moisture content with increased effluent loading. At the Alleghany Co. site, the soil had fragic characteristics near the 80 cm depth, however, the permeability must not have been as restrictive as the shale layer at the Montgomery Co. site. The moisture content curves demonstrate that deeper percolation of the effluent occurred at the



Alleghany Co. site, which was attributed to coarser soil textures, a deeper soil profile and more permeable soil horizons (Tables 3 and 4).

Table 13. Soil matric potential (30 cm depth) at both spray irrigation sites.

Rate	Montgomery Co.†	Alleghany Co.‡
cm wk <sup>-1</sup>	-----kPa-----	
0	-109b*	-64a
1.25	-53a	-61a
2.5	-47a	-51a

\* Means followed by the same letter for each site are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† mean of 7 sampling dates from Feb 1990 to Feb 1991.

‡ mean of 5 sampling dates from Feb 1990 to Jan 1991.

Table 14. Moisture content with depth under effluent irrigation treatments at both field sites.

Depth	Effluent Loading Rate (cm wk <sup>-1</sup> )		
	0	1.25	2.5
cm	-----g H <sub>2</sub> O g <sup>-1</sup> soil-----		
	-----Montgomery Co. site-----		
60	0.27 c*	0.29 b	0.30 a
90	0.27 b	0.28 b	0.31 a
120	0.23 b	0.21 c	0.25 a
150	0.20 ab	0.19 b	0.21 a
	-----Alleghany Co. site-----		
60	0.24 a	0.25 a	0.24 a
90	0.27 a	0.27 a	0.27 a
120	0.25 a	0.26 a	0.26 a
150	0.24 b	0.25 a	0.26 a

\* Means in the same row followed by the same letter are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

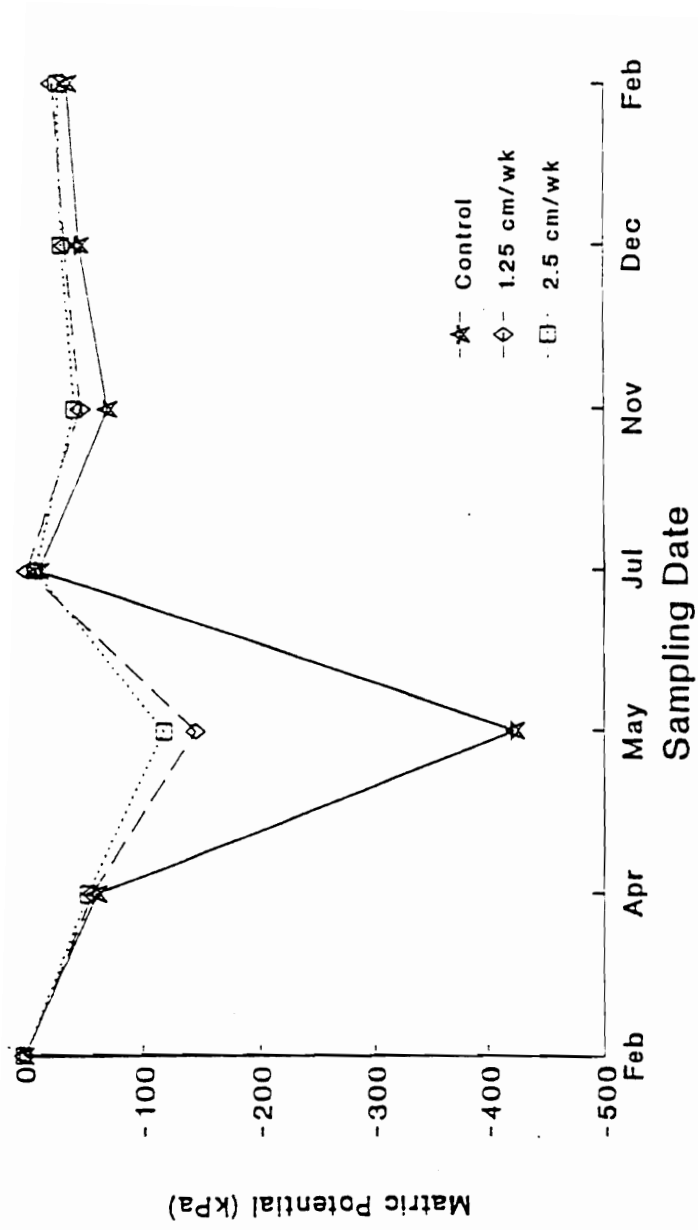


Figure 9. Soil matric potentials from February 1990 - February 1991 at the Montgomery Co. site.

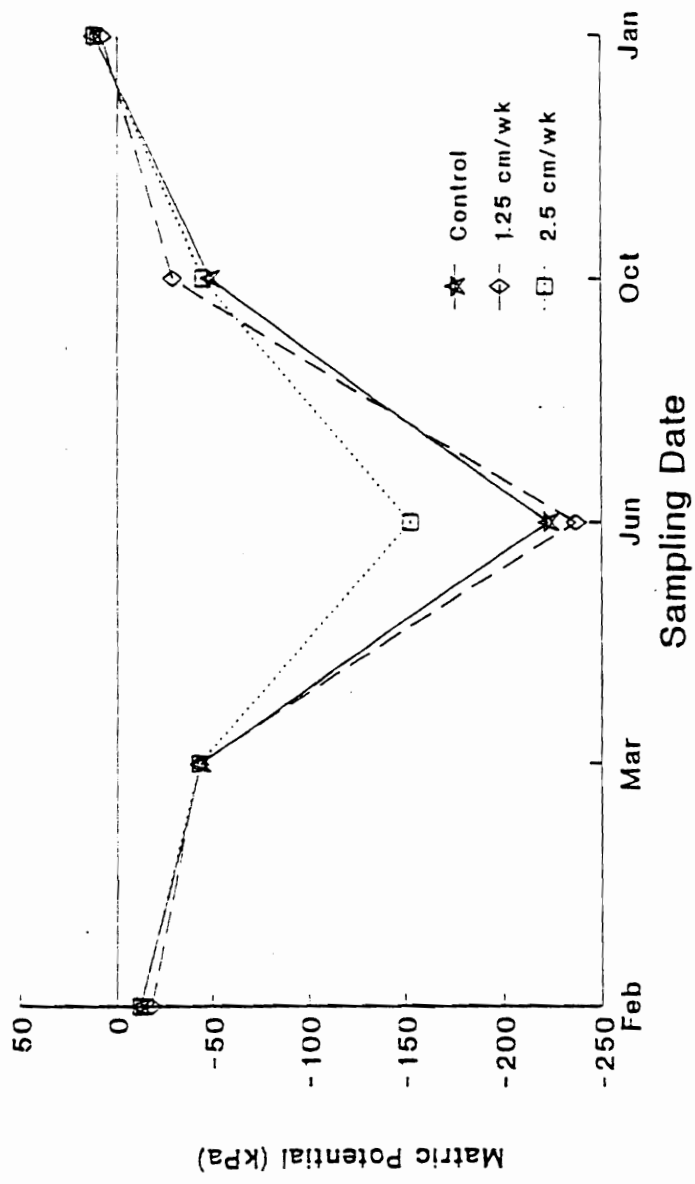


Figure 10. Soil matric potentials from February 1990 - January 1991 at the Alleghany Co. site.

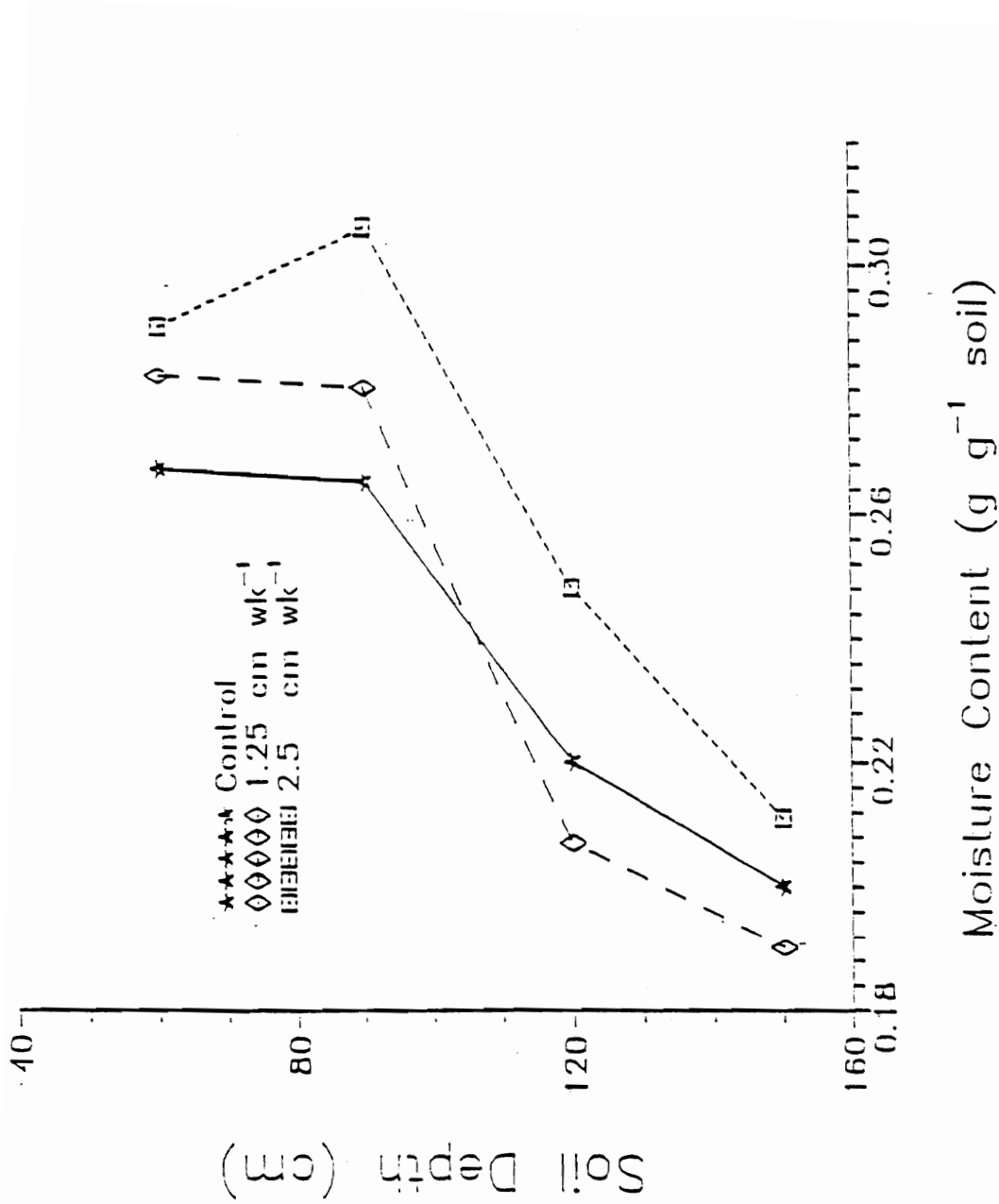


Figure 11. Soil moisture content with depth as measured by neutron moisture meter, Montgomery Co. site.

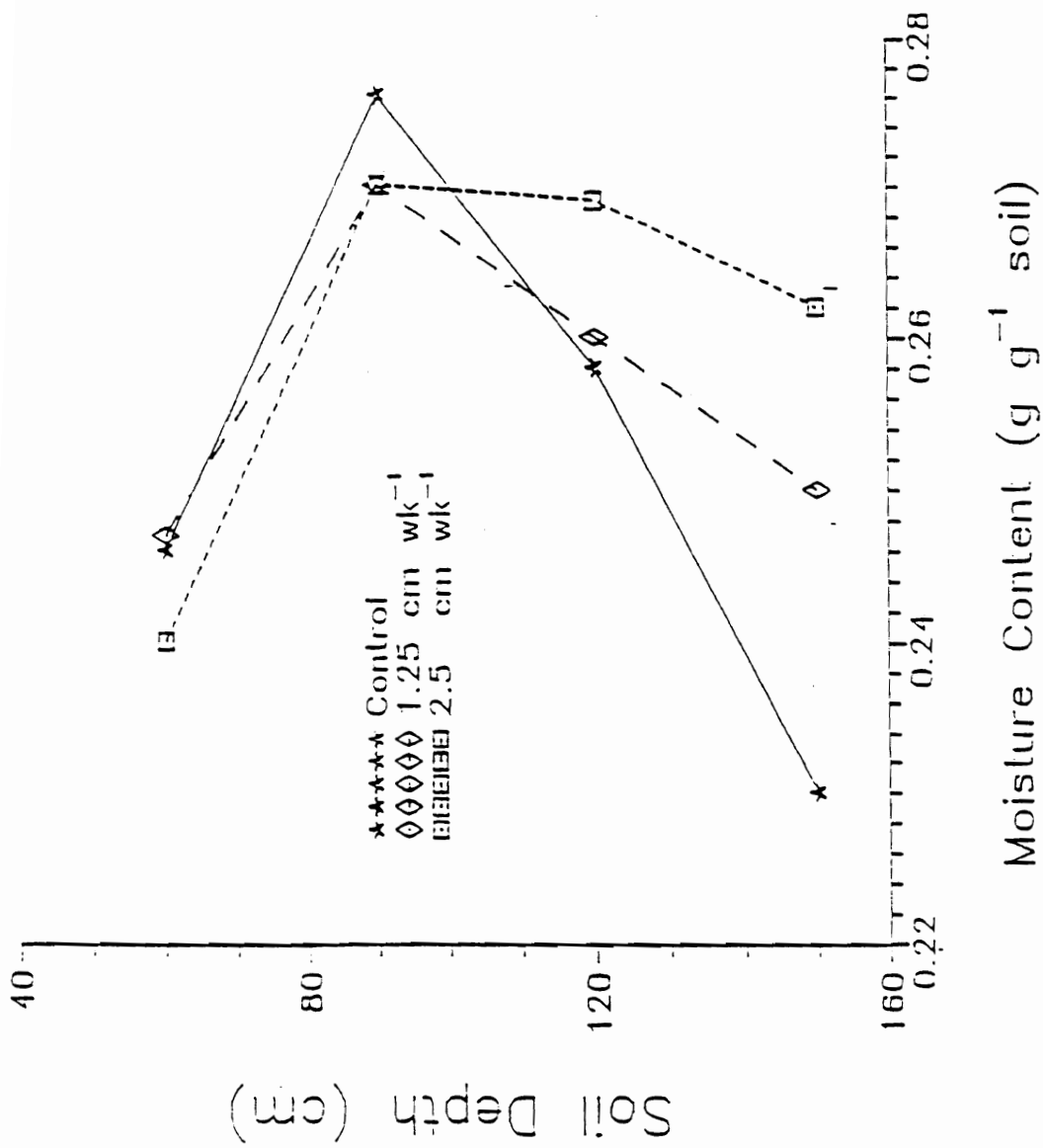


Figure 12. Soil moisture content with depth as measured by neutron moisture meter, Alleghany Co. site.

## SUBSURFACE WATER QUALITY

### Chloride

Subsurface water quality was monitored at the 60 (Tables 15 and 16) and 150 cm (Tables 17 and 18) depths at both the Montgomery and Alleghany Co. sites. Chloride ( $\text{Cl}^-$ ) concentration is a good indicator of effluent movement since it is highly mobile in the soil systems and undergoes no biochemical transformations and limited soil adsorption. Chloride concentrations increased in the subsurface water at the 60 cm depth at both sites with application of effluent (Tables 15 and 16). At the Montgomery Co. site,  $\text{Cl}^-$  levels at the 60 cm depth increased with each increase in effluent application. Chloride concentrations in subsurface waters at the 1.25 ( $16.6 \text{ mg CL}^- \text{ L}^{-1}$ ) and 2.5 ( $29.6 \text{ mg CL}^- \text{ L}^{-1}$ )  $\text{cm wk}^{-1}$  application rates were 2 and 3 fold higher than concentrations under the control rate, respectively. Chloride concentrations were also used as an indicator in determining the percentage of effluent in the natural groundwaters. The  $\text{Cl}^-$  concentrations indicate that at the 60 cm depth 17 and 42% of the water under the 1.25 and 2.5  $\text{cm wk}^{-1}$  application rates, respectively, originated from wastewater. Wastewater dilution, at the 60 cm depth, resulted from precipitation, residual soil water, and intrusion of foreign groundwaters. Frequently after rainfall events, large amounts of subsurface water originating up slope of the spray site moved laterally on top of the shale layer through the site.

The  $\text{Cl}^-$  concentrations followed an inverse trend with the calculated



recharge percolation as a result of precipitation (Figure 13). The  $\text{Cl}^-$  concentrations at the 60 cm depth for each sampling date in the Montgomery Co. site is plotted with the calculated monthly recharge volume calculated from rainfall subtracting estimates for evapotranspiration (Figure 13). In months with a high recharge volume,  $\text{Cl}^-$  concentrations in subsurface water at both application rates decreased. In months with low recharge volumes,  $\text{Cl}^-$  concentrations increased. These data indicate that  $\text{Cl}^-$  concentrations in the subsurface water were strongly influenced by the recharge volume and or foreign groundwater. These data also show that the  $\text{Cl}^-$  concentrations under the  $2.5 \text{ cm wk}^{-1}$  rate were always higher than concentrations under the  $1.25 \text{ cm wk}^{-1}$  rate (Figure 13). Although  $\text{Cl}^-$  concentrations were different, both loading rates followed the same trend with time which indicates that the dilution by precipitation and groundwater was proportional for both application rates. Leaching percentages have been shown to be highly influenced by the interrelationship between the application rate and the evapotranspiration demands. Bole et al. (1981) determined leaching percentages of 38 and 62% from effluent irrigation application rates of 75 and 150  $\text{cm yr}^{-1}$ . Lund et al. (1981) observed leaching percentages of 72% from irrigated effluent which was applied at a rate of  $9.98 \text{ cm wk}^{-1}$  during a 12 hour period. The application rate used by Lund et al. (1981) was well above the evapotranspiration demands and thus produced high leaching percentages. The  $\text{Cl}^-$  data presented in Figure 13 agrees with Rebeck (1967) who demonstrated that seasonal changes affected subsurface movement of wastewater applied by irrigation. There was no

differences in  $\text{Cl}^-$  concentration between application rates at the 150 cm depth which suggest that effluent was not reaching the 150 cm depth in sufficient quantities to alter the groundwater quality. In the Montgomery Co. site, the shale layer, which typically occurred between 70 and 80 cm, might have been impervious to effluent. This is supported by the moisture content data which showed decreased soil moisture content at depths deeper than 90 cm.

At the Alleghany Co. site,  $\text{Cl}^-$  concentrations also increased with effluent irrigation at the 60 cm depth (Table 16). Concentrations in subsurface waters increased about 3 fold where effluent was applied as compared to the control rate. There was no difference between  $\text{Cl}^-$  concentrations for the two effluent application rates. Dilutions based on  $\text{Cl}^-$  data at the Alleghany Co. site indicate that approximately 58 and 48% of the water at the 60 cm depth originated from wastewater at the 1.25 and 2.5  $\text{cm wk}^{-1}$  application rates, respectively. Wastewater dilution at the 60 cm depth was smaller in the Alleghany Co. site than in the Montgomery Co. site. This was probably due to the less restrictive fragic horizon in the Alleghany Co. site compared to the shale layer in the Montgomery Co. site. Since the shale layer was more restrictive, a greater amount of groundwater above the restrictive layer was probably present. This is supported by field observations which indicated movement of large amounts of water laterally down slope during saturated conditions at the Montgomery Co. site. At the Alleghany Co. site  $\text{Cl}^-$  concentrations at the 150 cm depth were similar to concentrations at the 60 cm depth. However the  $\text{Cl}^-$  concentrations also increased

in the control rate and there were no differences between control and effluent rate concentrations.

At both sites the results derived from the  $\text{Cl}^-$  data were supported by the electrical conductivity (EC) measurements of the same samples. In most cases, increases in  $\text{Cl}^-$  concentrations coincided with increases in EC (Table A-2). This result would be expected since EC is partially a function of  $\text{Cl}^-$  concentration.

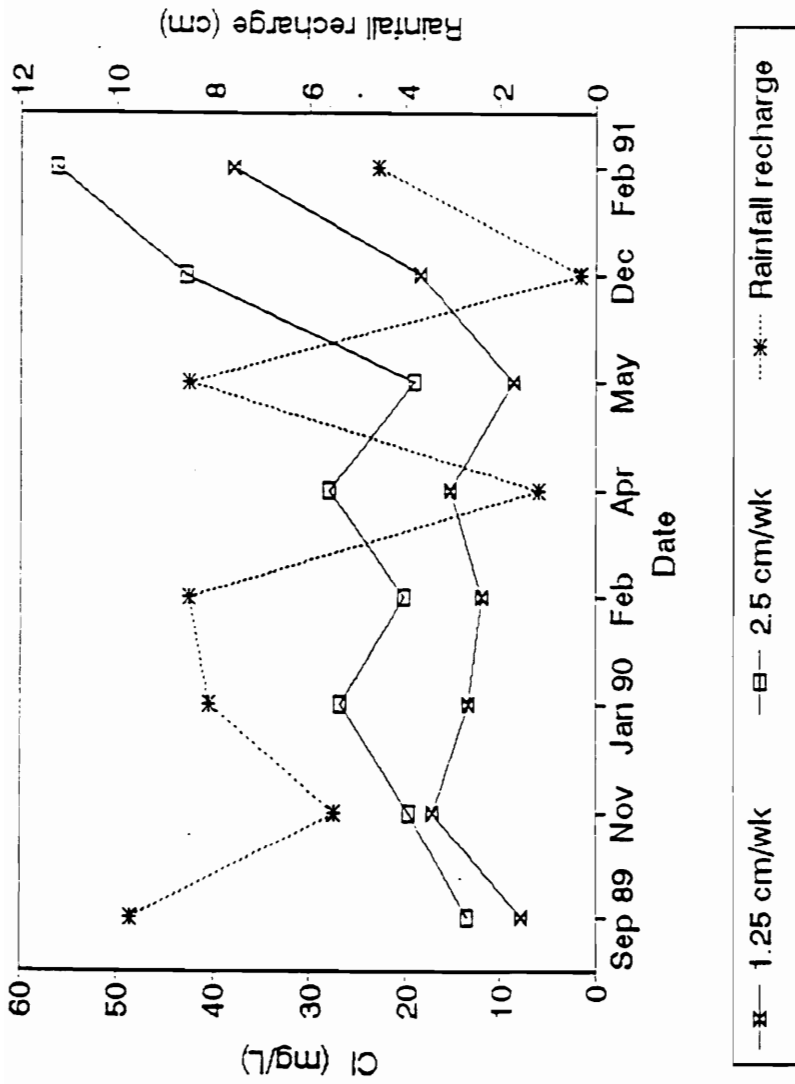


Figure 13. Chloride concentrations at the 60 cm depth and rainfall recharge volume for each sampling date at the Montgomery Co. site.

## **Nitrogen**

Nitrate concentrations also indicate that effluent irrigation had minimal effect on the subsurface water quality (Tables 15, 16, 17, 18). The concentration of  $\text{NO}_3^-$ -N in the subsurface water below the effluent irrigation systems averaged  $<0.8 \text{ mg L}^{-1}$  for both sites and both sampling depths. The  $\text{NO}_3^-$ -N concentrations at the 60 cm depth at the Montgomery Co. site were below  $0.20 \text{ mg L}^{-1}$  for the effluent loading rates (Table 15). The  $1.25 \text{ cm wk}^{-1}$  application rate had a higher concentration of  $\text{NO}_3^-$ -N than the  $2.5 \text{ cm wk}^{-1}$  rate, but was not different from the control. There was no difference between the control and highest rate of effluent application. The time of sampling did have an effect on the  $\text{NO}_3^-$ -N concentrations, however no trend with seasonal changes could be established (Figure 14). Subsurface water could not be collected from either wells or solution samplers during the summer months. The water tables decreased below the sampling well depth and the  $\Psi_m$  was too low for sample collection from the suction lysimeter. A majority of the applied wastewater probably did not travel beyond the root zone during the summer months since evapotranspiration is higher than the average rainfall (Table 7).

Taking into consideration dilution, based on  $\text{Cl}^-$  data, the estimated inorganic N concentration should be  $2.2 \text{ mg L}^{-1}$  at the low application rate and  $5.3 \text{ mg L}^{-1}$  at the high application rate. When the calculated and measured concentrations were compared, 85 and 95% of the N in the wastewater had been lost from the  $1.25$  and  $2.5 \text{ cm wk}^{-1}$  application rates, respectively (Table 19). The  $\text{NO}_3^-$ -N

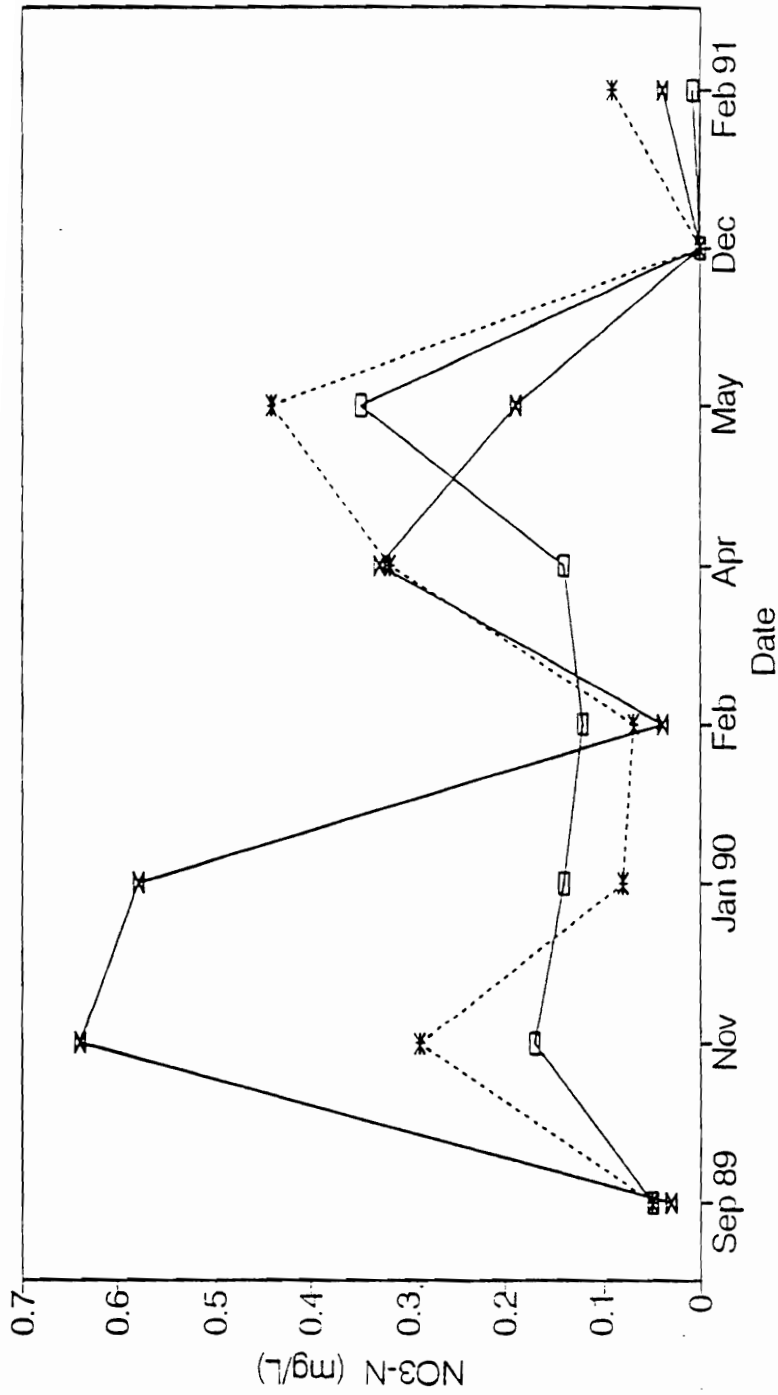
concentrations increased at the 150 cm depth compared to the 60 cm depth at the Montgomery Co. site (Tables 15 and 17). Since the  $\text{NO}_3^-$ -N concentrations were also higher under the control plots at this depth, the higher values at the 150 cm depth was attributed to existing background concentrations caused by up slope contamination sources. This is also supported by the  $\text{Cl}^-$  data which showed no differences between the effluent application and control rates at the 150 cm depth.

In the Alleghany Co. site, there was a trend of higher  $\text{NO}_3^-$ -N concentrations at the 60 cm depth under the irrigated plots compared to the control plots, however all levels were also relatively low ( $< 0.4 \text{ mg L}^{-1}$ ). There were no differences in  $\text{NO}_3^-$ -N concentrations between the control and effluent application rates at both the 60 and 150 cm depths. There was no seasonal trend in  $\text{NO}_3^-$ -N concentrations in both effluent loading rates at the Alleghany Co. site (Figure 15). Estimates of  $\text{NO}_3^-$ -N concentrations at the 60 cm depth based on  $\text{Cl}^-$  dilution indicate that concentrations should have been 25.4 and 21.1  $\text{mg L}^{-1}$  for the 1.25 and 2.5  $\text{cm wk}^{-1}$  rates of effluent application, respectively (Table 19). The differences between measured and calculated concentrations indicated that 98 and 90% N removal from aqueous solution occurred in the low and high rates, respectively (Table 19).

Ammonium-N levels in the subsurface waters were also low at both sites and depths with no difference between control and irrigated areas (Tables 15, 16, 17, 18). At the Montgomery Co. site,  $\text{NH}_4^+$ -N concentrations were less than 0.2  $\text{mg L}^{-1}$ . At the Alleghany Co. site,  $\text{NH}_4^+$ -N concentrations were 0.4  $\text{mg L}^{-1}$  except for the high rate of application at the 60 cm depth which averaged 1.67  $\text{mg L}^{-1}$ .

Low  $\text{NH}_4^+\text{-N}$  concentrations in the subsurface water was not unexpected. Ammonium most probably under went rapid nitrification during warm temperatures. In addition  $\text{NH}_4^+\text{-N}$  can accumulate in the soil during cold temperatures via the ion exchange complex.

The high N removal rates from subsurface waters is attributed to denitrification, plant uptake and soil adsorption of  $\text{NH}_4^+\text{-N}$ . These results demonstrate that N applied by effluent spray irrigation appears not to be a threat to groundwaters.



-x- 1.25 cm/wk    -□- 2.5 cm/wk    -\*- 0 cm/wk

Figure 14. Nitrate concentrations of groundwater at the 60 cm depth for each sampling date at the Montgomery Co. site.



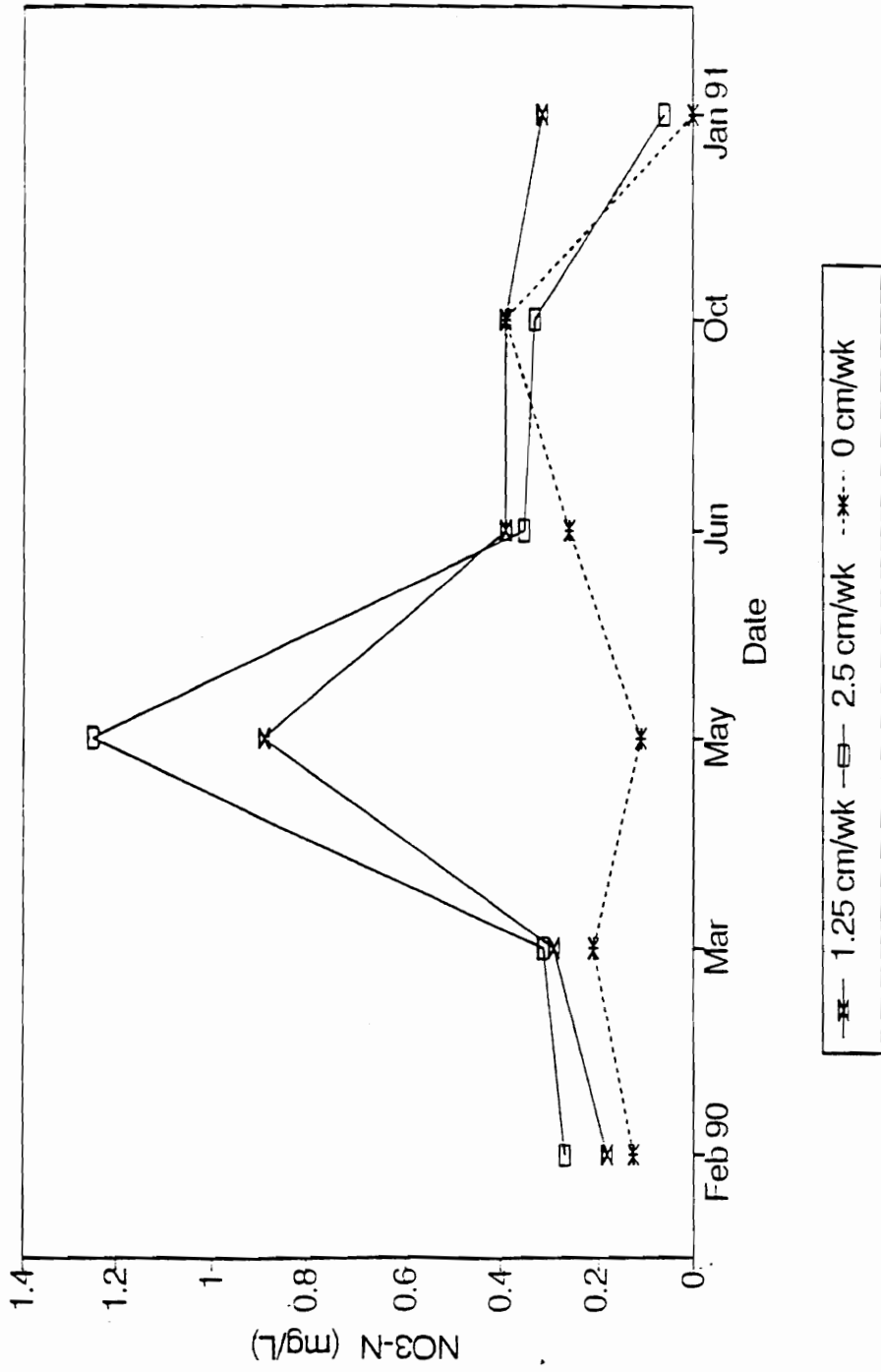


Figure 15. Nitrate concentrations of groundwater at the 60 cm depth for each sampling date at the Alleghany Co. site.

## **Phosphorus**

Ortho phosphate concentrations in subsurface waters were low ( $\leq 0.04 \text{ mg L}^{-1}$ ) at both depths at each site (Tables 15, 16, 17, and 18). This is expected and probably results from soil adsorption and precipitation reactions. There were no differences between control and irrigated rates for  $\text{PO}_4\text{-P}$  levels at either site or depth. Predicated  $\text{PO}_4\text{-P}$  concentrations based on  $\text{Cl}^-$  dilutions were 0.56 and 1.32  $\text{mg L}^{-1}$  in the Montgomery Co. site and 1.57 and 1.3  $\text{mg L}^{-1}$  in the Alleghany Co. site for the 1.25 and 2.5  $\text{cm wk}^{-1}$  application rates, respectively (Table 20). The  $\text{PO}_4\text{-P}$  removal rates based on the difference between calculated and measured values were between 96 and 98% for both loading rates and sites. These data are in agreement with reports in the literature which show up to 90% adsorption in the soil of P from surface applied effluent (Nagpal, 1985; Tare and Bokil, 1982). According to  $\text{PO}_4\text{-P}$  solubility diagrams which considers the pH and  $\text{PO}_4\text{-P}$  concentration (Lindsay and Moreno, 1960),  $\text{PO}_4\text{-P}$  in both field sites was in equilibrium with calcium phosphate compounds.

## **pH**

At the Alleghany Co. site, increased effluent application caused a stepped increase in pH of the subsurface water at the 60 cm depth. Levels of pH went from 6.4 under the control rate to 6.8 and 7.1 under the 1.25 and 2.5  $\text{cm wk}^{-1}$  effluent application rates, respectively. This result was not unexpected since the pH of the applied effluent averaged 7.1. The differences between the pH levels

in the 1.25 and 2.5 cm wk<sup>-1</sup> application rates was probably due to the total quantity of base added in the irrigation water. Even though Cl<sup>-</sup> data indicated that effluent percolated to the 150 cm depth at the Alleghany Co. site, the pH at the 150 cm depth decreased from the 60 cm depth and there were no differences between the control and irrigated treatments. The decrease in pH from the 60 to 150 depth was attributed to the acid buffering capacity of the soil. The pH levels of the subsurface waters at the Montgomery Co. site did not follow the same trend as in the Alleghany Co. site. The pH where wastewater was applied tended to be lower. This result might have been caused by the greater amount of NH<sub>4</sub><sup>+</sup>-N applied at the Montgomery Co. site compared to the Alleghany Co. site. Acidity produced as a result of nitrification of the NH<sub>4</sub><sup>+</sup>-N may have lowered the pH.

Table 15. Subsurface water quality at the 60 cm depth at the Montgomery Co. site.

Parameter		Effluent Loading Rate -----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	60	57	61
	SE‡	0.07	0.06	0.05
	Mean	6.70 a*	6.62 ab	6.53 b
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	90	79	73
	SE	0.02	0.04	0.02
	Mean	0.13 ab	0.19 a	0.09 b
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	90	79	73
	SE	0.03	0.02	0.02
	Mean	0.18 a	0.15 a	0.14 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	70	66	64
	SE	0.8	1.3	2.1
	Mean	8.2 c	16.6 b	29.6 a
PO <sub>4</sub> <sup>-</sup> -P (mg L <sup>-1</sup> )	N	69	58	66
	SE	0.010	0.003	0.009
	Mean	0.04 ab	0.02 b	0.04 a

\* Means in a row followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.

Table 16. Subsurface water quality at the 60 cm depth at the Alleghany Co. site.

Parameter		Effluent Loading Rate -----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	42	34	34
	SE‡	0.09	0.12	0.14
	Mean	6.44 c*	6.77 b	7.05 a
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	53	48	43
	SE	0.02	0.09	0.10
	Mean	0.17 a	0.37 a	0.35 a
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	53	48	44
	SE	0.04	0.08	1.45
	Mean	0.19 a	0.28 a	1.67 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	43	33	31
	SE	0.5	1.3	0.9
	Mean	2.9 b	8.8 a	7.8 a
PO <sub>4</sub> <sup>-</sup> -P (mg L <sup>-1</sup> )	N	40	34	31
	SE	0.004	0.006	0.008
	Mean	0.02 a	0.04 a	0.03 a

\* Means in a row followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.

Table 17. Subsurface water quality at the 150 cm depth at the Montgomery Co. site.

Parameter		Effluent Loading Rate -----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	10	13	6
	SE‡	0.15	0.13	0.18
	Mean	7.05 a*	6.77 ab	7.00 b
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	19	24	18
	SE	0.17	0.11	0.19
	Mean	0.64 ab	0.40 b	0.76 a
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	19	24	18
	SE	0.04	0.04	0.04
	Mean	0.18 a	0.16 a	0.17 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	13	18	13
	SE	2.6	2.1	3.9
	Mean	15.0 a	14.9 a	19.2 a
PO <sub>4</sub> -P (mg L <sup>-1</sup> )	N	11	18	8
	SE	0.007	0.006	0.016
	Mean	0.03 a	0.03 a	0.04 a

\* Means followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.

Table 18. Subsurface water quality at the 150 cm depth at the Alleghany Co. site.

Parameter		Effluent Loading Rate -----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	21	20	22
	SE‡	0.12	0.17	0.18
	Mean	6.25 a*	6.43 a	6.44 a
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	27	25	25
	SE	0.10	0.06	0.1
	Mean	0.60 a	0.50 a	0.62 a
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	26	27	25
	SE	0.06	0.17	0.05
	Mean	0.23 a	0.35 a	0.27 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	18	17	18
	SE	1.0	1.4	1.0
	Mean	4.6 b	11.0 a	9.1 a
PO <sub>4</sub> <sup>-</sup> -P (mg L <sup>-1</sup> )	N	14	17	15
	SE	0.010	0.005	0.010
	Mean	0.04 a	0.02 a	0.03 a

\* Means followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.

Table 19. N dilutions and removal rates based on measured and calculated values and N:Cl ratios at both field sites.

Loading Rate	Effluent content†	NO <sub>3</sub> -N Concentration		N Loss	
		Measured	Calculated†	M/C‡	N/Cl¶
cm wk <sup>-1</sup>	-----%----	-----mg L <sup>-1</sup> -----		-----%-----	
1.25	17	0.2	2.2	85	99
2.5	42	0.1	5.3	95	98
1.25	58	0.37	25.4	98	99
2.5	48	0.35	21.1	90	99

† Based on dilutions determined from Cl<sup>-</sup> data.

¶ N:Cl ratios based on total inorganic N.

‡ Measured/Calculated.



Table 20. P dilution and removal rates based on measured and calculated values at both field sites.

Loading Rate cm wk <sup>-1</sup>	Effluent Content† ---%---	PO <sub>4</sub> -P Concentration -----mg L <sup>-1</sup> -----		P Loss M/C‡
		Measured	Calculated†	
1.25	17	0.02	0.56	96
2.5	42	0.04	1.32	97
		Montgomery Co. site		
1.25	58	0.04	1.57	97
2.5	48	0.03	1.30	98
		Alleghany Co. site		

† Based on dilutions determined from Cl<sup>-</sup> data.

‡ N:Cl ratios based on total inorganic N.

# Measured/Calculated.

## RUNOFF WATER

The contaminant concentrations present in runoff waters collected at the Montgomery Co. site were only minimally influenced by application of effluent. The runoff quality parameter means for each treatment and each collection date are listed in Appendix Table A-3. The volume of runoff collected and the amount of precipitation causing each runoff event is also listed. The volume of runoff collected did not correlate with the amount of precipitation. High amounts of precipitation (>3 cm) in August 1989 and August and September 1990 resulted in small volumes of runoff. Dry soil conditions and large amounts of plant growth during these months are believed to be the main causes of the small volume of runoff. At the other sample dates, runoff was normally present in volumes in excess of 50 L. This was related to either snow melt or rainfall during periods when the residual moisture content was high in the soil. In November 1989, runoff from a snow melt was collected. The runoff occurred 4 days after a 5 cm snowfall. A thin layer of ice crystals from the sprayed effluent had formed on top of the snow. The quality of runoff from the snow melt and accumulated effluent was very acceptable in that total N was < 1.0 mg L<sup>-1</sup> and PO<sub>4</sub>-P was <0.1 mg L<sup>-1</sup>.

Runoff results were grouped into categories to determine if seasonal changes had an effect on contaminants present in runoff water. Data collected from April to September were grouped into a 'growing season' category (Table 21) and data collected from October to March were grouped into a 'non-growing season' category (Table 22). The overall quality of runoff from the spray circles

in both seasons was acceptable with low levels of  $\text{NO}_3^-$ -N and  $\text{PO}_4$ -P.

### **Chloride**

Higher  $\text{Cl}^-$  concentrations were present in runoff collected during the non-growing season when compared with the growing season for the control and effluent application rates (Table 23). In the non-growing season,  $\text{Cl}^-$  concentrations were higher in runoff from the high rate treatment than from the control and low rates. However, during the growing season there was no difference in  $\text{Cl}^-$  concentrations between treatments (Tables 21 and 22).

Dilution of effluent based on  $\text{Cl}^-$  concentrations indicated that during the non-growing season approximately 3.6 and 19.8% of the collected runoff water originated from effluent at the 1.25 and 2.5  $\text{cm wk}^{-1}$  application rates, respectively. During the growing season a smaller percentage of irrigated effluent, 0 and 3%, was included in the runoff waters for the 1.25 and 2.5  $\text{cm wk}^{-1}$  rates, respectively (Table 24). The EC data also showed that more effluent was contained in the high application rate runoff than the low rate for both seasons (Tables 21 and 22).

Saturated soil conditions may have also contributed to the higher quantities of effluent present in the runoff water. The  $\Psi_m$  indicated that both irrigated and non-irrigated plots were near saturation in the winter months (Table 13). Saturated soil conditions would enhance runoff water movement down slope and thus increase the proportion of effluent present in runoff water collected in the non-growing season. The highest percentage of effluent present in runoff waters was

<20%. This data reveals that a large amount of dilution occurred in the runoff water after traveling only 4.5 m beyond the bottom of the spray circles. During the growing season, infiltration of applied effluent probably increased because of drier soil conditions resulting from evapotranspiration in excess of precipitation and irrigation.

### **Nitrogen**

Nitrate and  $\text{NH}_4^+$ -N concentrations in the runoff waters were  $< 3.3 \text{ mg L}^{-1}$  in both the growing and non-growing seasons. In each season, there were no differences in inorganic N concentrations in runoff waters from the irrigated and non-irrigated control plots (Tables 21 and 22). The highest concentrations of  $\text{NO}_3^-$ -N present in the runoff waters were present in samples collected in September 1990. This sample date also had the smallest amount of rainfall (0.9 cm) (Table A-3). The  $\text{NO}_3^-$ -N concentrations in the runoff water at this date were higher than the N concentrations in the effluent. The  $\text{Cl}^-$  data indicates that there was substantial dilution of effluent in the Sept 90 runoff event. The elevated  $\text{NO}_3^-$ -N concentrations apparently originated from  $\text{NH}_4^+$ -N in the surface soil that was nitrified.

When considering effluent dilution based on  $\text{Cl}^-$  data, the calculated  $\text{NO}_3^-$ -N concentrations in the runoff waters during the non-growing season were 2.86 and 4.48  $\text{mg L}^{-1}$  at the 1.25 and 2.5  $\text{cm wk}^{-1}$  rates of application, respectively (Table 24). The difference between calculated and measured concentrations indicate N

loss of 15.7 and 90.8%, respectively, at the 1.25 and 2.5 cm wk<sup>-1</sup> effluent application rates during the non-growing season. Even though there was a large difference in N removal between the loading rates, the difference in concentrations represented only 2 mg L<sup>-1</sup> N. The amount of measured NO<sub>3</sub><sup>-</sup>-N in runoff waters collected during the growing season was higher than the calculated values for each effluent loading rate (Table 24). This increase suggests that increased activity of the topsoil microflora may have contributed additional N during the growing season. Evidence of increased microflora activity was also observed in PO<sub>4</sub>-P concentrations where there were differences between the growing season and non-growing seasons in the control treatment (Table 23).

### **Phosphorus**

There was little difference in P concentrations in the runoff waters from plots that received effluent and the control plots in both seasons (Tables 21 and 22). Phosphate concentrations in collected runoff waters were higher (0.16 mg L<sup>-1</sup>) in the 2.5 cm wk<sup>-1</sup> application rate than in the control (0.08 mg L<sup>-1</sup>) and 1.25 cm wk<sup>-1</sup> rates (0.11 mg L<sup>-1</sup>) during the non-growing season (Table 22). There were no differences in P concentrations in runoff collected during the growing season. There were also no differences in P concentrations in runoff waters between non-growing and growing seasons for both effluent loading rates. However, there was a trend of higher P concentrations in the growing season runoff waters (Table 23).

Calculations based on Cl<sup>-</sup> dilution show that <3% of the runoff volume during

the growing season for both loading rates originated from applied effluent. According to P concentrations collected from the control plots, a majority (68-100%) of the  $\text{PO}_4$  mass collected from the irrigated plots originated from native sources (Table 25). During the non-growing season, higher concentrations of P in the runoff were predicted since higher amounts of effluent were contained in the non-growing season runoff. However, P inputs into the runoff water from native sources were calculated to be less. The dilution calculations for runoff waters in the non-growing season predicated that 60-89% of the  $\text{PO}_4$ -P collected originated from effluent. Phosphorus concentrations present in the runoff waters showed that P values were 42 and 76% lower than calculated values for the non-growing season. However, measurements during the growing season show good agreement between the measured and calculated values. Since a larger content of effluent was collected in the runoff during the non-growing season, losses of  $\text{PO}_4$  originating from effluent were more easily detected. The P removal measurements in runoff during the growing season were probably influenced by the levels of P applied during the non-growing season. The removal of P most probably was caused by soil adsorption and precipitation reactions. High amounts of P adsorption could have occurred near the soil surface as the runoff water moved down slope. Nagpal (1985) showed that a majority of surface applied P can be adsorbed in the surface layer of a soil profile.

Table 21. Runoff water quality during the growing season (April to September, 1990) at Montgomery Co. site.

Parameter		Effluent Loading Rate -----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	15	15	10
	SE‡	0.23	0.24	0.21
	Mean	7.33 a*	7.17 a	7.21 a
EC (S m <sup>-1</sup> )	N	9	8	8
	SE	0.0024	0.0024	0.0047
	Mean	0.0169 b	0.0151 b	0.0238 a
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	15	15	10
	SE	0.36	1.16	1.5
	Mean	1.41 a	2.12 a	3.23 a
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	15	15	10
	SE	0.11	0.56	0.32
	Mean	0.43 a	0.91 a	0.83 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	8	8	4
	SE	1.3	0	3.7
	Mean	1.9 a	0 a	3.7 a
PO <sub>4</sub> -P (mg L <sup>-1</sup> )	N	15	15	10
	SE	0.04	0.100	0.1
	Mean	0.19 a	0.22 a	0.24 a

\* Means followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.

Table 22. Runoff water quality during the non-growing season (October to March, 1990) at Montgomery Co. site.

Parameter		Effluent Loading Rate		
		-----cm wk <sup>-1</sup> -----		
		0	1.25	2.5
pH	N†	15	15	15
	SE‡	0.20	0.18	0.18
	Mean	6.54 *	6.56 a	6.68 a
EC (S m <sup>-1</sup> )	N	15	15	15
	SE	0.0011	0.0012	0.0032
	Mean	0.0117 b	0.0141 b	0.0238 a
NO <sub>3</sub> <sup>-</sup> -N (mg L <sup>-1</sup> )	N	15	15	15
	SE	1.68	1.51	0.1
	Mean	2.23 a	2.41 a	0.41 a
NH <sub>4</sub> <sup>+</sup> -N (mg L <sup>-1</sup> )	N	15	15	15
	SE	0.08	0.04	0.05
	Mean	0.27 a	0.24 a	0.26 a
Cl <sup>-</sup> (mg L <sup>-1</sup> )	N	15	15	15
	SE	1.79	2.1	3.04
	Mean	9.7 b	11.5 b	19.5 a
PO <sub>4</sub> -P (mg L <sup>-1</sup> )	N	15	15	15
	SE	0.02	0.03	0.04
	Mean	0.08 b	0.11 b	0.16 a

\* Means followed by the same letter for individual parameters are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N indicates number of samples.

‡ SE indicates standard error.



Table 23. Runoff water quality parameters tested for seasonal differences.

Effluent Rate	Season	pH	EC	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	PO <sub>4</sub> -P	Cl <sup>-</sup>
cm/wk			S m <sup>-1</sup>	-----mg L <sup>-1</sup> -----			
0	N-G†	6.54b*	0.0117b	2.23a	0.27a	0.08b	9.6a
	G‡	7.32a	0.0169a	1.41a	0.43a	0.19a	1.9b
1.25	N-G	6.56b	0.0141a	2.41a	0.24a	0.11a	11.5a
	G	7.17a	0.0151a	2.12a	0.91a	0.23a	0b
2.5	N-G	6.68b	0.0238a	0.41b	0.26b	0.16a	19.5a
	G	7.21a	0.0238a	3.23a	0.83a	0.25a	3.6b

\* Means followed by the same letter within each loading rate are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† N-G indicates non-growing season.

‡ G indicates growing season

Table 24. Dilution and N removal rates in runoff waters based on measured and calculated values for the growing and non-growing seasons.

Loading Rate	Effluent content†	NO <sub>3</sub> <sup>-</sup> -N Concentration		N Loss
		Measured	Calculated‡	M/C¶
cm wk <sup>-1</sup>	--%---	-----mg L <sup>-1</sup> -----		--%--
Non-growing season				
1.25	3.6	2.41	2.86	15.7
2.5	19.8	0.41	4.48	90.8
Growing season				
1.25	0	2.12	1.84	0
2.5	3.0	3.23	2.15	0

† Percentage of runoff water originating from effluent based on Cl<sup>-</sup> data

‡ Based on dilutions determined from Cl<sup>-</sup> data.

¶ Measured/Calculated.

Table 25. Dilution and P removal rates in runoff waters based on measured and calculated values for the growing and non-growing seasons.

Loading Rate	Effluent Content†	PO <sub>4</sub> -P Concentration		P Loss
		Measured	Calculated‡	M/C¶
cm wk <sup>-1</sup>	--%---	-----mg L <sup>-1</sup> -----		--%--
Non-growing season				
1.25	3.6	0.11	0.19	42
2.5	19.8	0.16	0.68	76
Growing season				
1.25	0	0.22	0.19	0
2.5	3.0	0.24	0.28	14

† Percentage of runoff water originating from effluent based on Cl<sup>-</sup> data.

‡ Based on dilutions determined from Cl<sup>-</sup> data.

¶ Measured/Calculated.

## PLANT YIELD AND N UPTAKE

### Montgomery Co.

The tall fescue cover was harvested in July and August 1989 after the start of effluent application in June 1989 and once a month during the growing season (May-August) in 1990 and again in October 1990. Plant growth was minimal after the October 1990 harvest and no measurements were taken during the winter months. There were no differences between control and effluent application treatments in plant tissue production during the first summer of effluent application (1989). However in the second year of application (1990), effluent irrigation increased the yield of fescue at both application rates (Figure 16). Fescue yield from the 1.25 cm wk<sup>-1</sup> application rate followed the same trend as the yield from the control plots which decreased between May and October 1990 (Figure 16). However yield at the 2.5 cm wk<sup>-1</sup> application rate maintained high levels for the same period. The average yield over the entire study was higher at the 2.5 cm wk<sup>-1</sup> application rate than at the 1.25 cm wk<sup>-1</sup> and control rates (Table 26). There were also no differences between the control and 1.25 cm wk<sup>-1</sup> application rates. Effluent irrigation increased overall yields 19 and 48% in the 1.25 cm wk<sup>-1</sup> and 2.5 cm wk<sup>-1</sup> application rates, respectively.

There were also no differences between treatments for N uptake during 1989. However, as with the yield data, N uptake increased with increased effluent application during the summer of the second year (Figure 17). Increases in N

uptake were in part derived from the increased yields. The high N uptake in samples collected in October 1990 resulted from the high tissue yields for that period. Overall the removal of N by the fescue cover was increased by 34 and 69% in the 1.25 and 2.5 cm wk<sup>-1</sup> application rates, respectively.

The average amount of effluent N applied in one month is graphically represented with the N uptake data in Figure 17. Plant uptake of N during the summer months in 1990 accounted for a large majority of the N applied. In July and August, over 100% of the applied N was present in plant tissue. The differences between N uptake for the first summer (1989) and the second summer (1990) indicate that N uptake in 1990 may have included N that accumulated during the previous winter. Samples collected in July and August 1989 represent only 2 and 3 months of effluent application while samples collected in 1990 received effluent for over one year. Nitrogen accumulations from effluent application during winter months probably contributed to the high N uptake by the grass cover in the summer months of 1990. Nitrogen accumulations would also explain why N recovery rates in the plant tissue were over 100% in certain months. Similar high removal rates of N from applied wastewater by forage grasses are reported in the literature. Palazzo (1981) observed high N uptake in municipal wastewater irrigation system, where orchardgrass accumulated 85% of the applied N between May and September.

The total amount of N uptake from August 1989 and August 1990 in the Montgomery Co. site was 108 and 127 kg ha<sup>-1</sup> yr<sup>-1</sup> for the 1.25 and 2.5 cm wk<sup>-1</sup>

application rates, respectively. These values are low compared to N accumulations under wastewater irrigation reported by Palazzo (1981) for orchardgrass (256 - 323 kg ha<sup>-1</sup> yr<sup>-1</sup>) and Boardman et al. (1984) who observed 294 - 412 kg ha<sup>-1</sup> yr<sup>-1</sup> N accumulation in fescue. The fescue cover was not harvested in the autumn of 1989 at the Montgomery Co. site, which may have reduced the amount of annual N uptake measured for the site. The fescue cover was harvested again in the following spring, however, plant material that had accumulated in the autumn may have been lost during the winter. It is well documented that yields of forage grasses and removal of nutrients are dependent on cutting management (Heath, 1980). Research where higher N uptake was reported were more intensively managed with more cuttings and received a higher amount of applied N (Palazzo, 1981; Hook and Burton, 1979).

### **Alleghany Co.**

The yield and N uptake of the fescue cover in the Alleghany Co site were measured in the summer of 1990 (Figures 18 and 19). There were no differences between the control and irrigated treatments in the yield of fescue in the early part of the growing season (May and June). As the growing season progressed, yield differences became apparent. There was a stepped increase in yield with effluent irrigation in July and August 1990. Even though rainfall in July and August 1990 was above normal in Alleghany Co. (Appendix Table A-4), the additional water from effluent increased plant growth. Additional water from irrigation could have

had an impact in the months with above normal rainfall since evapotranspiration is high in July and August. Effluent irrigation at the Alleghany Co. site, increased yield by 16.5 and 46% at the 1.25 and 2.5 cm wk<sup>-1</sup> application rates, respectively, when compared with the control (Table 26). The increases in yield (percent) were very similar to yield increases measured at the Montgomery Co. site.

As in the Montgomery Co. site, N uptake in the Alleghany Co. site increased as yield increased (Figure 19). The highest N uptake was measured in August, and corresponded with large increases in yield between effluent irrigation treatments and the control treatment. The amount of N applied in the effluent on a monthly basis is shown in Figure 19. The amount of N uptake in August was much higher than the total amount of N applied in the effluent for that month. This agrees with the N uptake data from the Montgomery Co. site which indicated that the fescue cover utilized N that had accumulated in the soil as a result of spray irrigation. The removal of N by the fescue cover was increased by 73 and 139% at the 1.25 and 2.5 cm wk<sup>-1</sup> rates, respectively.

The total N accumulated between May and September 1990 was 143 and 198 kg ha<sup>-1</sup> yr<sup>-1</sup> which, when corrected by subtracting the controls values, represent 117 and 109% of the N applied in the effluent for the same period. This agrees with the Montgomery Co. where over 100% of the applied N was recovered in the plant uptake during the summer. These data indicate that much of the N applied during periods of plant growth was not lost from the soil system.

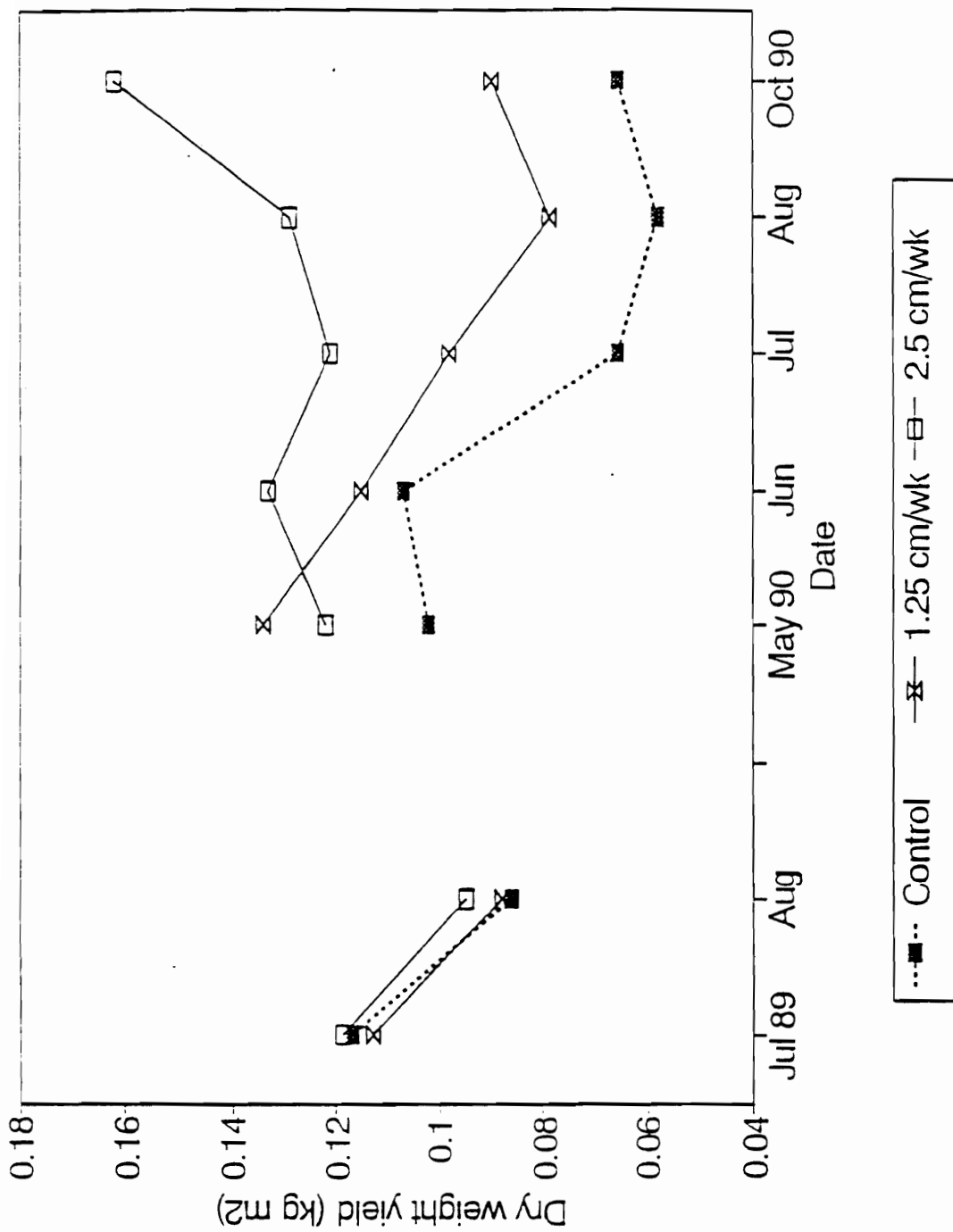


Figure 16. Dry weight yield of fescue cover for each sampling date at the Montgomery Co. site.



Table 26. Nitrogen uptake and dry weight yield of fescue cover for one year (1990) at both field sites.

Site	Effluent Rate	N Uptake		Dry Weight Yield	
		N†	Mean	N	Mean
	cm wk <sup>-1</sup>		---mg N m <sup>-2</sup> ---		---kg m <sup>-2</sup> ----
Montgomery Co.	0	20	7715 b*	21	0.40 b
	1.25	21	11214 b	21	0.52 b
	2.5	19	15201 a	19	0.67 a
Allegheny Co.	0	20	7680 c	20	0.36 b
	1.25	12	14337 b	12	0.57 b
	2.5	12	19832 a	12	0.72 a

\* Means followed by the same letter for individual sites are not significantly different ( $P \leq 0.05$ )

as determined by Duncan's Multiple Range Test.

† N indicates number of measurements.

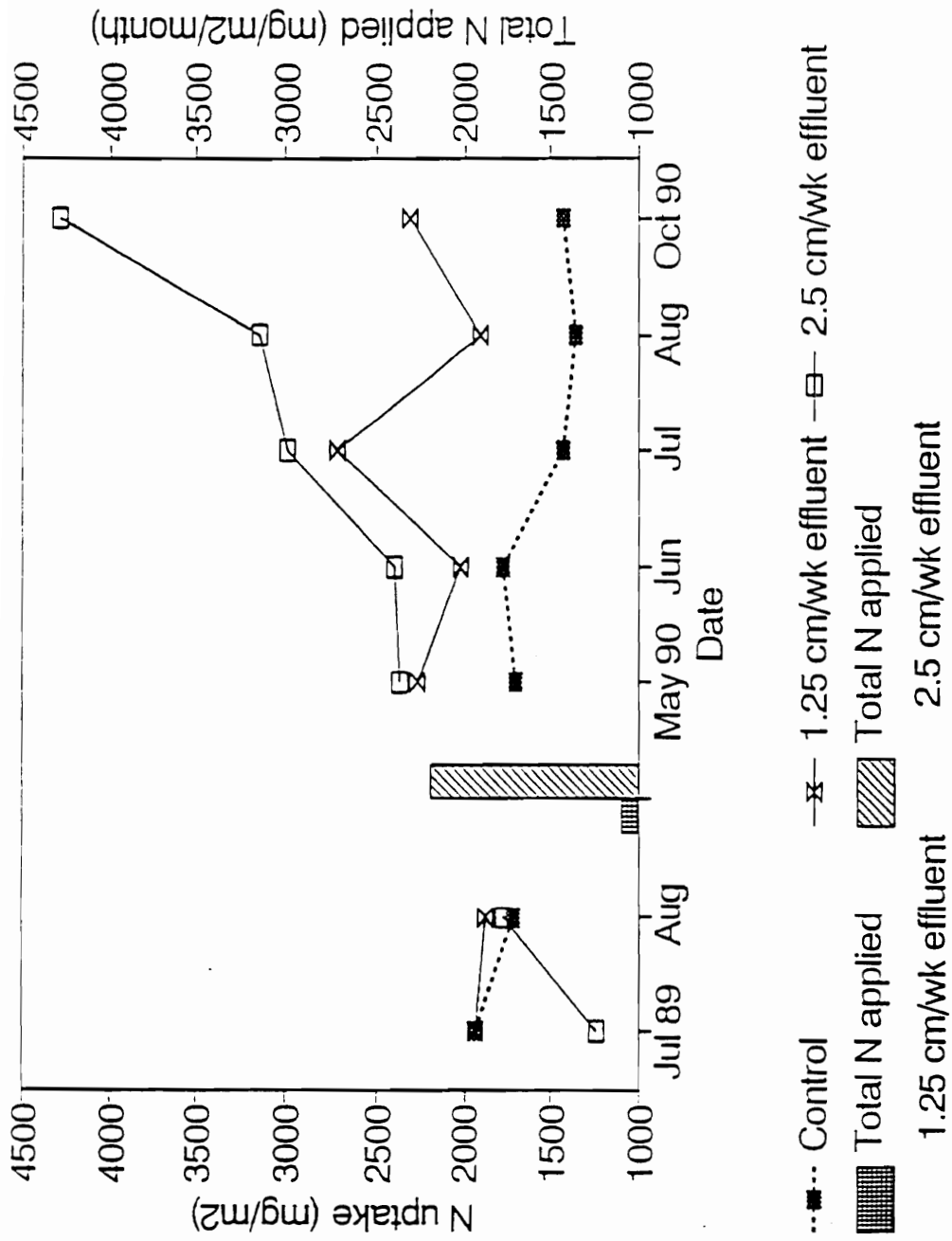


Figure 17. Nitrogen uptake by fescue cover for each sampling date at the Montgomery Co. site.

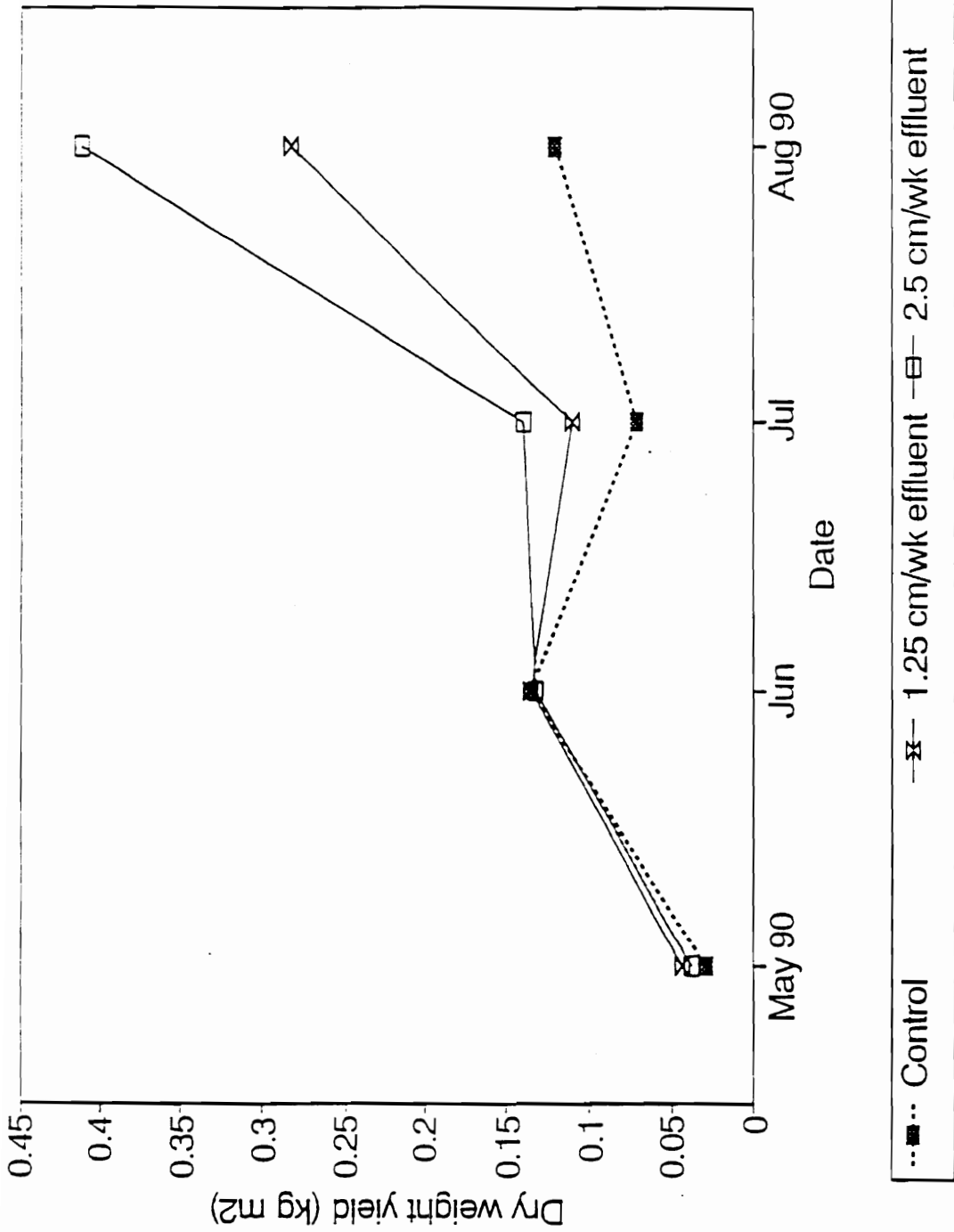
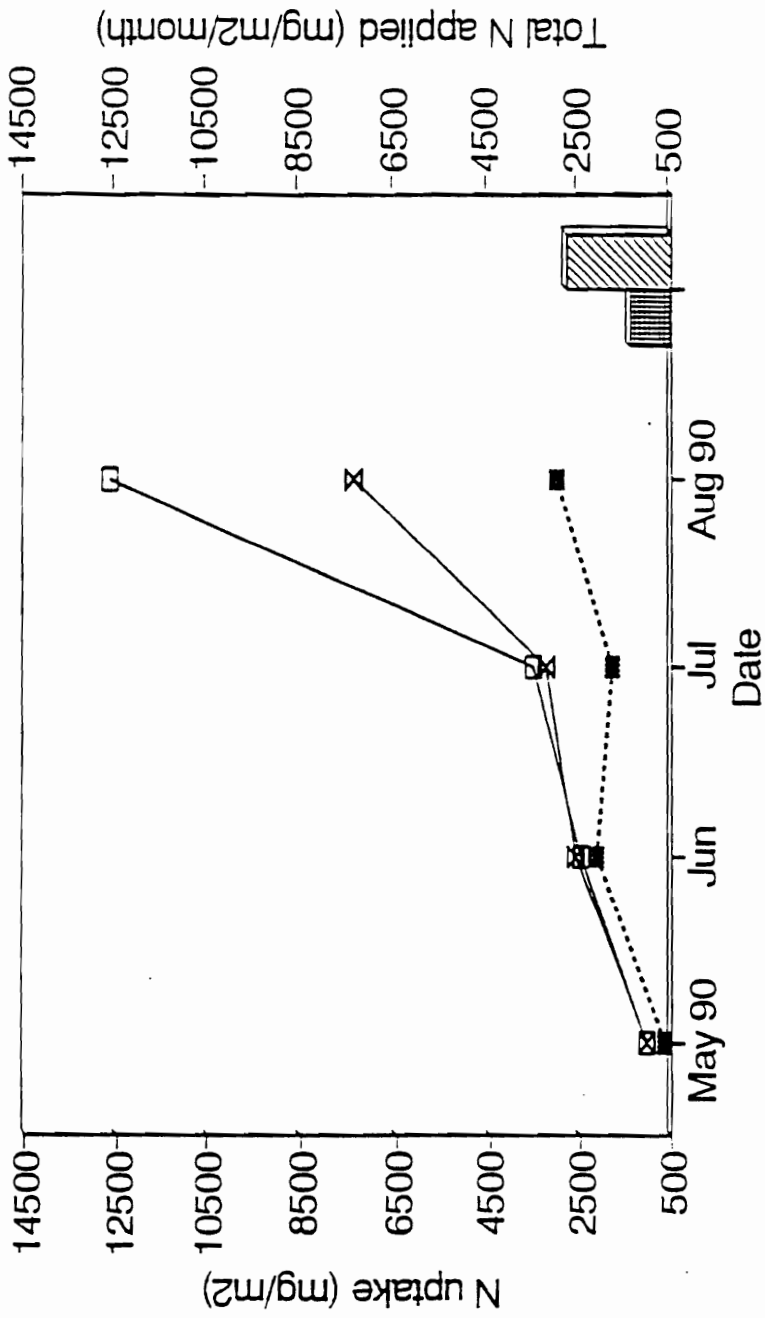


Figure 18. Dry weight yield of fescue cover for each sampling date at the Alleghany Co. site.



---■--- Control  
 ---x--- 1.25 cm/wk effluent    -□- 2.5 cm/wk effluent  
 ■■■■ Total N applied  
 1.25 cm/wk effluent    2.5 cm/wk effluent

Figure 19. Nitrogen uptake by fescue cover for each sampling date at the Alleghany Co. site.

## DENITRIFICATION

Periodic measurements of denitrification over a 10 month span using the acetylene blocking technique at the Montgomery Co. site showed a relatively small amount of gaseous N loss (Figure 20). The N<sub>2</sub>O emission rates were highly variable with coefficients of variation (CV) of 203, 196 and 310% in the 0, 1.25 and 2.5 cm/wk application rates, respectively. Large CV's are not uncommon with denitrification measurements and coefficients in excess of 200% have been reported by Folorunso and Rolston (1985a). At the Montgomery Co. site, the largest source of variation was between sample dates, which is not unexpected because environmental conditions such as soil moisture content and temperature have strong influences on denitrification rates. Burton and Beauchamp (1985) attempted to correlate N<sub>2</sub>O production with numerous soil and environmental parameters and observed that the amount of air-filled porosity had the strongest correlation.

Effluent irrigation did not greatly affect N<sub>2</sub>O emissions collected from the field. Distance away from the sprinkler within the spray circles had no effect on the amount of N<sub>2</sub>O collected. There were no differences between the control plots and irrigated plots when all dates are considered. However, there was a trend toward increased denitrification with irrigation. The highest denitrification rates were observed in May and June for all treatments. These results were not unexpected because the high rates of denitrification coincided with the highest soil

temperatures (Table 27). Thus the warmer temperatures enhanced microbiological activity. Since the levels of  $N_2O$  emission were low and there were no differences between the control and effluent irrigated plots it is difficult to determine how much if any of the  $N_2O$  gas collected originated from effluent. In May 1990, when gas collection from effluent irrigated plots were higher than from control plots, the amount of  $N_2O$  collected represented only 5.4 and 5.0% of the applied effluent at the low and high rates of application, respectively. Losses of N from denitrification in fertilized agricultural plots under irrigation have been reported between 5 and 15% (Rolston et al., 1982; Sexton et al., 1985). However it has been noted that denitrification losses from land treatment are thought to be higher than normal agricultural fields due to the high hydraulic loading and more microanaerobic sites (Brar et al., 1978). In studies where secondary sewage effluent was intermittently flooded instead of spray irrigated, denitrification losses were between 24 and 80% (Reddy and Patrick, 1975; Gilbert et al., 1979). At the Montgomery Co. site, effluent was applied at 1 and 3 a.m. and the samples were placed in the incubation tubes at 8 a.m. after which  $N_2O$  emissions were collected for 24 hours. It has been documented that denitrification can occur in "burst" after an irrigation event or a rainfall event (Rolston et al., 1980; Folorunso and Rolston, 1985b). It is possible that the field sampling for denitrification may have missed 'bursts' of denitrification activity thus resulting in the underestimation of  $N_2O$  emission. It is also possible that anaerobic conditions were not achieved in the field during the summer months.

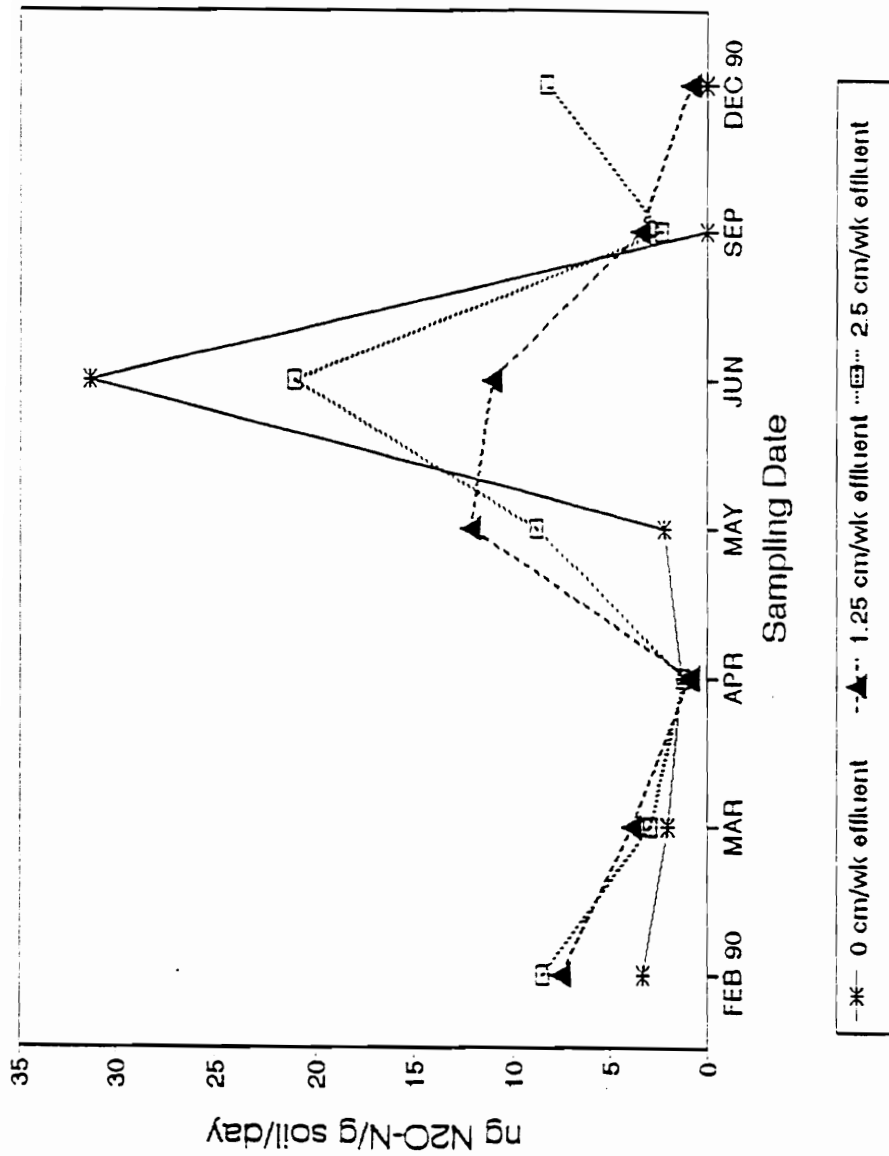


Figure 20. Denitrification rates for each treatment from Feb - Dec 1990 at the Montgomery Co. site.

Table 27. Soil temperature and denitrification rates for each treatment from Feb - Dec 1990 at the Montgomery Co. site.

Month	Temp†	Effluent Loading		
		-----cm wk <sup>-1</sup> -----		
	°C	0	1.25	2.5
		-----ng N <sub>2</sub> O-N g <sup>-1</sup> soil day <sup>-1</sup> -----		
Feb	5.5	3.3 b*	7.5 ab	8.5 a
Mar	7.5	2.0 b	3.9 ab	2.9 a
Apr	10.5	1.3 b	0.9 b	1.2 a
May	21.0	2.2 a	12.2 a	8.8 a
Jun	21.1	31.4 b	11.1 a	21.1 a
Sep	16.1	0.0 b	3.4 ab	2.3 a
Dec	2.7	0.0 b	0.8 b	8.3 a

\* Means followed by the same letter within the same rate are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† Soil temperature median at 10 cm depth during 24 hr sampling period.



## INFLUENCE OF WASTEWATER ON SOIL CHEMICAL PROPERTIES

### Nitrogen

The concentrations of extractable  $\text{NH}_4^+$ -N and  $\text{NO}_3^-$ -N in the top 15 cm of the soil profile at the Montgomery Co. site from February to December 1990 are presented in Figures 21 and 22, respectively. The  $\text{NH}_4^+$ -N concentrations in the soil fluctuated with the time of the year during the 10 month span. For 8 months (May to Dec), the  $\text{NH}_4^+$ -N concentration was the highest where 2.5 cm  $\text{wk}^{-1}$  of wastewater was applied. Concentrations of  $\text{NH}_4^+$ -N decreased in April, 1990 and remained at relatively low concentrations until September 1990 indicating  $\text{NH}_4^+$ -N loss from the system was probably caused by increased nitrification during the warmer temperatures in the summer months. After September 1990,  $\text{NH}_4^+$ -N accumulated in the soil profile. This accumulation was attributed to reduced nitrification at colder temperatures. The  $\text{NO}_3^-$ -N concentrations did not fluctuate as much as  $\text{NH}_4^+$ -N concentrations and were generally low (Figure 22). There were few differences in  $\text{NO}_3^-$ -N concentrations between the 0, 1.25, and 2.5 cm  $\text{wk}^{-1}$  effluent loading rates for all the months measured except for March 1990. Samples collected in March had between 20 and 25  $\mu\text{g NO}_3^- \text{N g}^{-1}$  of soil where 1.25 cm  $\text{wk}^{-1}$  of wastewater was applied. Warmer temperatures in March may have caused an increase in nitrification of accumulated  $\text{NH}_4^+$ -N.

The low amount of  $\text{NO}_3^-$ -N in the topsoil is attributed to losses from plant uptake and denitrification. Plant uptake of N is expected to be greatest in the

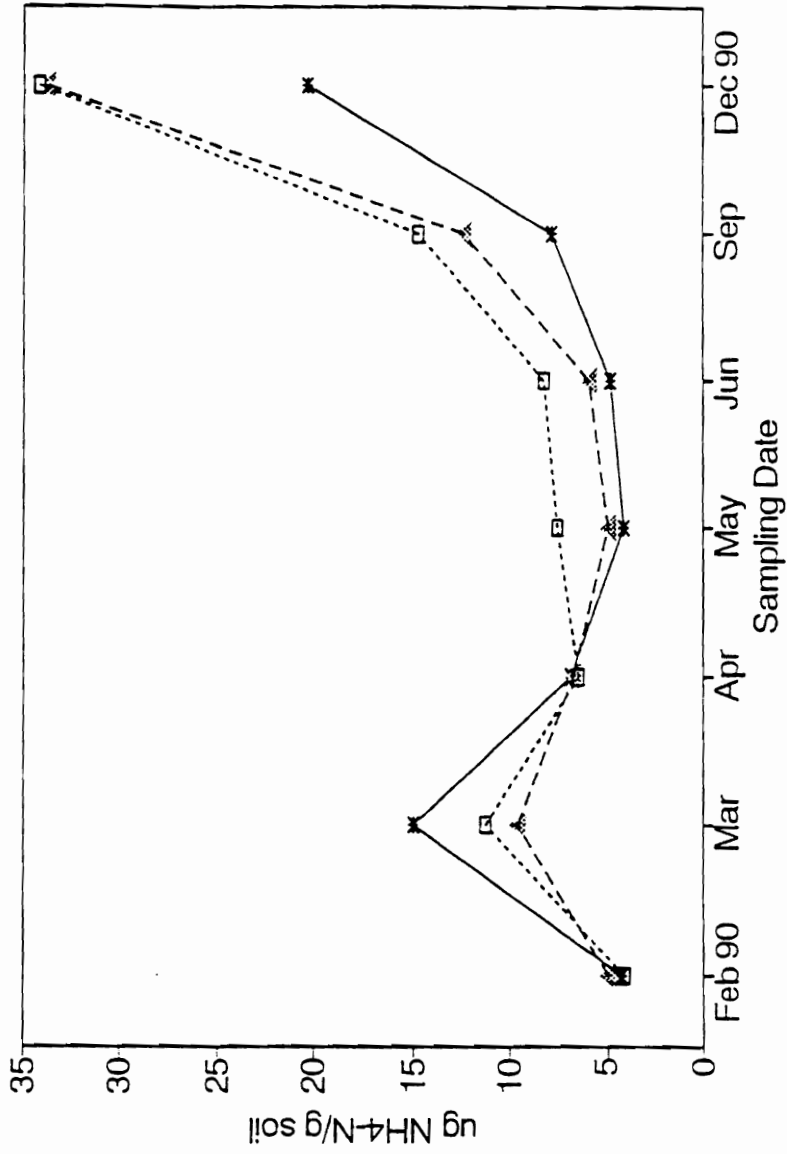
surface horizon because of higher root density. Greater  $\text{NO}_3^-$ -N loss by denitrification would be expected in the surface horizon because of high levels of available C and the potential for creation of anaerobic microsites during an irrigation or rainfall event.

The extractable  $\text{NH}_4^+$ -N,  $\text{NO}_3^-$ -N and TKN concentrations with soil depth from samples collected in November 1990 for the Montgomery Co. and Alleghany Co. sites are shown in Tables 28 and 29, respectively. At the Montgomery Co. site  $\text{NO}_3^-$ -N concentrations in the soil samples were  $<2.5 \mu\text{g g}^{-1}$  which is in agreement with the subsurface water samples which contain  $\text{NO}_3^-$ -N concentrations  $< 1 \text{ mg L}^{-1}$ . Even though there were no differences between treatments, there was a trend toward higher  $\text{NO}_3^-$ -N levels at the  $2.5 \text{ cm wk}^{-1}$  irrigation rate and at deeper depths in the profile. Nitrate concentrations in the 0-15 and 15-30 cm depths were higher for the  $2.5 \text{ cm wk}^{-1}$  application rate than for the  $1.25$  and  $0 \text{ cm wk}^{-1}$  rates at the Alleghany Co. site. However,  $\text{NO}_3^-$ -N concentrations remained low, averaging  $<5 \mu\text{g g}^{-1}$  soil. The observation of increased soil  $\text{NO}_3^-$ -N in the Alleghany Co. site and not the Montgomery Co. site was probably due to the higher amount of  $\text{NO}_3^-$ -N in the applied effluent at the Alleghany Co. site.

As expected the highest  $\text{NH}_4^+$ -N concentrations at both sites were in the 0-15 cm depth (Tables 28 and 29). At the Montgomery Co. site,  $\text{NH}_4^+$ -N concentrations decreased with depth, but there was no difference in extractable  $\text{NH}_4^+$ -N concentrations between irrigation rates at any of the depths sampled. However, as observed in the monthly topsoil measurements,  $\text{NH}_4^+$ -N

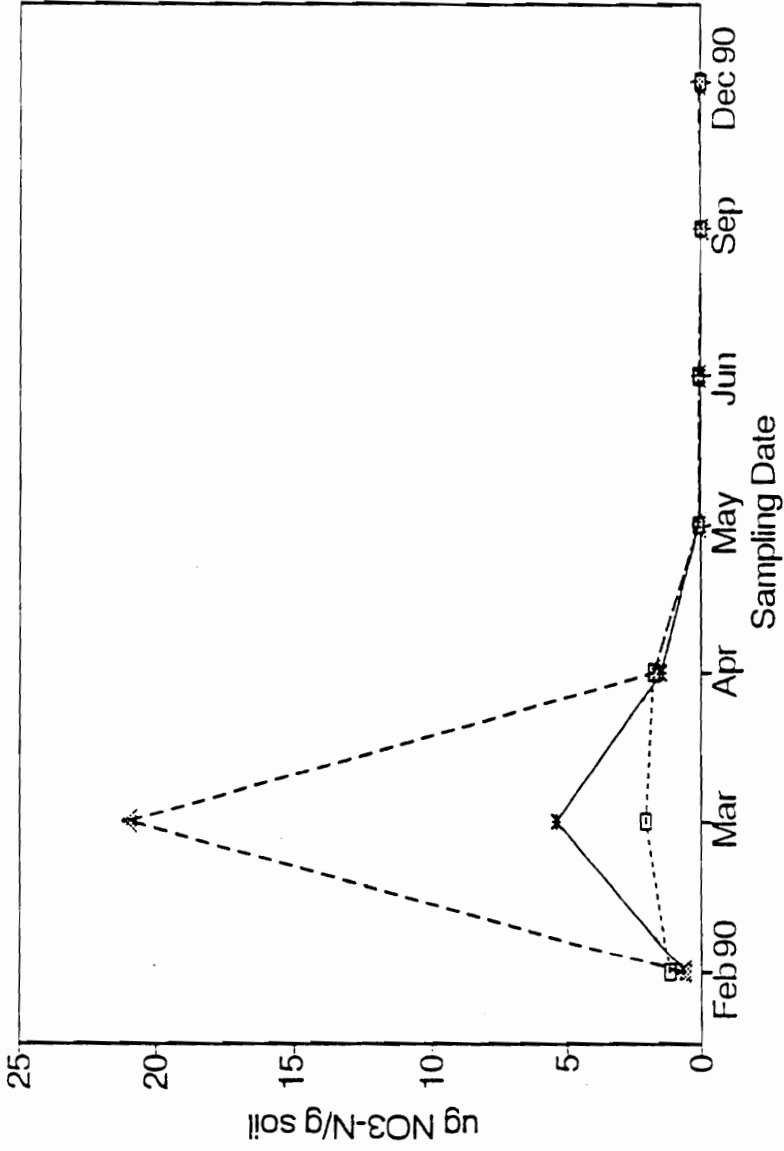
concentrations fluctuated with the season (Figure 21). In the Alleghany Co. site, there were differences in soil  $\text{NH}_4^+$ -N concentrations in the upper 30 cm of the soil (Table 29). There was a trend of higher  $\text{NH}_4^+$ -N concentrations under the irrigated rates in the 0-15 cm depth and an increase in  $\text{NH}_4^+$ -N at the 15-30 cm depth. With further increases in depth, 30-45 and 45-60 cm there were no differences in extractable  $\text{NH}_4^+$ -N concentrations with effluent application. Ammonium appeared to travel further in the profile at the Alleghany Co. site compared to the Montgomery Co. site. The deeper percolation of  $\text{NH}_4^+$ -N at the Alleghany Co. site was probably due to the slightly coarser soil texture, lower CEC and higher permeability as compared to the Montgomery Co. site.

The TKN values did not change in any of the soil layers at both sites as a result of effluent application with the exception of the 45-60 cm depth at the Alleghany Co. site where there was a decrease under the high loading rate. Since this same result was not observed at the Montgomery Co. site, the decrease in TKN at this depth may reflect sample variability. The TKN present in the soil profile was higher in the surface horizon at both sites (Tables 28 and 29). This result was not unexpected since higher levels of organic matter are normally present in the surface horizons. Changes in the soil TKN content were also not expected where effluent was applied due to the small amount of TKN applied in the effluent relative to what is already present in the soil.



—\*— 0 cm/wk effluent    -▲- 1.25 cm/wk effluent    ···□··· 2.5 cm/wk effluent

Figure 21. Ammonium concentration means in the top 15 cm of soil from periodic sampling from Feb - Dec 1990 at the Montgomery Co. site.



\*— 0 cm/wk effluent    -▲- 1.25 cm/wk effluent    -□- 2.5 cm/wk effluent

Figure 22. Nitrate concentrations means in the top 15 cm of soil from periodic sampling from Feb - Dec 1990 at the Montgomery Co. site.

Table 28. Soil N and Cl<sup>-</sup> analysis from the Montgomery Co. site after 20 months of effluent application.

Rate -cm wk <sup>-1</sup> -	TKN♦			NH <sub>4</sub> <sup>+</sup> -N			NO <sub>3</sub> <sup>-</sup> -N		
	N†	SE‡	Mean	N	SE	Mean	N	SE	Mean
	-----µg g <sup>-1</sup> -----			-----µg g <sup>-1</sup> -----			-----µg g <sup>-1</sup> -----		
0-15 cm depth									
0	7	244	2214 a*	9	3.3	15.2 a	9	0.3	1.4 a
1.25	5	324	2368 a	8	2.2	16.5 a	8	0.5	1.6 a
2.5	6	150	1761 a	8	2.1	14.2 a	8	0.8	1.3 a
15-30 cm depth									
0	8	108	850 a	9	0.5	2.4 a	9	0.1	0.1 a
1.25	7	71	801 a	9	1.1	2.3 a	9	0.3	0.7 a
2.5	5	81	870 a	9	0.3	1.4 a	9	0.3	0.7 a
30-45 cm depth									
0	8	64	751 a	9	0.5	1.3 a	9	0.3	0.9 a
1.25	9	68	847 a	9	0.3	0.3 a	9	0.7	1.6 a
2.5	8	56	868 a	9	2.0	4.1 a	9	1.9	2.7 a
45-60 cm depth									
0	9	65	916 a	9	0.3	0.5 a	9	0.3	0.6 a
1.25	6	86	888 a	9	0.0	0.0 a	9	0.5	0.9 a
2.5	7	85	895 a	9	0.8	1.5 a	9	0.6	1.1 a

\* Means followed by the same letter in the same column for each depth are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

♦ Total kjeldahl nitrogen.

† N indicates sample size.

‡ SE indicates standard error.

Table 29. Soil N analysis from the Alleghany Co. site after 12 of effluent application.

Rate -cm wk <sup>-1</sup> -	TKN♦			NH <sub>4</sub> <sup>+</sup> -N			NO <sub>3</sub> <sup>-</sup> -N		
	N†	SE‡	Mean	N	SE	Mean	N	SE	Mean
		-----µg g <sup>-1</sup> -----			-----µg g <sup>-1</sup> -----			-----µg g <sup>-1</sup> -----	
0-15 cm depth									
0	9	109	1163 a*	9	5.0	18.6 a	9	0.7	2.5 ab
1.25	9	122	1418 a	8	4.1	23.2 a	8	0.3	0.7 b
2.5	9	55	1153 a	9	7.3	20.5 a	9	1.0	4.8 a
15-30 cm depth									
0	9	61	613 a	9	0.9	4.4 b	9	0.2	0.4 b
1.25	9	84	721 a	9	1.0	7.0 ab	9	0.2	0.5 b
2.5	9	72	682 a	9	1.1	8.1 a	9	0.4	1.3 a
30-45 cm depth									
0	9	69	408 a	9	0.9	2.6 a	9	0	0 a
1.25	9	25	425 a	9	0.4	2.9 a	9	0.1	0.1 a
2.5	9	51	327 a	9	0.8	3.9 a	9	0.2	0.2 a
45-60 cm depth									
0	9	95	564 a	9	0.4	1.6 a	9	0	0 a
1.25	9	44	522 a	9	0.3	2.5 a	9	0	0 a
2.5	9	44	279 b	9	0.3	2.4 a	9	0.2	0.2 a

\* Means followed by the same letter in the same column for each depth are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

♦ Total kjeldahl nitrogen.

† N indicates sample size.

‡ SE indicates standard error.

## **Phosphorus**

After at least 1 year of effluent application, changes were slight in the measured extractable soil  $\text{PO}_4\text{-P}$  concentrations at both sites (Tables 30 and 31). At the Montgomery Co. site, there were no differences in the extractable  $\text{PO}_4\text{-P}$  concentrations between treatments at any depth sampled. However, there was a trend of increased  $\text{PO}_4\text{-P}$  in the top 15 cm with increased effluent application. This higher concentration in the upper part of the soil profile was expected since  $\text{PO}_4\text{-P}$  is readily absorbed and precipitated in the soil profile. The trend of increased  $\text{PO}_4\text{-P}$  concentrations in the soil surface was not reflected in  $\text{PO}_4\text{-P}$  concentrations in the surface runoff waters where no differences were observed. At the Alleghany Co. site there was no differences in  $\text{PO}_4\text{-P}$  concentrations between effluent application except at the 0-15 cm depth, where there was a decrease at the 2.5  $\text{cm wk}^{-1}$  application rate. It was expected that increase effluent application would raise soil  $\text{PO}_4\text{-P}$  concentrations, as was the case in the Montgomery Co. site. The reduced level was most probably a result of a sampling or analysis error.

## **Cations**

Few changes in the exchangeable soil cations as a result of effluent application were observed at either site (Tables 30 and 31). At the Alleghany Co. site, there was an increase of Mn concentrations at the 15-30 cm depth in plots that received effluent irrigation. This may have resulted from changes in the soil redox potential where effluent was applied. Manganese concentrations have been



observed to increase under anaerobic conditions or with decreased redox potential (Ponnamperuma, 1972). Effluent irrigation did not change the concentrations of Ca, Mg, K and Zn at any depth at either site. The results from Tables 30 and 31 illustrate that effluent irrigation did not change the distribution of exchangeable cations in the soil profile.

Table 30. Soil P and cation content with depth after effluent applications to the Montgomery Co. site‡.

Effluent Rate	Parameters						
	P	K	Ca	Mg	Zn	Mn	
0-15 cm depth							
							µg g <sup>-1</sup>
0	2.1 a*	163 a	1178 a	130 a	4.5 a	134 a	
1.25	3.3 a	121 a	1172 a	162 a	5.0 a	98 a	
2.5	3.7 a	111 a	1043 a	110 a	5.5 a	148 a	
15-30 cm depth							
0	<0.01 a	39 a	698 a	77 a	2.5 a	47 a	
1.25	<0.01 a	55 a	644 a	111 a	3.5 a	53 a	
2.5	<0.01 a	52 a	806 a	111 a	3.8 a	43 a	
30-45 cm depth							
0	<0.01 a	52 a	918 a	139 a	4.6 a	39 a	
1.25	<0.01 a	57 a	733 a	156 a	4.1 a	18 a	
2.5	<0.01 a	56 a	1017 a	178 a	3.6 a	38 a	

\* Means followed by the same letter in individual parameters for each depth are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

‡ Collection of soil samples after 20 months of effluent application.

Table 31. Soil P and cation content with depth after effluent applications to the Alleghany Co. site†.

Effluent Rate cm wk <sup>-1</sup>	Parameters						
	P	K	Ca	Mg	Zn	Mn	
0-15 cm depth							
	-----µg g <sup>-1</sup> -----						
0	22.5 ab*	45.6 a	298 a	19.9 a	7.6 a	217 a	
1.25	33.9 a	58.8 a	600 a	97.9 a	8.6 a	289 a	
2.5	16.6 b	52.3 a	549 a	24.3 a	8.4 a	293 a	
15-30 cm depth							
0	8.9 a	44.6 a	262 a	11.0 a	8.8 a	81 b	
1.25	10.5 a	42.7 a	380 a	9.4 a	7.8 a	125 ab	
2.5	5.8 a	37.6 a	367 a	7.7 a	14.3 a	150 a	
30-45 cm depth							
0	1.0 a	51.8 a	245 a	21.8 a	7.8 a	5 a	
1.25	0.6 a	50.8 a	308 a	16.8 a	12.9 a	21 a	
2.5	0.8 a	43.3 a	286 a	16.2 a	13.6 a	51 a	

\* Means followed by the same letter in individual parameters for each depth are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† Collection of soil samples after 12 months of effluent application.

## DISCUSSION SUMMARY

The results from both field studies indicate that spray irrigation as an alternative OSWTDS properly renovated the applied domestic wastewater. Even though both sites were limited by shallow depths to a restrictive layer, the utilization of the topsoil as a treatment medium in spray irrigation of effluent proved to be effective. Effluent application had little apparent effect on soil moisture conditions. The soil  $\Psi_m$  was influenced more by climate changes than effluent application. No surface ponding of effluent was observed at either site at any time. Nitrogen analysis indicated that N in effluent applied at appropriate rates via spray irrigation was not a threat to the quality of surface and subsurface waters. Nitrogen removal rates from subsurface waters were between 85 and 95%, with  $\text{NO}_3^-$ -N concentrations in subsurface waters at both field sites less than  $0.8 \text{ mg L}^{-1}$ .

Chloride concentrations indicated that shallow groundwaters (60 cm depth) contained between 17 and 58% applied wastewater. Dilution of effluent in the shallow groundwaters appeared to be highly influenced by recharge from rainwater. During periods of high evapotranspiration, a majority of the applied effluent probably did not travel beyond the root zone. Phosphorus concentrations in the shallow groundwaters indicated that removal rates from the applied effluent were at least 96% at both sites which was attributed to soil adsorption and precipitation reactions.

Runoff water quality was only minimally affected by effluent application. Chloride concentrations indicated that <20% of the collected runoff waters

originated from effluent irrigation for both effluent application rates. Nitrogen and  $\text{PO}_4$  concentrations in runoff waters were higher during the growing season which was attributed to increased microflora activity in the topsoil during warmer temperatures. Overall N and P concentrations in runoff waters were shown not to be affected by effluent irrigation.

During the growing season, plant uptake was the largest sink for applied N in the wastewater. In summer months, plant uptake at both sites accounted for over 100% of the applied N, however, plant growth and N uptake during the winter months were nil. Plant N uptake increased with increased irrigation indicating the plants assimilative capacity for N did not reach a maximum at the effluent application rates employed in this study. This was supported by N uptake levels reported in the literature which showed much higher N accumulations in tall fescue. Soil  $\text{NH}_4$  concentrations were observed to fluctuate with changes in the season, with accumulations occurring during colder temperatures which was attributed to reduced nitrification. An accumulation of soil  $\text{NO}_3$  levels was not observed at any time. There was no indication of increased leaching of soil cations due to effluent application. Field measurements of denitrification most probably missed large  $\text{N}_2\text{O}$  emission rates directly after effluent applications which potentially caused an underestimation of N losses due to denitrification.

## CONCLUSIONS

1. The small aerobic treatment unit was not as reliable as the literature generally indicates. Most of the N in the effluent was in the  $\text{NH}_4^+$ -N form. Also the wastewater disinfection was not consistent as indicated by frequent elevated fecal coliform counts in the effluent from the unit. This lack of consistency indicates that improvements need to be made to the chlorination system. However, with improved disinfection, these systems are suitable for effluent pretreatment prior to spray irrigation.
2. The sand filter performed satisfactorily and resulted in effluent where 65% of the inorganic N was in the  $\text{NO}_3^-$ -N form. The tablet chlorinator also operated in a satisfactory manner with this pretreatment system.
3. Adverse weather conditions did not affect the performance of the spray systems. No freezing of the sprinklers was observed. Runoff water quality in saturated winter conditions was shown to be acceptable.
4. Effluent renovation in the top 60 cm of the soil profile was high. Over 85 and 90% of the N applied at the Montgomery and Alleghany Co. sites, respectively, were removed in the top 60 cm of soil. A major sink for N removal was plant uptake. Phosphorus removal was also high (>96%) at both sites. The reduction in P was attributed to soil adsorption and precipitation reactions.
5. The  $\text{NO}_3^-$ -N concentrations in shallow ground waters was very low, averaging less than  $0.8 \text{ mg L}^{-1}$  at both sites. There was no indication of elevated  $\text{NO}_3^-$ -N

concentrations in shallow groundwaters above background levels at both the Montgomery and Alleghany County sites. The soil moisture status at both sites was not strongly influenced by effluent irrigation.

6. Runoff water quality was acceptable and posed no serious threat to nearby surface waters. The low loading rate per area and the filtering action of the soil surface were attributable mechanisms for the acceptable runoff water quality.

7. Levels of N<sub>2</sub>O emissions from the Montgomery Co. site represented only 5.0% of the applied N from effluent. However, measurements of high N<sub>2</sub>O emission rates directly after effluent application may have been missed.

8. Phosphorous and NO<sub>3</sub><sup>-</sup>-N concentrations beneath the spray irrigation system were much lower than those concentrations measured beneath most OSWTDS. Spray irrigation appears to be a suitable OSWTDS alternative at sites with large tracts of land with marginal soil conditions.

## **PART II**

# **DENITRIFICATION COLUMN STUDIES**

### **INTRODUCTION**

Field measurements of  $N_2O$  flux generally contain high variation. Some researchers reported coefficients of variation between 160 and 500% (Folorunso and Rolson, 1985a). The variation of  $N_2O$  fluxes was due to the complexity of the denitrification process. Many variables such as temperature, water content, pH, C, N and S concentrations and their interactions affect denitrification rates. Rolston et al. (1976) reported an inability to accurately measure gradients of flux near the soil surface because of this variability. Folorunso and Rolston (1985b) noted that denitrification flux data had the tendency to behave in a sharp spike-like function with large peaks instead of smooth curves. They noted that the fluxes are indicative of the denitrification process where denitrification only occurs at high water contents. Researchers who have examined spatial variation of denitrification in the field have reported that the cyclic nature of the soil-water content or air filled porosity in the surface soil was the primary factor affecting variability (Folorunso and Rolston, 1985b; Burton and Beauchamp, 1985).

Column experiments where rainfall and irrigation amounts can be more carefully controlled can provide more reliable denitrification measurements. Soil



columns are used widely as microcosms which are laboratory simulations of a natural system. Advantages of a microcosm study include: possible replication, controlled environments and cost effectiveness. A major disadvantage is that containerization can lead to structural and functional changes that are different from the natural environment. A column study was used in this research to more effectively examine denitrification from wastewater irrigation.

The denitrification column study consisted of three parts. The objectives of the first part and main focus of the column study was to determine the effect of irrigation frequencies on denitrification rates and to more accurately estimate denitrification rates where spray irrigation is employed. The two daily effluent loading rates that were used in the field study were divided into 3 irrigation cycles or frequencies for the column study. The second part of the column study was designed to determine if either N or C limited denitrification with spray irrigation of domestic wastewater. Soil samples were collected from the greenhouse columns after the conclusion of the irrigation frequency study and tested for C and N limitations in a laboratory microcosm experiment. The objectives of the laboratory microcosm study were to determine if different wastewater irrigation frequency treatments had changed the denitrification capacity of the soils. The second was to determine if C or N was limiting denitrification in these soils. The objectives of the third part of the column study was to determine the origin of the denitrifying gases using  $^{15}\text{N}$  tracer and to compare  $^{15}\text{N}_2$  flux to  $\text{N}_2\text{O}$  flux measurements from the frequency study.

## LITERATURE REVIEW

### **EFFECT OF IRRIGATION FREQUENCY ON DENITRIFICATION**

There has been limited research on the long term effects of different irrigation frequencies of applied wastewater. A workshop in 1984 in the Fourth National Symposium on Individual and Small Community Sewage Systems listed a major research need as determining the effect of alternating dry/wet conditions on a soils ability to receive wastewater (Ward and Morrison, 1985). With respect to N loss in wastewater irrigation systems, the frequency and the amount of irrigation can play an important role. Different irrigation frequencies and amounts can possibly affect microbial respiration, soil moisture status, and soil C and N dynamics. It is widely accepted that denitrification rates are strongly influenced by the moisture content of the soil.

There are scattered reports in the literature on responses of denitrification to different wetting and drying frequencies. Reddy and Patrick (1975) examined different frequencies of aerobic and anaerobic conditions and N loss from soils using incubation flasks and  $^{15}\text{N}$ . When comparing the alternating cycles, ranging from 2 to 128 days, the greatest loss of N was in the shortest cycle of 2 days aerobic and 2 days anaerobic. Rolston et al. (1982) performed a large scale study on the response of denitrification to irrigation frequency using both  $\text{C}_2\text{H}_2$  inhibition and  $^{15}\text{N}$  methods on a well drained loam soil. Although wastewater was not the irrigation source in this study, the results are of significance to wastewater

irrigation. The largest fluxes of  $N_2$  and  $N_2O$  occurred within 3-6 h after irrigation. Denitrification rates tended to decrease quickly within one to two days after irrigation. In general, frequent irrigation with small amounts of water resulted in the greatest amount of denitrification. They concluded that the small frequent irrigation application tended to keep the  $NO_3^-$ -N near the soil surface where the infrequent large applications tended to move the  $NO_3^-$ -N deeper into the soil profile. The most frequent irrigation treatment also resulted in the highest plant uptake of N and a higher concentration of  $NO_3^-$ -N in the root zone. Lance and Whisler (1972) examined N loss in soils under different flooding frequencies of secondary treatment effluent and noted that alternate flooding and drying periods were required for N removal. Longer flooding cycles (9-23 days) consistently produced a net N removal of 30%. They also reported that  $NH_4^+$ -N retention in the soil was important for N removal since N was held until nitrification occurred during dry conditions. The nitrified N was then denitrified during the next flooding event. This research demonstrates that the form of the applied N influenced  $N_2O$  emissions under different flooding frequencies.

Sexstone et al. (1985) examined temporal response of denitrification to moisture conditions using  $C_2H_2$  inhibition in sandy and clay loam soil cores. The sandy loam required almost three times more water input than the clay loam soil to achieve a maximum denitrification rate. In the sandy loam soil, the results were similar to that of Rolston et al. (1982) in that the denitrification rate increased immediately after water addition and reached a maximum within 3-5 h and returned

to initial levels after 12 h. The clay loam required 8-12 h before the maximum denitrification rate occurred and 48 h before background levels returned. In another related study, Groffman and Tiedje (1988) performed denitrification measurements using  $C_2H_2$  blockage and reported a hysteresis effect in wetting up and wetting down conditions. The authors concluded that the response was associated with the C dynamics in the soil since the microbial population was stressed by drying and rewetting.

### **EFFECT OF CARBON ON DENITRIFICATION**

The influence of C on denitrification is well documented in the literature. Carbon is utilized by microbes in denitrification as an energy source and recently has been noted by Tiedje (1988) for creating anaerobic microsites by enhancing microbial respiration which reduces the  $O_2$  supply. Researchers have established a linear relationship between soil organic C and denitrification (Stanford et al., 1968). Stanford et al. (1975) and Burford and Bremner (1975) confirmed that denitrification rates correlated better with water soluble or mineralized C than with total C, which is generally expected since most of the total C in soils is resistant to fast decomposition. Gilbert et al. (1979) evaluated denitrifying bacteria populations and N removal in columns intermittently flooded with secondary sewage effluent and reported that C additions increased N removal and denitrifying populations. Increased denitrification rates from glucose additions are also

documented by Brar et al. (1978), Jacobson and Alexander (1980), and Ardakani et al. (1973).

Parkin (1987) noted that natural denitrification rates were highly variable in that some samples in a given data set have extremely high rates while the remaining samples in the same data set have low rates. These "outliers" were attributed to "hot-spots" of denitrification activity caused by particulate organic material in the soil. Christensen et al. (1990) also noted denitrification hot-spots and added that the activity of these spots were close to the maximum denitrification capacity of the soil.

There is some debate in the literature on the long term availability of soil C for denitrification. Sextone et al. (1985) suggested that continuous wet and dry cycles exhausted available C and limited denitrification. Meek et al. (1970) demonstrated that periodic drying increased soluble C in the soil solution which tended to promote denitrification. In another column study, Ardakani et al. (1975) noted that even with continuous C and  $\text{NO}_3^-$ -N additions the rate of denitrification dropped after several weeks. This drop was attributed to reductions in indigenous soil C and an increase in the ratio of nitrifiers to denitrifiers which caused a decrease in net rate of denitrification. Stewart et al. (1979) examined N loss from wastewaters applied to packed soil columns containing a Histic epipedon (27% OM). They concluded that the available C for denitrification in the columns lasted only 130 to 180 days. Magdoff et al. (1974) also suggested that the rate of denitrification should decline with time as the soil organic matter decomposes.

However there are C inputs in the soil. Rolston and Cervelli (1978) demonstrated that the presence of crops increased denitrification which is due in part to the added C from root exudates. Tiedje (1988) suggested that situations where C would be a limiting factor in denitrification would be rare since, under anaerobic conditions, obligate aerobes cannot compete for available C and O<sub>2</sub> and stressed cells excrete more C. These conditions would make more C available for denitrifiers.

### **DENITRIFICATION CAPACITY**

Much work has been conducted on determining environmental conditions that limit denitrification. Available C, NO<sub>3</sub><sup>-</sup>-N, pH and aeration are the most common factors evaluated in denitrification limitations studies. Most of the research with factors that limit denitrification capacity are conducted in incubation flasks such as the work of Myrold and Tiedje (1985a). The forementioned authors studied the effects of moisture, NO<sub>3</sub><sup>-</sup>-N concentrations and C additions on changes in denitrification rates and observed that the total microbial biomass was only affected by C additions, however the soil did contain high residual NO<sub>3</sub><sup>-</sup>-N. The same researchers in a different article (1985b) used a model to describe NO<sub>3</sub><sup>-</sup>-N diffusion through soil aggregates in order to describe, on a micro scale, denitrification with NO<sub>3</sub><sup>-</sup>-N limitations. The model was based on Fick's second law of diffusion and Michaelis-Menten kinetics. The model showed that NO<sub>3</sub><sup>-</sup>-N

diffusion was not a limiting factor when the concentrations of  $\text{NO}_3^-$ -N in the bulk solution was high. However they concluded that  $\text{NO}_3^-$ -N diffusion may limit denitrification in aggregates larger than 20 mm.

## DENITRIFICATION MEASUREMENTS

Measurements of denitrification can be conducted by direct and indirect methods. Hauck (1986) pointed out that indirect methods include: measurement of  $\text{NO}_3^-$ -N disappearance,  $\text{NO}_3^-/\text{Cl}$  ratios, and N balance determinations which can utilize  $^{15}\text{N}$ . The  $\text{NO}_3^-/\text{Cl}$  ratio uses the assumption that  $\text{Cl}^-$  moves with  $\text{NO}_3^-$ -N through the soil and the change in ratios symbolizes  $\text{NO}_3^-$ -N loss. Difficulty with the ratio method stem from spatial variability of residual  $\text{NO}_3^-$ -N and  $\text{Cl}^-$ . Problems with N balance method determinations are with certain parameters in the N cycle which are difficult to measure without using  $^{15}\text{N}$ , they include  $\text{N}_2$  fixation and N immobilization. Another difficulty with using  $^{15}\text{N}$  tracer is that the equilibrium constants between organic and inorganic pools of N must be known. Both acetylene ( $\text{C}_2\text{H}_2$ ) inhibition and measurements of  $^{15}\text{N}_2$  flux are common direct methods.

### Acetylene Inhibition

The acetylene inhibition method or  $\text{C}_2\text{H}_2$  blockage technique is based upon the capability of  $\text{C}_2\text{H}_2$  to impede microbial reduction of  $\text{N}_2\text{O}$  to  $\text{N}_2$  by blocking the

$\text{N}_2\text{O}$  reductase system. This blockage greatly simplifies measurements of denitrification since the end product yields  $\text{N}_2\text{O}$  which is normally found in low concentrations in the atmosphere. Several researchers have observed that  $\text{C}_2\text{H}_2$  at about 1.01 kPa will inhibit the reduction of  $\text{N}_2\text{O}$  to  $\text{N}_2$  (Bladerston et al., 1976; Yoshinari and Knowles, 1976). It is desirable to apply  $\text{C}_2\text{H}_2$  for short periods since microorganisms that are inhibited can recover quickly from  $\text{C}_2\text{H}_2$ . Tiedje (1988) noted that if  $\text{C}_2\text{H}_2$  is present for long periods, microorganisms are able to metabolize the gas which in turn can effect  $\text{N}_2\text{O}$  reduction and cause changes in C availability that could influence denitrification measurements.

There are several different methods for introducing and maintaining  $\text{C}_2\text{H}_2$  levels in the soil atmosphere for denitrification blockage. McConnaughey and Duxbury (1986) described a method for the introduction of  $\text{C}_2\text{H}_2$  at low concentrations into the soil by diffusion. This is sometimes called static or passive diffusion. The method involves perforated tubes inserted into the soil from which the  $\text{C}_2\text{H}_2$  is allowed to diffuse into the soil. Introducing  $\text{C}_2\text{H}_2$  by pumping is also utilized. Parkin et al. (1984) compared a pumping method to static diffusion. The diffusion method was found to give lower denitrification rates which was presumably due to slower diffusion of  $\text{C}_2\text{H}_2$  and  $\text{N}_2\text{O}$ . They also noted that pumping is of a benefit since measurements can be taken rapidly and soil microorganisms are less affected by  $\text{C}_2\text{H}_2$ . Peck and Hern (1988) placed a vacuum on enclosed containers and then introduced  $\text{C}_2\text{H}_2$ . Although they used this method for  $\text{C}_2\text{H}_2$  reduction experiments, it was of a benefit since  $\text{C}_2\text{H}_2$  was



introduced rapidly by the vacuum.

Other methods of using  $C_2H_2$  inhibition include application with water or isolating soil cores. Acetylene has a high water solubility and can be introduced to a soil column by applying it with water. Hallmark and Terry (1985) applied  $C_2H_2$  through diffusion tubes and through irrigation water and obtained results that were about the same for both methods, although they noted that diffusion tubes did not function adequately in flooded soils. Acetylene inhibition can also be utilized with soil cores that are taken from the field and placed in sealed flask where  $C_2H_2$  is introduced and denitrification measurements are made. This method can be subject to a high degree of variability since it involves small volumes of soil. Rolston (1986) noted that  $C_2H_2$  can also inhibit nitrification which may not be desirable in certain cases.

### **$^{15}N$ Tracer**

Another direct method of denitrification measurement is the use of  $^{15}N$  as a tracer. Nitrogen 15 is a non-radioactive isotope, and analysis for  $^{15}N$  tracer can be conducted in gas, plant, soil and leachate samples. Several researchers have compared denitrification rates as measured by acetylene blockage methods to  $^{15}N$  methods and have found no significant differences (Parkin et al., 1985; Mosier et al., 1986). Quantification of  $NO_3^-$ -N reduction by  $^{15}N$  can be utilized to separate losses from dissimilatory and assimilatory reductions. The isotope can also be utilized to determine  $N_2$  flux. Researchers who have utilized  $^{15}N$  in denitrification

experiments have reported lag differences between  $N_2O$  and  $N_2$  flux. Rolston et al. (1976) noted that lags in  $N_2$  flux was partially due to the high  $H_2O$  solubility of  $N_2O$  which held  $N_2O$ . The lag may also be due to the fact that  $N_2O$  is the precursor of  $N_2$  in denitrification. Hauck (1986) noted that the use of the isotope for flux determinations was dependent on the uniformity of the  $^{15}N$  enrichment within the soil. It is important to recognize the N transformations and fractionations of  $^{15}N$  that take place in the soil system. Dilution of  $^{15}N$  caused by MIT needs to be determined. Karamanos and Rennie (1980) noted that most  $^{15}N/^{14}N$  fractionations occurred in the soil and not in the plant and that exchange reactions between  $NH_4^+$ -N in solution and clay colloids tended to result in  $^{15}N$  depletion of the  $NH_4^+$ -N in solution. Karamanos and Rennie (1980) observed that the conversion of  $NH_4^+$ -N to  $NO_3^-$ -N resulted in a depletion of  $^{15}N$  in the  $NO_3^-$ -N pool and an increase in the  $NH_4^+$ -N pool. Delwiche and Steyn (1970) also demonstrated fractionation between  $^{14}N$  and  $^{15}N$  in N transformations performed by several microbial species. Parkin et al. (1985) noted that it is important to sample frequently to obtain accurate ratios and depth distributions of the  $^{15}N$  pools in the soil.

## **MATERIAL AND METHODS**

### **IRRIGATION FREQUENCY STUDY**

Undisturbed soil cores were collected from within the spray circles at the Montgomery Co. site for use in the irrigation frequency study. Soil cores were removed within the spray circles in order to obtain soils which were under effluent irrigation for a period of time. The site had received effluent application for 4 months (June-Sept, 1989) before the soil cores were taken. Soil cores were collected by pressing 15 cm diameter and 30 cm long schedule 40 PVC pipe into the ground with a tractor attached Giddens hydraulic press. The PVC pipe was pushed into the soil profile to a depth of 20 cm. Twelve replications were collected from each respective loading rate (1.25 and 2.5 cm wk<sup>-1</sup>) and 4 replications were collected from the control plots. The columns containing the soil cores were then transferred to a greenhouse on VPI&SU campus. A PVC cap was attached to the bottom of each column. A 2 cm hole was placed in the center of each cap and a 2 cm diameter by 6 cm long PVC pipe was glued into each cap as a leaching outlet. Plastic screen was placed over the leaching outlet tube and approximately 450 g of sand was placed above the patch of plastic screen in the bottom of each cap. The caps were attached to the PVC column with PVC cement to insure an air tight seal. The sand was in contact with the soil at the bottom of the columns. Each column contained approximately 6-7 kg of soil.

The experiment was initiated in February 1990 and was designed as a 2 x 3 factorial. The columns received applications of secondary treatment effluent from a sand filter. The quality of effluent applied to the columns from the prototype sandfilter pretreatment is listed in Table 32. It was evident that nitrification was occurring in the sandfilter since the mean  $\text{NO}_3^-$ -N concentration of the effluent was  $32.9 \text{ mg L}^{-1}$ . The effluent treatments consisted of two loading rates (1.25 and 2.5  $\text{cm wk}^{-1}$ ) divided into 3 different daily irrigation frequencies. The three daily irrigation frequencies include: daily effluent split into 3 applications 4 hours apart (3@ irrigation frequency), daily effluent split in 2 applications 8 hours apart (2@ irrigation frequency) and total daily effluent applied in one application (1@ irrigation frequency). The control treatment consisted of columns which did not receive effluent. Each treatment was replicated four times. The amounts of effluent applied for each treatments employed are shown in Table 33. Before effluent was applied to the soil columns, each was reseeded with tall fescue grass.

An automated dosing system was constructed to apply the daily treatments to the soil columns (Figure 23). The automated system consisted of a main effluent reservoir and dosing chambers designed to apply appropriate quantities of liquid to the columns. A 21 L container was used as the main effluent reservoir chamber which stored effluent prior to dosing the individual columns. The main effluent reservoir was placed on a stand above the columns in order to supply effluent to the dosing chambers through distribution lines by gravity flow. Solenoid valves (110 v) released effluent from the main reservoir into distribution lines

constructed of 2 cm diameter PVC pipe. Effluent subsequently flowed from the distribution lines into smaller dosing chambers that were above each column. The appropriate dosing volume for each column was governed in the individual dosing chambers by an overflow line which allowed a known volume of effluent to be applied to the columns. Solenoid valves placed on the bottom of each dosing chamber were activated to apply the effluent in the dosing chambers to the columns (Figure 23). All the column and main reservoir solenoid valves were activated by timers (Radio Shack). Time delay relay switches which allowed for additional programming control were used in conjunction with the timers. This was necessary since the smallest time interval was one minute.

In order to maintain soil moisture conditions similar to field conditions the equivalent of 102 cm yr<sup>-1</sup> of rainfall (30 yr mean) for Blacksburg, VA (HISARS, 1988) was applied to the soil columns. Rainfall applications consisted of 120 mL of distilled deionized H<sub>2</sub>O applied to each column 3 times a week. Rainfall was applied by hand using a graduated cylinder. The columns were dosed for a total of 10 months (Feb-Nov 1990).

Leachate samples were collected by vacuum extraction in March, April, May and November (1990). These samples were collected in a stoppered side arm flask attached to the bottom of each column and to a vacuum pump. To collect samples, a pressure of approximately 17 kPa was applied for 3 minutes. Leachate samples were immediately analyzed for NO<sub>3</sub><sup>-</sup>-N and NH<sub>4</sub><sup>+</sup>-N concentrations using the procedures described in Part 1.

Nitrous oxide evolution from the columns was determined as a function of three different simulated rain events. Rain events equal to a normal rainfall (0.65 cm), double the normal rainfall (1.3 cm) and triple the normal rainfall (1.95 cm) for a 2 day period in Blacksburg, VA (HISARS, 1988) were applied to all of the columns for three separate denitrification sampling periods. The sampling periods under normal, double, and triple the normal rainfall conditions were performed on 8/17/90, 9/28/90 and 11/28/90, respectively. The mean temperature over the 24 h sampling periods were 25, 24, and 25° C for the normal, double, and triple rainfall conditions, respectively. At least one month of equilibration was allowed between each sampling period where the columns received the normal rainwater and effluent treatments.

Nitrous oxide emissions were measured using the  $C_2H_2$  blockage technique described earlier. Gas samples were collected over a 24 h sampling period after each rainfall event. The columns were sealed for  $N_2O$  collection in order to contain  $C_2H_2$  and  $N_2O$  gases. The top of each column was sealed with a rubber sheet fastened with hose clamps and the leachate outlet on the bottom was closed. In order to ensure  $C_2H_2$  diffusion throughout the soil column, the columns were evacuated before  $C_2H_2$  was introduced. Approximately -20 kPa pressure was maintained in the columns using a vacuum pump before  $C_2H_2$  was added. The pressure in the columns was then equalized by filling the column with  $C_2H_2$ . Immediately following  $C_2H_2$  introduction, rainfall and effluent applications were applied. Both applications were applied by needle and syringe through a septum

located in the side of the columns. The time table used for collecting gas samples within the 24 h period from the treatments is shown in Table 34. The headspace gases in each column was mixed prior to sample collection by drawing in and expelling headspace gases using a 10 mL syringe. Samples were subsequently collected by a 3 mL syringe. Gas samples were transferred to 3 mL vacutainers for storage prior to analysis. Nitrous oxide analysis was performed using procedures described in Part 1. After every 3 gas samples collected, 30 mL of  $C_2H_2$  was injected into the columns to replace the gas volume collected during sampling.

Table 32. Effluent quality applied to columns in the irrigation frequency study.

pH	EC	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	TKN
	S m <sup>-1</sup>	-----mg L <sup>-1</sup> -----		
8.3	0.277	32.9	7.5	11.5



Table 33. The amount of effluent applied in the irrigation frequency treatments.

Loading Rate	Irrigation Frequency	Label	Effluent Application
<i>cm wk<sup>-1</sup></i>	<i>applications day<sup>-1</sup></i>		<i>mL application<sup>-1</sup></i>
2.50	1	1@	66.0
2.50	2	2@	33.0
2.50	3	3@	22.0
1.25	1	1@	33.0
1.25	2	2@	16.5
1.25	3	3@	11.0
0	0		0

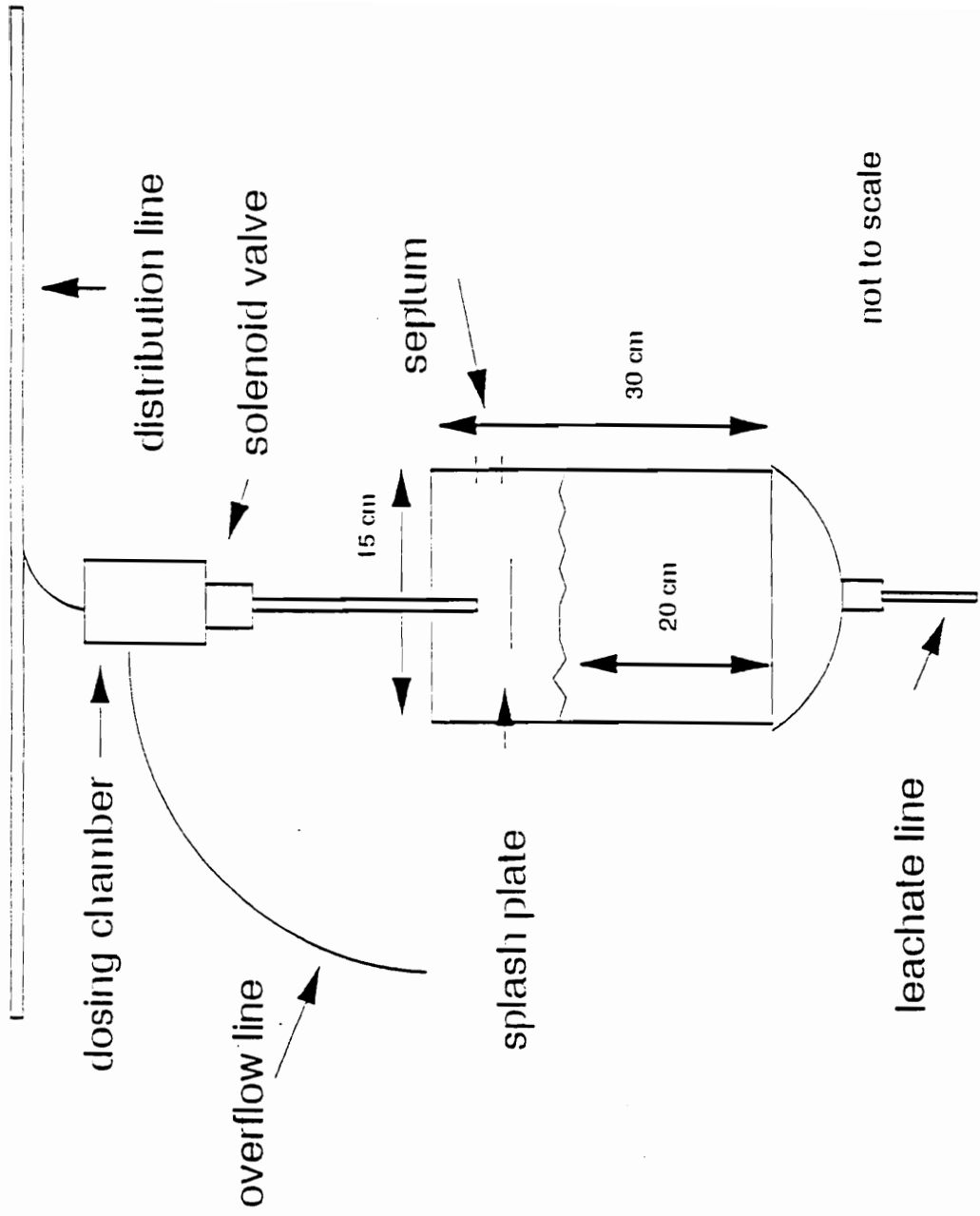


Figure 23. Schematic of a column dosing system.

Table 34. Sampling time table for column denitrification measurements.

-----Irrigation Frequency-----		
1@	2@	3@
-----Sampling time (h )-----		
<i>(DOSE) † 0</i>	<i>(DOSE) 0</i>	<i>(DOSE) 0</i>
0.5	0.5	0.5
1.0	1.0	1.0
3.0	3.0	3.0
		<i>(DOSE) 4.0</i>
		4.25
		4.5
5.0	5.0	5.0
	<i>(DOSE) 8.0</i>	<i>(DOSE) 8.0</i>
	8.25	8.25
	8.5	8.5
9.0	9.0	9.0
	11.0	11.0
13.0	13.0	13.0
24.0	24.0	24.0

† Dose amounts = 1@ : 33 or 66 ml.  
 2@ : 16.5 or 33 ml.  
 3@ : 11 or 22 ml.

Oxidation-reduction potentials in the column soils were measured at the 5 cm depth at different times after rainfall applications. Redox measurements were made 0.5 and 24 h after rainfall applications and 2 h after effluent applications. Redox electrodes were constructed as described by Mueller et al. (1985). The electrodes were placed in the soil for 5 days prior to potential measurements. The reference electrode was placed in a 5 cm deep hole by 2 cm diameter. A porous ceramic cup filled with a saturated KCL solution was used for protection of the reference electrode. The reference electrode and ceramic cup were placed in a column when a redox measurement was taken. The soil removed from each column for the reference hole was measured for pH using a 1:1 soil water ratio using the procedures described in part 1. Potential measurements were corrected to pH 7.0 and calibrated using Zobell solution (Langmuir, 1971).

Plant tissue samples (fescue) in the columns were collected 7 times during the irrigation frequency experiment. Tissue samples were collected in February, March, May, July, August, October and December. Samples were taken by removing the top growth (2 cm above soil surface). The samples were then oven dried, weighed and analyzed for TKN as described in part 1.

Effluent and rainfall applications were continued for 19 days after the last denitrification sampling period before soil samples were collected from the columns. Grass cover including the thatch layer was first removed from the columns. Soil cores 2 cm in diameter were collected at 0-7, 7-14 and 14-20 cm depths. The soil cores were placed in sterilized plastic bags and stored in a

freezer until analysis. Soil samples were measured for moisture,  $\text{NO}_3^-$ -N, and  $\text{NH}_4^+$ -N content using the procedures referred to in Part 1. Additional soil cores were collected and placed in sterilized plastic bags and stored in an incubator for a denitrification capacity study.

### DENITRIFICATION CAPACITY STUDY

The denitrification capacity of the soils used in the greenhouse study was determined in a laboratory microcosm study. This study was designed to determine if C or  $\text{NO}_3^-$ -N was limiting denitrification in soils which had received different frequencies of effluent application for 10 months. The soils had received irrigation frequency treatments (1@, 2@, and 3@ application  $\text{day}^{-1}$ ) across 2 effluent loading rates of 2.5, 1.25  $\text{cm wk}^{-1}$ . A control which received no effluent application was also evaluated. The effluent loading treatment of 2.5  $\text{cm wk}^{-1}$ , split into 2 applications a day, was omitted from the study for use in a  $^{15}\text{N}$  tracer study. Each soil was evaluated for denitrification capacity in a 3x2 factorial experiment using 3 rates of C and 2 rates of  $\text{NO}_3^-$ -N. The 6 treatments used in the factorial experiment are listed in Table 35. Treatment were replicated 3 times for each soil. The rates of C addition were 0, 900 and 1800  $\mu\text{g C g}^{-1}$  soil added as glucose. Rates of  $\text{NO}_3^-$ -N addition were 0 and 84  $\mu\text{g g}^{-1}$  soil as  $\text{KNO}_3$ . For each replication approximately 10 g of soil was placed in 125 mL Erlenmeyer flasks. An air tight septum was placed in the neck of each flask. Each flask was flushed with Helium

(He) gas for 1 min at a flow rate  $>10 \text{ L min}^{-1}$  while the flask was shaken by hand in order to remove  $\text{O}_2$  and create an anaerobic environment. The flasks were incubated in the dark at  $20^\circ \text{C}$ . Gas samples were removed from each flask after 24 and 48 h incubations. Gas samples were removed with 3 mL syringes by withdrawing and injecting the headspace in the flask 10 times before removal of the sample. Samples were transferred from the syringe to 3 mL vacutainers for storage until analysis. Analysis for  $\text{N}_2\text{O}$  concentration were determined in the same manner as described in Part 1.

Table 35. Carbon and Nitrate treatments used in the denitrification capacity study.

Treatment	Carbon	Nitrate-N
	----- $\mu\text{g g}^{-1}$ soil-----	
1	0	0
2	0	84
3	900	0
4	900	84
5	1800	0
6	1800	84

## **<sup>15</sup>N TRACER STUDY**

A <sup>15</sup>N tracer study was performed on 4 of the greenhouse columns which had received secondary treated domestic wastewater at a rate of 1.25 cm wk<sup>-1</sup> for approximately 10 months. A synthetic secondary treated wastewater containing 29.9 mg L<sup>-1</sup> NO<sub>3</sub><sup>-</sup>-N as KNO<sub>3</sub> and 72 mg L<sup>-1</sup> of anhydrous glucose (dextrose corn sugar)(C<sub>6</sub>H<sub>12</sub>O<sub>6</sub>) was applied to each column for 48 days at 66 mL per day. During the last week, the reagent grade KNO<sub>3</sub> was replaced with KNO<sub>3</sub> containing a 99% enrichment of <sup>15</sup>N. The columns were kept at room temperature and received the enriched solution for 8 days at the same rate of application as the unenriched solution. On day 1 the enriched solution was applied and no leachate or gas samples were taken. The enriched solution was surface applied each day by hand when the columns were not sealed for gas sampling.

Denitrifying gas samples were taken from the columns on days 2 and 8. Gas samples were collected from the columns as described earlier in part 1. The enriched solution was injected into each column by a 60 mL syringe through an airtight septum on the side of the columns. Gas samples were collected from each column after being sealed for 24 h. The headspace gases in each column were mixed prior to gas sample collection by drawing in and expelling headspace gases using a 60 mL syringe. Duplicate 10 mL gas samples were then removed and placed in 10 mL vacutainers prior to mass spectral analysis for <sup>15</sup>N<sub>2</sub>. The headspace in each column was then measured by the pressure/volume method



described in part 1. Mass spectral analysis was conducted by Isotec Analytical Services of Miamisburg, Ohio.

Leachate samples from the bottom of the columns were collected from day 2 through 8 during the tracer study. Leachate collection tubing was closed during the entire tracer study except for approximately 10 minutes each day for leachate collection. The volume of leachate collected from each column was measured and immediately frozen until analysis.

In order to properly measure  $^{15}\text{N}$  content by a direct combustion mass spectrometer, total N in the samples must be within a range of 25 to 100  $\mu\text{g}$ . Three mL aliquots of the leachate samples were analyzed for  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N by an auto-analyzer as described in part 1 in order to determine the proper dilutions needed for mass spectral analysis. Analysis and sample preparation for  $^{15}\text{N}$  measurements were modified from the procedure described by Kelley et al. (1991). Aliquots of the leachate samples were diluted 25 fold and placed in 140 mL plastic specimen containers. Ammonium was volatilized from the samples by adding magnesium oxide (MgO) to the samples (0.2 g per 40 mL) in order to measure the  $^{15}\text{NO}_3^-$ -N content. The samples were then left uncovered for 3 days for volatilization of  $\text{NH}_4^+$ -N. The remaining  $\text{NO}_3^-$ -N in solution was reduced to  $\text{NH}_4^+$ -N by placing 0.4 g of Devarda's alloy to each sample (Kelley et al., 1991). Glass fiber disks were placed on stainless steel wire and suspended in the containers in order to absorb diffusing  $\text{NH}_3$ , the containers were then immediately sealed with caps. Glass fiber disks were prepared by cutting acid washed Whatman GF/D

glass fiber filter paper with a standard paper punch. Ten  $\mu\text{L}$  of 2.5 M  $\text{KHSO}_4$  was pipetted onto the fiber disk for absorption of diffused  $\text{NH}_3$ . The disks were placed on "S" shaped hooks of stainless steel wire and suspended about 2 cm above the fluid (Kelly et al., 1991). The samples remained sealed at room temperature for 8 days. After 8 days the disk were removed and placed on a rack inside a desiccator and allowed to dry for 2 days. The disks were prepared for mass-spectral analysis by placing in individual 4 x 6 mm Sn foil cups which were then folded to enclose the disks. The disks were analyzed for N recovery and  $^{15}\text{N}$  concentration by automated N and C analysis/mass spectrometry (ANCA/MS) at TVA's National Fertilizer and Environmental Research Center (NFERC).

Triplicate soil cores 3 cm in diameter and 21 cm in length were collected from the columns after the tracer study. Soil cores were separated into depth increments of 0-7, 7-14, and 14-21 cm. Approximately 50 mg soil subsamples from the cores were weighed on a 7 place g balance in the Chemistry Dept at VPI&SU, and placed in 4 x 6 mm Sn foil cup and folded to enclose the soil. The samples were analyzed for N recovery and  $^{15}\text{N}$  concentration by ANCA/MS at NFERC. The percentage of  $^{15}\text{N}$  recovery in the soil was based on the soil weight of the columns. The soil weight at each depth increment was estimated from bulk density measurements taken from the field soil profile. The total weight of the soil in the columns was directly measured after the tracer study. Plant growth in the columns was minimal. After the removal of the fescue cover at the conclusion of the irrigation frequency study the grass cover never recovered fully while the

columns were in the lab environment.

## **RESULTS AND DISCUSSION**

### **IRRIGATION FREQUENCY STUDY**

#### **N in Leachate From Columns**

The amount of N collected in the leachate exiting the soil columns (Table 36) showed no differences between the effluent application rates for each of the sampling dates. However, the amount of N collected represented 0.1 to 30.8% of the N applied in the effluent. For each sampling date for both the 1.25 and 2.5 cm wk<sup>-1</sup> application rates, the lowest amount of leachate N collected was from columns which received the 3@ irrigation frequency application treatment.

These results demonstrate that the more frequent applications of smaller volumes of effluent reduced the amount of N leached. By reducing individual application volumes, a larger portion of the applied effluent would remain near the soil surface thus reducing the leachate volume. In addition, evapotranspiration would have more influence on effluent movement when smaller volumes are applied, since evapotranspiration losses may be higher when more effluent was maintained near the soil surface. Since the topsoil is the most active biological zone in the profile, a higher portion of effluent in the topsoil might have increased N losses due to denitrification, immobilization and plant uptake. These results are in agreement with Rolston et al. (1980) who also observed lower leachate losses with more frequent applications using smaller volumes of irrigation water.

The mean  $\text{NO}_3^-$ -N concentrations in the leachate for all irrigation treatments were below  $10 \text{ mg L}^{-1}$  (Table A-5) which is in agreement with  $\text{NO}_3^-$ -N concentrations collected from subsurface waters in the field studies (Tables 15 and 16). The  $\text{NH}_4^+$ -N concentrations of the leachate were all below  $1 \text{ mg L}^{-1}$  (Table A-5). There was no evidence of deep effluent percolation of  $\text{NH}_4^+$ -N as a result of effluent application. This was not unexpected since  $\text{NH}_4^+$ -N ions are subject to biological transformations, electrostatic retention, and fixation in the soil.

### **Influence on Column Soil pH and Redox Potentials**

There was an increase in column soil pH in the treatments that received wastewater (Table 37). Control columns had an average pH of 5.8, while the effluent applied columns had pH means ranging from 7.0 to 8.5. The increased pH was probably due to the high pH of the applied effluent. A trend between the irrigation frequency treatments and soil pH levels could not be established. There were also large differences in the redox potentials between the control and effluent applied columns. The redox potentials of the control columns were approximately +500 mv while the redox potentials of the effluent applied columns ranged between +93 and -130 mv (Table 37). The redox potentials appeared to be stable since there was no change in each treatment immediately following a rainfall application. The redox potentials were probably influenced more by the application of effluent that occurred 2 h before the measurements than the rainfall applications. The redox potentials in all the effluent application treatments were below the range

where denitrification occurs (Meek et al., 1969; Magdoff et al., 1974; Smith et al., 1983). Some of the more negative values were close to redox potentials observed in waterlogged soils (Langmuir, 1971). Since the redox electrodes were inserted only to a depth of 5 cm, the measurements do not imply that the entire soil volume was anaerobic.

Table 36. Total inorganic N leached from columns and representative recovery percentage of applied N.

Loading Rate	Irrigation Frequency	Mar 90		Apr 90		Nov 90	
		Total Leached†	Recovery‡	Total Leached	Recovery‡	Total Leached	Recovery‡
cm wk <sup>-1</sup>		mg N	%	mg N	%	mg N	%
1.25	1	0.05 a*	3.7	0.03 a	2.2	0.40 a	30.1
	2	0.41 a	30.8	0.23 a	17.3	¶	
	3	0.01 a	0.7	0.002 a	0.1	0.01 a	0.7
2.5	1	0.24 a	9.0	0.33 a	12.4	0.19 a	7.1
	2	0.23 a	8.6	0.30 a	11.3	0.07 a	2.6
	3	0.01 a	0.3	0.01 a	0.3	0.02 a	0.7
Control		0.36 a		0.07 a		0.04 a	

\* Means within the same column followed by the same letter are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range test.

† Total inorganic N concentration times the volume of leachate collected.

‡ Recovery percentage of the effluent N applied in a daily application.

§ Applications per day of applied effluent.

¶ No leachate collected.

Table 37. Soil pH and redox potentials in columns after 10 months of effluent application.

Loading Rate	Irrigation Frequency	pH	Redox Potential	
			0.5 h <sup>†</sup>	24 h <sup>‡</sup>
cm wk <sup>-1</sup> -----mv-----				
1.25	1	7.0 c*	44 bc	64 bc
	2	8.0 ab	-124 c	-130 d
	3	7.0 c	10 bc	-13 bcd
2.5	1	7.8 b	-74 bc	-81 d
	2	8.1 ab	-61 bc	-56 cd
	3	8.5 a	93 b	89 b
Control		5.8 d	526 a	509 a

<sup>†</sup> Applications per day.

<sup>‡</sup> Readings taken 2 h after effluent application and 24 h after a rainfall application.

<sup>¶</sup> Readings taken 2 h after effluent application and 0.5 h after a rainfall application.

\* Means within the same column followed by the same letter are not significantly different ( $P_{\leq 0.05}$ ) as determined by Duncan's Multiple Range test.



## **Nitrous Oxide Accumulations**

The amount of accumulated  $N_2O-N$  in the columns was generally similar at both the normal and double normal rainfall conditions (Figures 24 and 25). Nitrous oxide emissions were zero for each measurement in the control columns at both the normal and double normal rainfall conditions, which indicates that denitrification losses from these columns were strongly influenced by effluent application. The highest amount of  $N_2O-N$  accumulation was observed at the 2@ irrigation frequency for the  $2.5 \text{ cm wk}^{-1}$  effluent application rate for both rainfall conditions. At the normal rainfall condition, the  $N_2O-N$  produced at the 2@ irrigation frequency at the  $2.5 \text{ cm wk}^{-1}$  rate of effluent application was over 4 times greater than the next highest value and represented 122% of the N applied in that sampling period (Table 38). The high rates might have been caused by pockets of high denitrification activity or "hot spots" such as those noted by Parkin (1987) and Christensen et al. (1990) who attributed elevated  $N_2O-N$  emission rates to organic particulates present in the soil. The redox potential under this rate of application was approximately -60 mv (Table 37) which indicates that a strong reducing environment was present in the soil. The 2@ irrigation frequency application at the  $2.5 \text{ cm wk}^{-1}$  rate of effluent application continued to have the largest amount of  $N_2O-N$  accumulation in the double normal rainfall condition (Table 38).

At triple the normal rainfall condition, the  $N_2O-N$  accumulations were much lower compared to the normal and double normal rainfall conditions (Figure 26).

There was also no  $\text{N}_2\text{O-N}$  emissions from the control columns at this rainfall condition. The total amount of accumulated  $\text{N}_2\text{O-N}$  was very similar for each irrigation frequency treatment and effluent loading rate at the triple normal rainfall condition. A trend in the amount of accumulated  $\text{N}_2\text{O-N}$  among the irrigation frequencies could not be established. The low accumulations at the triple normal rainfall condition are attributed to the effect of the large rainfall application in a short period which probably enhanced  $\text{NO}_3^-$ -N movement past the microbially active zone in the topsoil and thus reduced  $\text{N}_2\text{O-N}$  emissions.

At the  $1.25 \text{ cm wk}^{-1}$  application rate, the 1@ irrigation frequency application had the highest accumulations in the double and triple normal rainfall conditions. While the 2@ irrigation frequency application had the highest accumulations at the  $2.5 \text{ cm wk}^{-1}$  application rate for all 3 rainfall conditions (Table 38). These results may be indicative of more favorable denitrification conditions under 2@ irrigation frequency application at the  $2.5 \text{ cm wk}^{-1}$  effluent application rate and in the 1@ irrigation frequency application at the low effluent application rate. When small quantities of effluent are applied, the total amount of denitrified N might be reduced by dividing the effluent application into smaller but more frequent applications. The smaller volumes of effluent applied at one time possibly diminishes the total  $\text{N}_2\text{O-N}$  emission because of a reduced anaerobic soil volume when compared to larger effluent application rates. While these results may be inconclusive, they do, however, illustrate the potential for optimizing N loss by changing the frequency of effluent application. Changes in the frequency of irrigation application most

probably influences  $N_2O$ -N emission rates by changing the volume of anaerobic soil and the duration of anaerobic conditions. Also the availability of  $NO_3^-$ -N in zones favorable to denitrification may be altered.

Accumulated concentrations of  $N_2O$ -N during the 24 h sampling periods for the normal, double and triple normal rainfall conditions were higher under the 2.5  $cm\ wk^{-1}$  application rate compared to the 1.25  $cm\ wk^{-1}$  rate (Figures 24, 25 and 26). This result was not unexpected since a higher amount of N as well a larger volume of effluent was applied at the 2.5  $cm\ wk^{-1}$  rate. The larger volume of effluent at the high application rate could have promoted longer anaerobic conditions compared to the lower application rate.

The coefficients of variation for the accumulated  $N_2O$ -N values ranged between 19 and 161%, with a majority of the measurements under 100% (Table 38). The coefficients are typical for soil denitrification measurements (Folorunso and Rolston, 1985a). The significance levels from the analysis of variance for the irrigation frequency treatments with each time increment are presented in Table 39. At the normal rainfall condition, there were significant differences between loading rates and application frequencies as well as a rate-frequency interaction, 5 h after the first effluent application. The analysis of the normal rainfall condition were most likely skewed by the 2@ irrigation frequency treatment which was suspected to contain pockets of high denitrification activity. This deduction was support the double and triple normal rainfall conditions which show only significant differences between loading rates and not between application frequencies.

When considering the average N<sub>2</sub>O emissions for each rainfall condition, the recovery rates of the N applied related to denitrification averaged 37% in the normal rainfall condition, 34% in the double normal rainfall condition and 5% in the triple normal rainfall condition. The recovery rates of the applied N for the normal and double normal rainfall conditions (Table 38) were generally much higher than those observed at the Montgomery Co. field study from Part 1. However the N recovery rates from denitrification in the triple normal rainfall condition (3-8%) were similar to rates collected in the Montgomery Co. field study (5%). The similar recovery rates in the triple normal rainfall condition and the field study are probably not comparable since the column study was more suitable than the field study for measuring denitrification rates. Denitrification "bursts" may have been missed in the field study. In addition the amount of water applied at one time to the columns in the triple normal rainfall condition was much higher than what the Montgomery Co. site field site would receive in the same short period of time from a rainfall event.

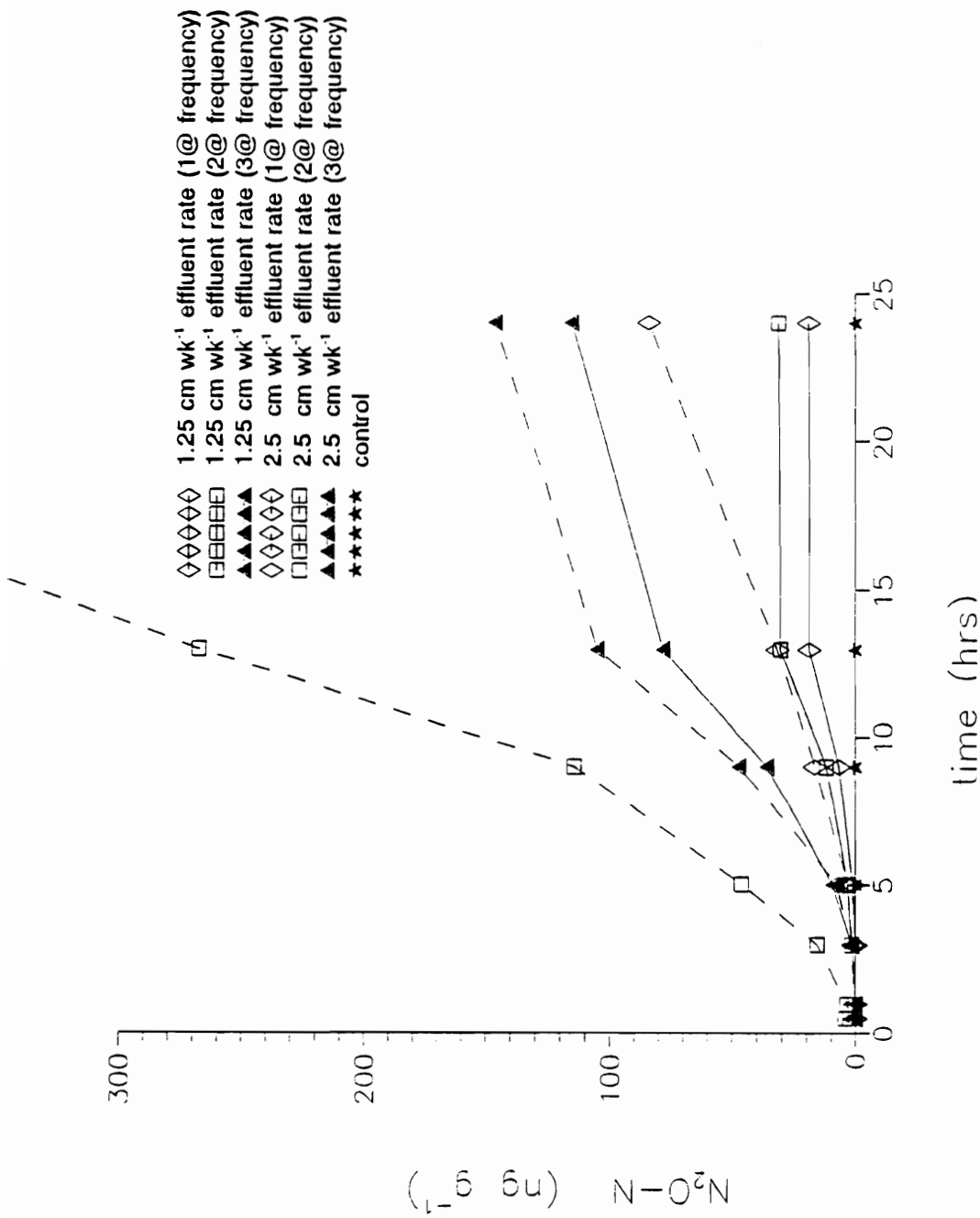


Figure 24. N<sub>2</sub>O-N accumulations as a function of irrigation frequency treatments with time in the normal rainfall condition.

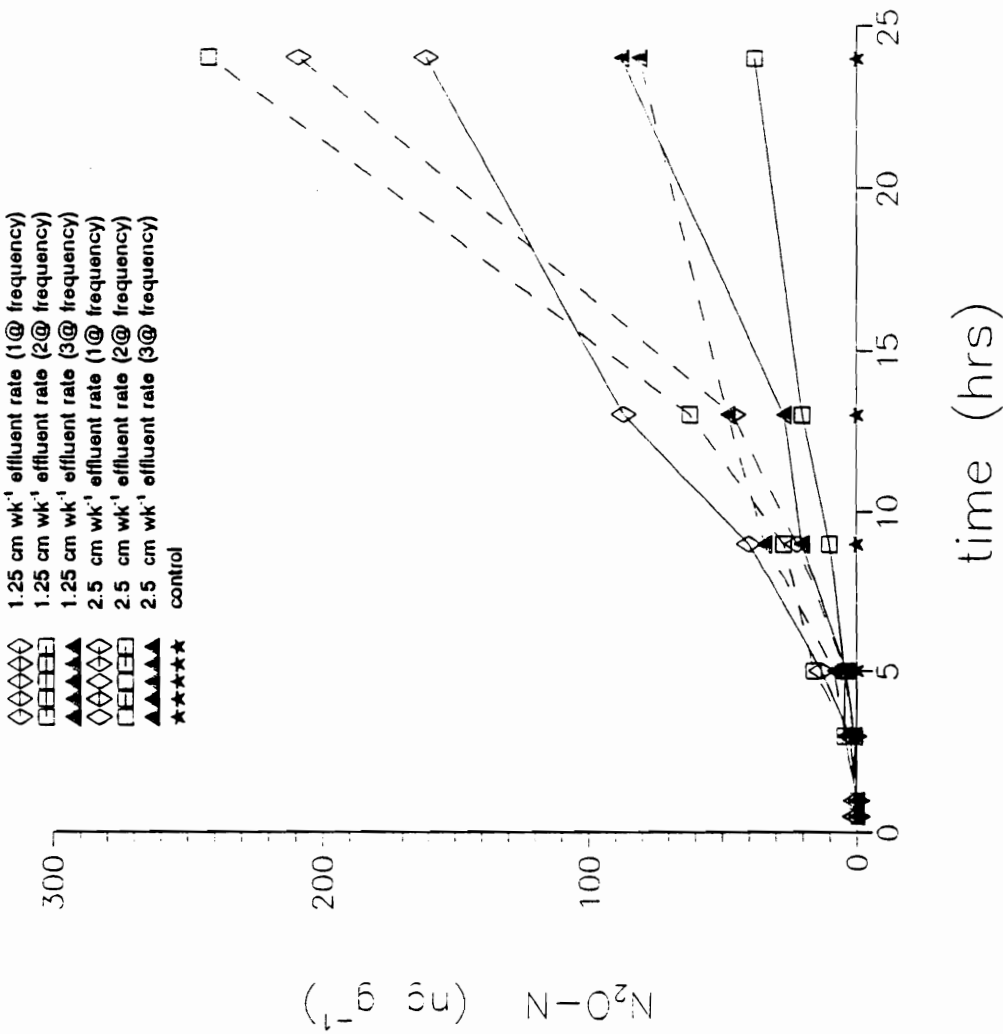


Figure 25. N<sub>2</sub>O-N accumulations as a function of irrigation frequency treatments with time in the double normal rainfall condition.

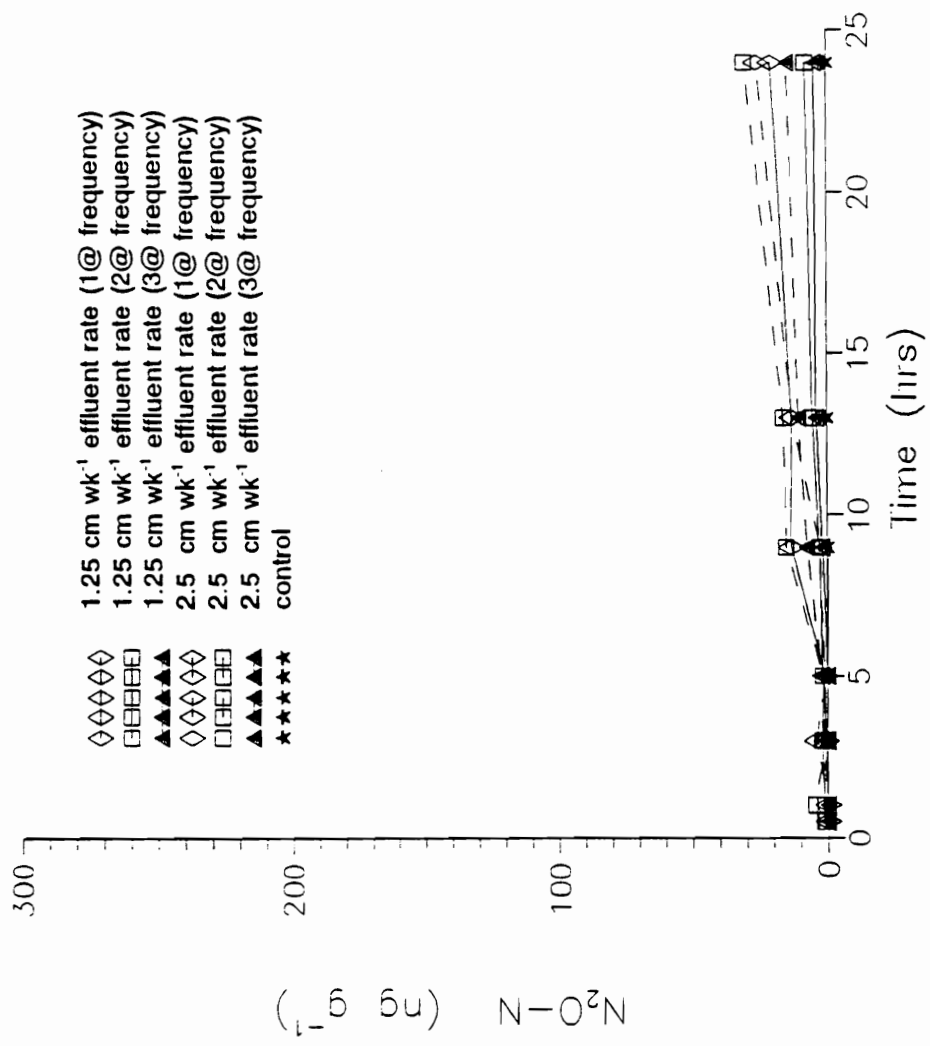


Figure 26. N<sub>2</sub>O-N accumulations as a function of irrigation frequency treatments with time in the triple normal rainfall condition.

Table 38. Nitrous oxide accumulations after 24 h and recovery rates of effluent N for the irrigation frequency treatments.

Rate	Irrigation Frequency	n§	Mean	CV‡	Recovery of N applied†
cm wk <sup>-1</sup>	appl day <sup>-1</sup>		ng g <sup>-1</sup> soil	-----%-----	
-----1x Rainfall†-----					
control	0	4	0 b*	-	-
	1	3	19 b	161	7
1.25	2	2	31 b	65	12
	3	4	115 b	101	41
2.5	1	3	84 b	130	15
	2	4	636 a	37	122
	3	4	146 b	70	28
-----2x Rainfall-----					
control	0	4	0 c	-	-
	1	4	161 abc	78	58
1.25	2	3	38 bc	144	14
	3	4	89 abc	99	32
2.5	1	3	209 ab	72	38
	2	4	242 a	82	46
	3	4	81 abc	65	15
-----3x Rainfall-----					
control	0	4	0 c	-	-
	1	3	21 abc	151	8
1.25	2	3	8 abc	19	3
	3	4	4 bc	59	2
2.5	1	4	26 ab	84	5
	2	4	30 a	24	6
	3	4	15 abc	46	3

\* Means within the same rainfall treatment followed by the same letter are not significantly different ( $P_{\geq 0.05}$ ) as determined by Duncan's Multiple Range test.

† Based on the application of effluent inorganic N within the same 24 h period.

‡ Coefficient of variation.

§ Number of samples.

¶ 1x rainfall = 0.65 cm.



Table 39. Significance level (probability value) of F ratios from analysis of variance for N<sub>2</sub>O emissions from soil columns.

Source	time						
	-----hours-----						
	0.5	1	3	5	9	13	24
Normal Rainfall‡							
Rep	NS†	NS	NS	NS	NS	NS	NS
Rate	NS	NS	NS	<0.05	<0.05	<0.01	<0.01
Frequency	NS	NS	NS	<0.05	<0.05	<0.01	<0.01
Rate*Frequency	NS	NS	NS	<0.05	<0.05	<0.01	<0.01
2x Normal Rainfall							
Rep	NS	NS	NS	NS	NS	NS	NS
Rate	NS	NS	NS	NS	NS	<0.05	<0.05
Frequency	NS	NS	NS	NS	NS	NS	NS
Rate*Frequency	NS	NS	NS	NS	NS	NS	NS
3x Normal Rainfall							
Rep	NS	NS	NS	NS	NS	NS	NS
Rate	NS	NS	NS	NS	NS	NS	<0.05
Frequency	NS	NS	NS	NS	NS	NS	NS
Rate*Frequency	NS	NS	NS	NS	NS	NS	NS

† NS indicates not significant

‡ Normal rainfall application = 0.65 cm

## **Nitrous Oxide Emission Rates**

### ***Normal Rainfall Condition***

When examining the rate of denitrification with each time increment within the 24 h sampling periods, different behaviors were observed among the irrigation frequency treatments (Figures 27 to 35). When effluent was applied in a single application (1@ frequency) at the 1.25 cm wk<sup>-1</sup> application rate there was a small N<sub>2</sub>O-N emission rate increase which started approximately 3 h after effluent dosing and continued up to 13 h (Figure 27). The emission rate increase was probably not detected until 3 h after the first dose due to N<sub>2</sub>O-N concentrations, in the headspace of the columns, being below measurable levels. When the effluent application was split into 2 applications 8 h apart (2@ frequency), the N<sub>2</sub>O-N emission rate after the first application (time 0 h) was very low, however, immediately following the second effluent application (time 8 h) there was a sharp increase N<sub>2</sub>O-N emission (Figure 28). The N<sub>2</sub>O-N emission rate subsequently decreased after 13 h as it did in the 1@ frequency treatment. When the same application rate (1.25 cm wk<sup>-1</sup>) was divided into 3 applications per day (3@ frequency), sharp increases in the N<sub>2</sub>O-N emission rate were also observed after effluent doses at 4 and 8 h (Figure 29). The emission rate subsequently decreased after 13 h as it did in the 1@ and 2@ irrigation frequency treatments. Although sharp emission rate increases were observed when the effluent was split into 2 or 3 applications per day, the high emission rates lasted only 1-3 h. Even

though the treatments behaved differently at the 1@ frequency application exhibiting a low but steady emission rate and the 2@ and 3@ frequency treatments exhibiting high but short emission rates, the total accumulations of N<sub>2</sub>O-N after 24 h were not different (Table 38). However, there was a trend of increased N<sub>2</sub>O-N accumulation with more frequent applications.

At the high effluent application rate (2.5 cm wk<sup>-1</sup>) N<sub>2</sub>O-N emission rates exhibited small increases for the entire 24 h period at the 1@ irrigation frequency treatment (Figure 27). At the 2@ irrigation frequency application (Figure 28), N<sub>2</sub>O-N emission rates were much higher than at the 1@ frequency treatment. Emission rates at the 2@ frequency also increased greatly after the second effluent dose at 8 h. This high emission rate continued for the 24 h period as it did at the 1@ frequency treatment, thus resulting in a large total N<sub>2</sub>O-N accumulation. This result demonstrates that the suspected pockets of high denitrification activity in this treatment were active beyond 24 h after effluent dosing. Nitrous oxide emissions also increased sharply after each effluent dose when the application was divided into 3 segments (3@ frequency) at the 2.5 cm wk<sup>-1</sup> application rate. However, the N<sub>2</sub>O-N emission rate did not remain at the peak levels for the entire 24 hour period as it did for the 1@ and 2@ irrigation frequency treatments.

Conclusions on the effect of irrigation frequency at the 2.5 cm wk<sup>-1</sup> loading rate are hindered by the likelihood of high denitrification activity pockets in some of the columns. However, it is noted that there was higher emission rates at the 2.5 cm wk<sup>-1</sup> application rate and that emission rates decreased faster at the 1.25

cm wk<sup>-1</sup> application rate. These results are attributed to the lower amount of NO<sub>3</sub><sup>-</sup>-N applied and the potential for less anaerobic conditions at the 1.25 cm wk<sup>-1</sup> application rate. These data agree with reports in the literature of denitrification emission rates behaving in spike-like functions due to changes in soil moisture (Rolston et al., 1982).

### ***Double Normal Rainfall Condition***

The N<sub>2</sub>O-N emission rates at the double normal rainfall condition behaved similar to emission rates at the normal rainfall condition. The rate of N<sub>2</sub>O-N emission at the 1@ irrigation frequency applications behaved very similar in the normal and double normal rainfall conditions (Figures 27 and 30). Both conditions exhibited a continuous increase in the N<sub>2</sub>O-N emission rate over the 24 h period for the 2.5 cm wk<sup>-1</sup> application rate while the 1.25 cm wk<sup>-1</sup> application rate decreased between 13 and 24 h. These results are probably indicative of the higher volume of liquid applied at the 2.5 cm wk<sup>-1</sup> high rate treatment which caused a longer anaerobic period in the microbial active zone near the soil surface even though a higher amount of rainwater was applied at the double normal rainfall condition. Both the 2@ and 3@ irrigation frequency applications in the double normal rainfall condition (Figures 31 and 32) exhibited short time periods of high N<sub>2</sub>O-N emission rates after effluent doses as they did in the normal rainfall condition.

### ***Triple Normal Rainfall Condition***

The  $\text{N}_2\text{O}$ -N emission rates in the 1@ irrigation frequency application at the triple normal rainfall condition (Figure 33) did not exhibit a continuous rate increase as it did in the normal and double normal rainfall conditions, which was probably due to the deeper movement of  $\text{NO}_3^-$ -N in the columns at the high rainfall condition. There were sharp  $\text{N}_2\text{O}$ -N emission rate increases after the 8 h dose in the more frequent effluent applications (2@ and 3@) (Figures 34 and 35) as in the double and normal rainfall conditions. However, these sharp rate increases decreased quickly (after 1 h). The sharper  $\text{N}_2\text{O}$  emission spikes in the triple normal rainfall condition may be due to the diffusion of  $\text{NO}_3^-$ -N away from the active microbial zone due to the larger amount of rainwater applied to the columns.

### **Plant N Uptake in Columns**

Plant N uptake was increased by increasing the application rate of effluent, but was not changed by the different irrigation frequency applications (Figure 36). Plant N uptake was higher in both effluent application rates than in the control columns. These results were similar to the plant N uptake rates in the Montgomery Co. and Alleghany Co. field studies which showed increased plant N uptake with increased effluent application. Any changes in the soil and water N content caused by the different frequency applications of effluent were apparently too minimal to be reflected in the N content of the grass cover.

### **Soil N Content in Columns**

There were small differences between irrigation frequency treatments in soil  $\text{NH}_4^+$ -N or  $\text{NO}_3^-$ -N concentrations after 10 month of effluent application to the column soils (Table 40). However the  $\text{NH}_4^+$ -N content was higher at all three soil depths measured for the control than where sand filter effluent was applied to the soil columns. One possibility for the higher  $\text{NH}_4^+$ -N concentrations is that the microbial populations were not as large in the control since the moisture content was always much lower than in the columns that received effluent. Since the populations might be lower, nitrification might have been less efficient. The pH was also much lower in the control columns compared to the columns which received effluent applications (Table 37). A low pH might have inhibited some nitrification in the control columns. The soil data also demonstrates that there was little residual  $\text{NO}_3^-$ -N left in the soil, which agrees with the Montgomery Co. and Alleghany Co. field data.

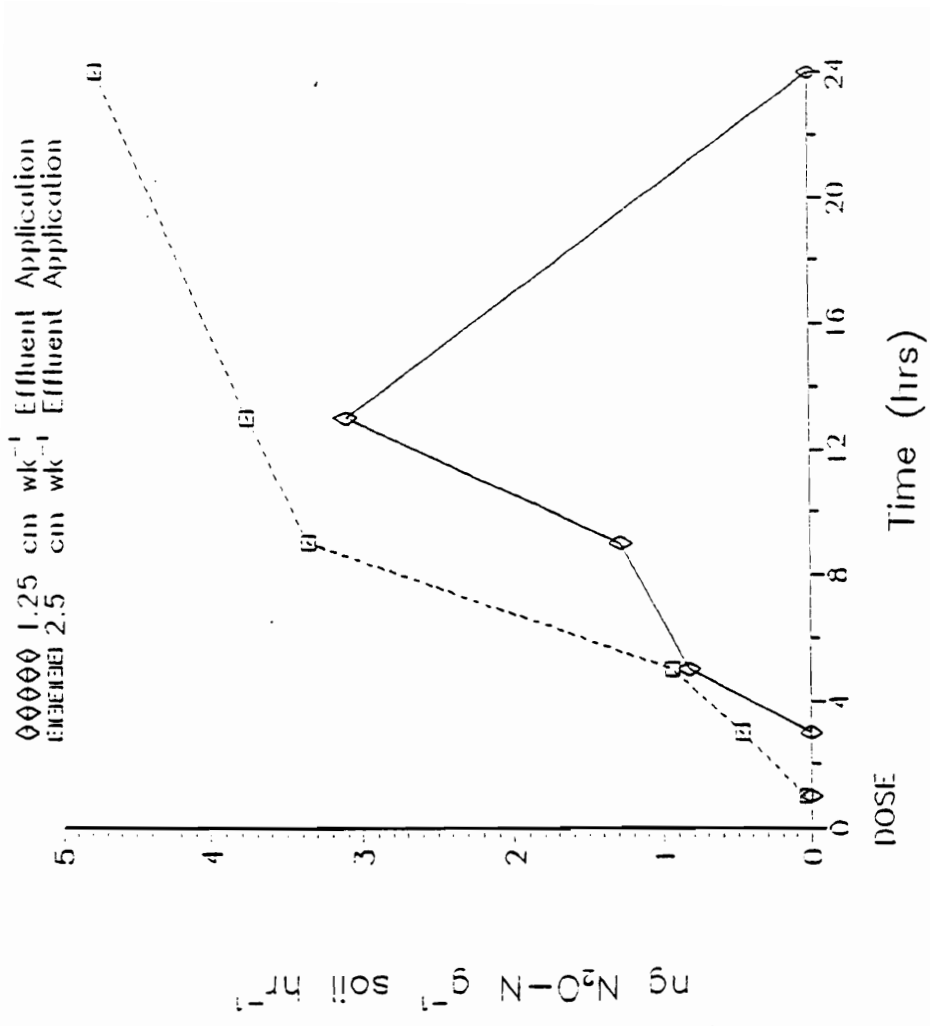


Figure 27. Nitrous oxide emission rates from the 1@ irrigation frequency application treatment under the normal rainfall application (0.65 cm).

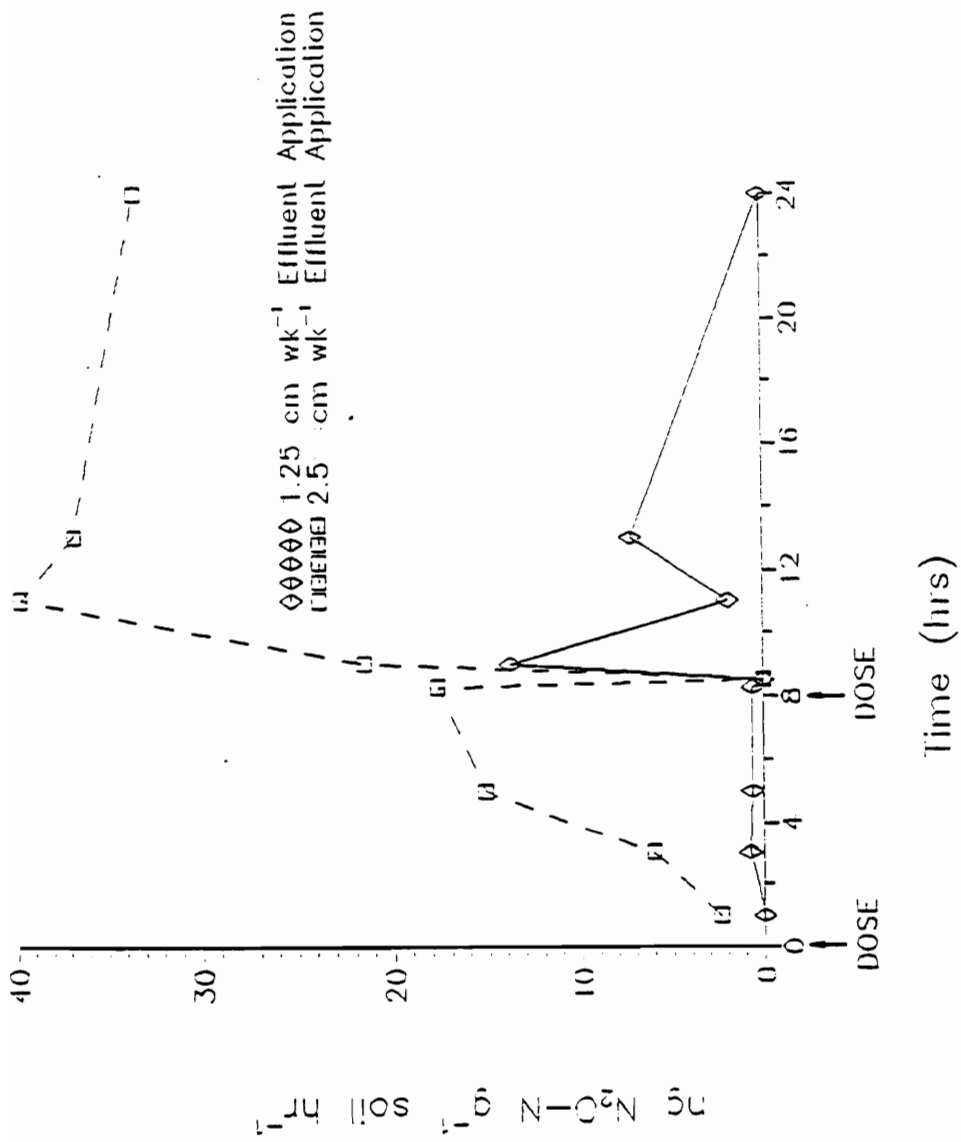


Figure 28. Nitrous oxide emission rates from the 2@ irrigation frequency application treatment under the normal rainfall application (0.65 cm).





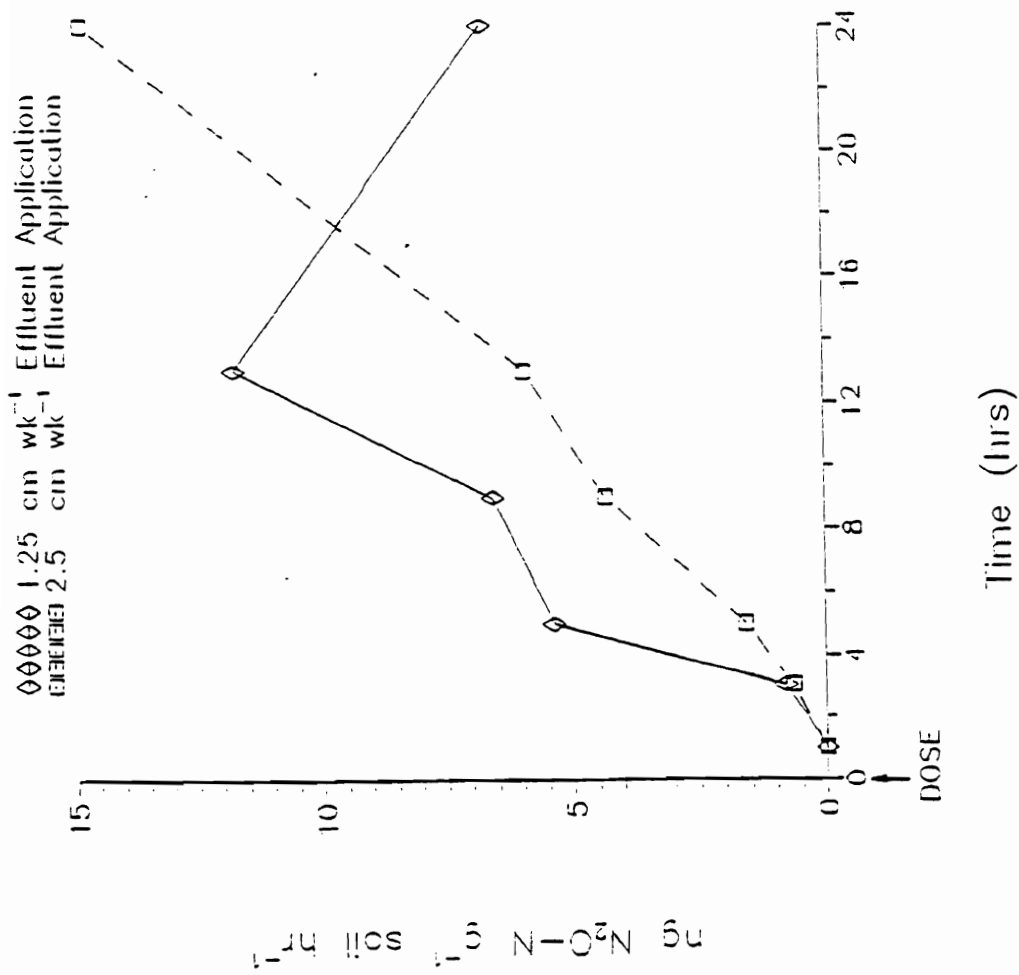


Figure 30. Nitrous oxide emission rates from the 1@ irrigation frequency application treatment at double the normal rainfall application (1.3 cm).

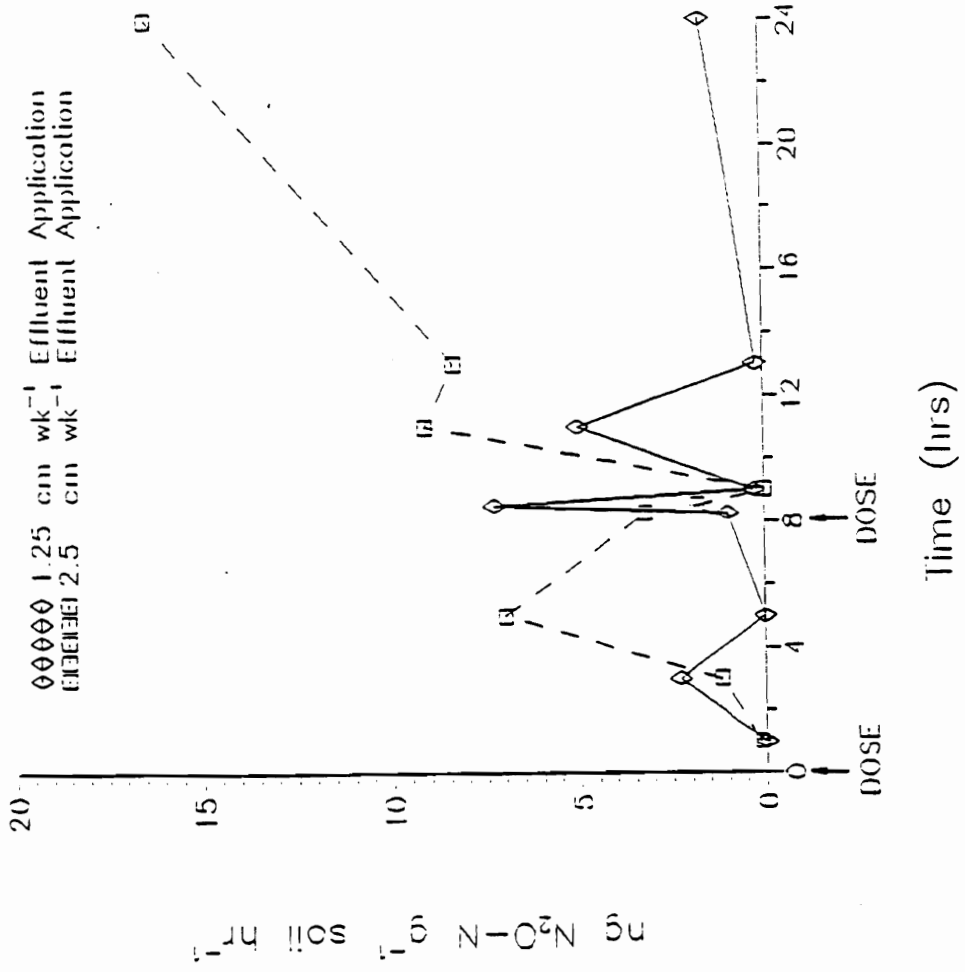


Figure 31. Nitrous oxide emission rates from the 2@ irrigation frequency application treatment at double the normal rainfall application (1.3 cm).

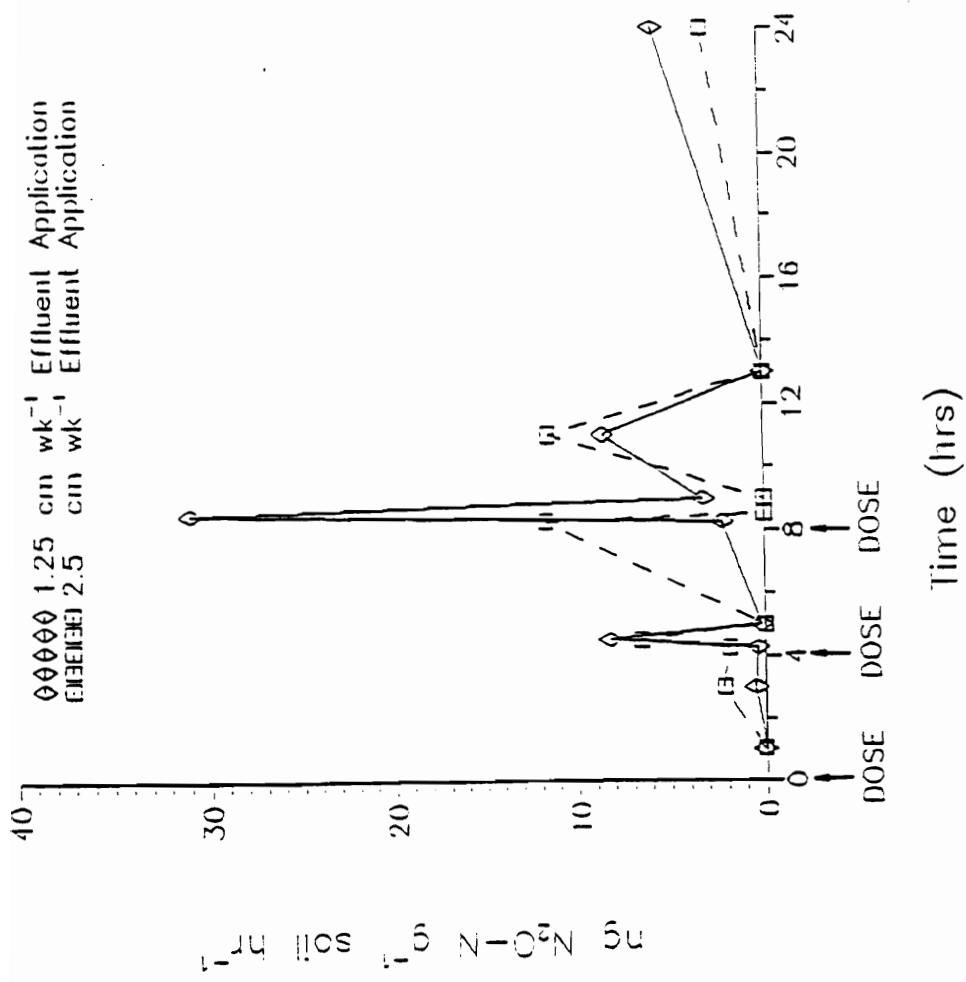


Figure 32. Nitrous oxide emission rates from the 3@ irrigation frequency application treatment at double the normal rainfall application (1.3 cm).

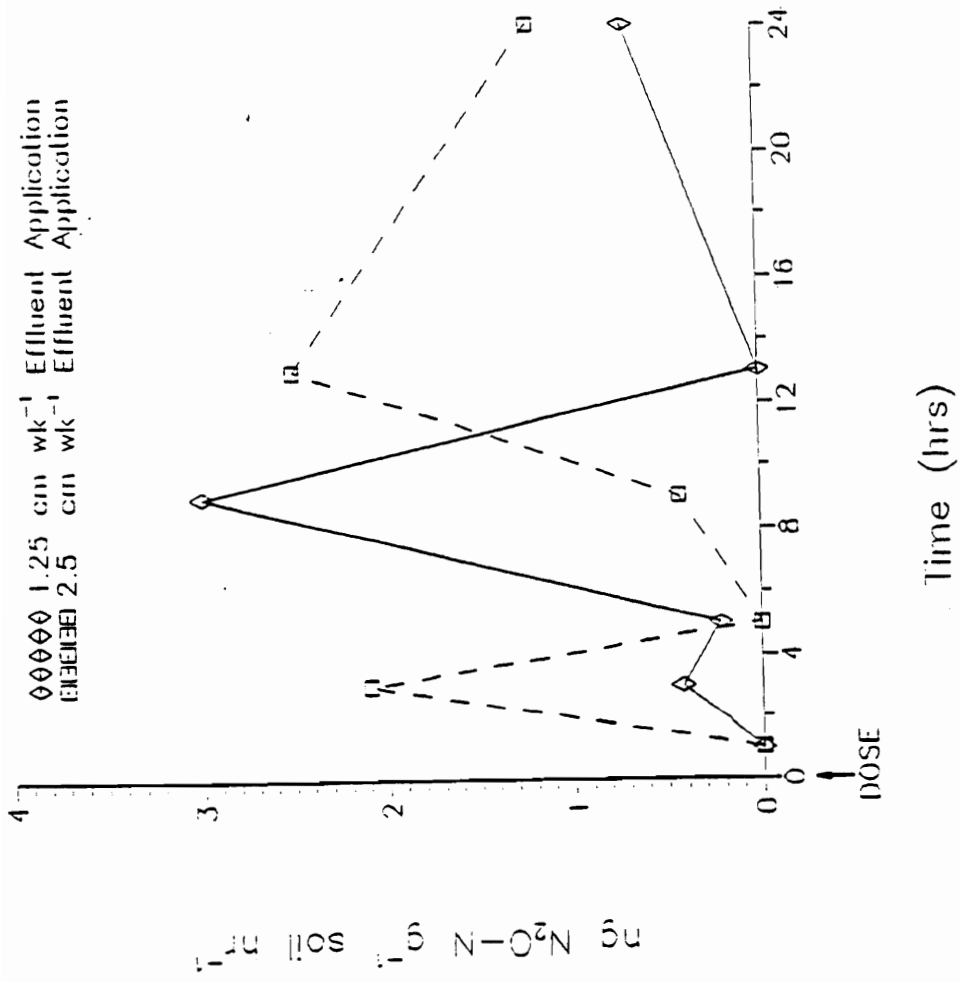


Figure 33. Nitrous oxide emission rates from the 1@ irrigation frequency application treatment at triple the normal rainfall application (1.95 cm).

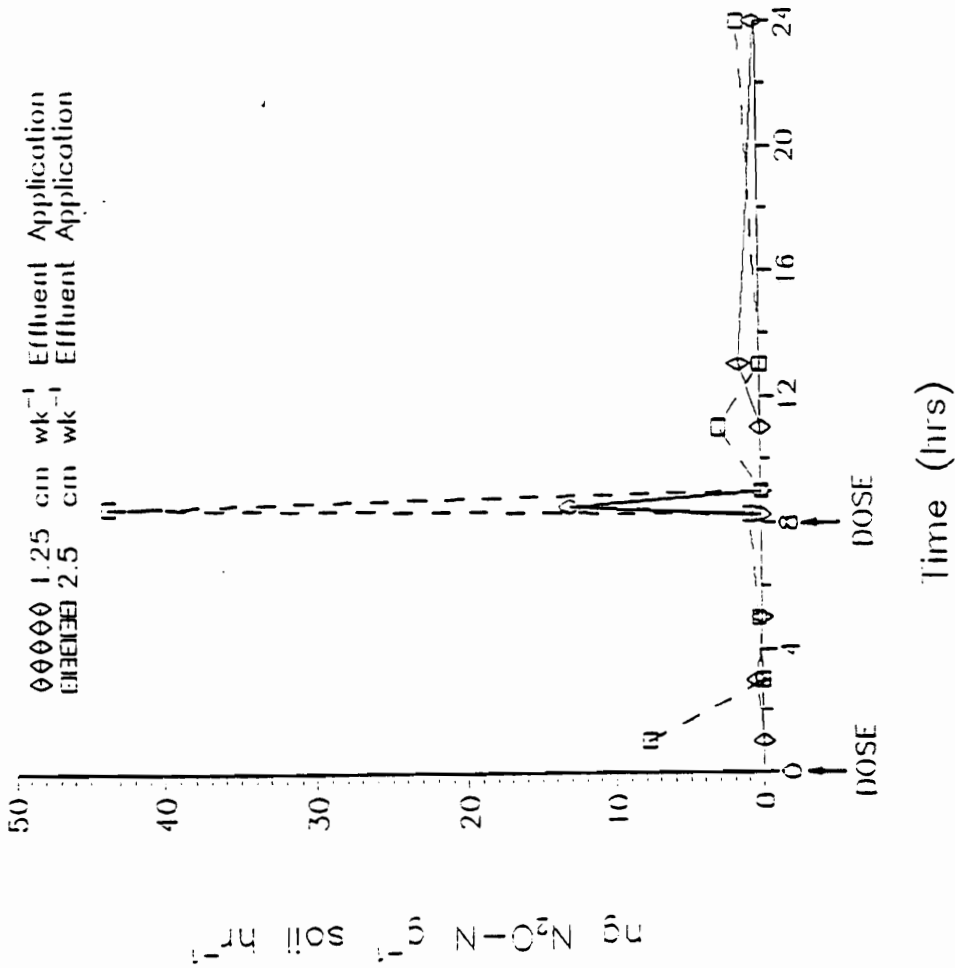


Figure 34. Nitrous oxide emission rates from the 2@ irrigation frequency application treatment at triple the normal rainfall application (1.95 cm).

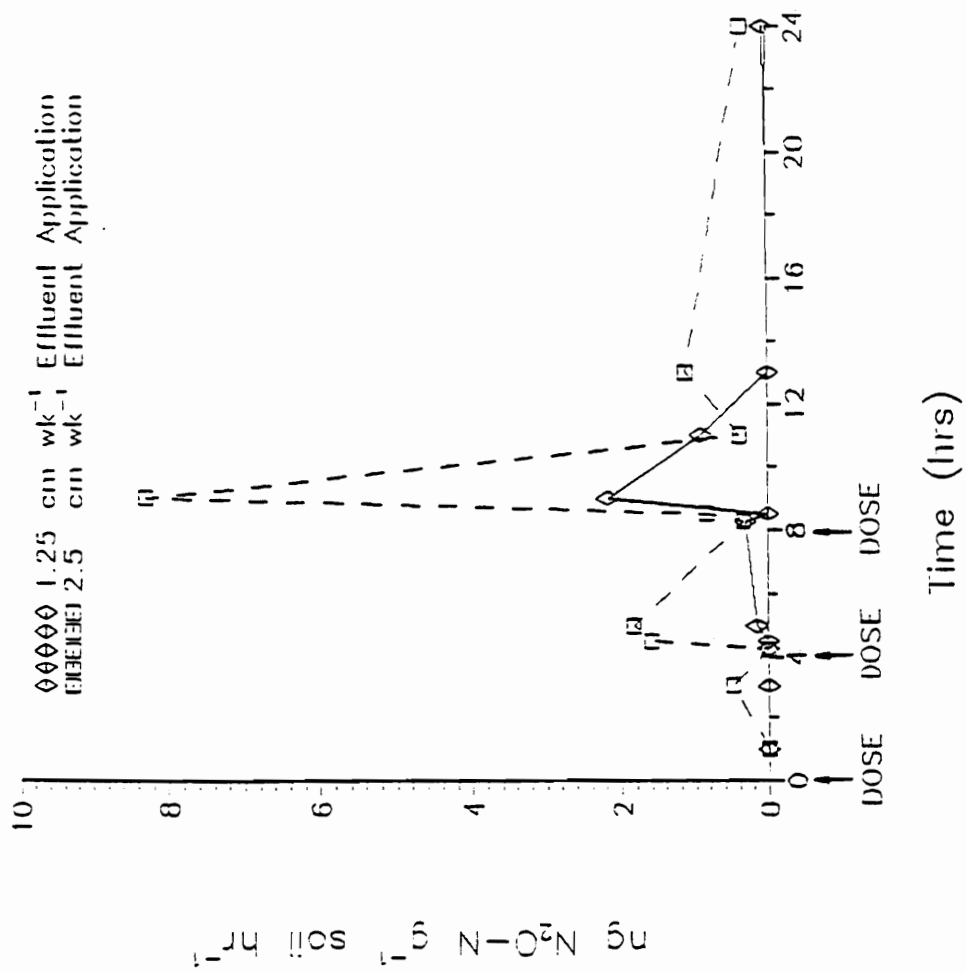


Figure 35. Nitrous oxide emission rates from the 3@ irrigation frequency application treatment at triple the normal rainfall application (1.95 cm).

## DENITRIFICATION CAPACITY

When comparing  $\text{N}_2\text{O-N}$  emission rates after 24 and 48 h (Table 41), it was evident that the soils denitrification capacity was both C and  $\text{NO}_3^-$ -N limited. Soils that received only C amendments had relatively low emission rates that were unchanged by incubation between 24 and 48 h (Table 41). Nitrous oxide emissions from the 0 C and applied  $\text{NO}_3^-$ -N were also low with no change between 24 and 48 h. The only large increases in  $\text{N}_2\text{O-N}$  emissions between 24 and 48 h were in treatments that received both C and  $\text{NO}_3^-$ -N amendments. The increases in  $\text{N}_2\text{O-N}$  emissions where both C and  $\text{NO}_3^-$ -N were applied were approximately 6-7 fold. Emission levels between the two C applied treatments with  $\text{NO}_3^-$ -N added were very similar in all the soils except at the 1@ irrigation frequency at both effluent application rates (Table 41). Emission rates remained relatively low at the 900  $\mu\text{g}$  C rate in the 1@ irrigation frequency soils. However with additional C, the  $\text{N}_2\text{O-N}$  emission increased greatly. It is difficult to determine why  $\text{N}_2\text{O-N}$  emission rates from the 1@ irrigation frequency treatments did not respond to the first addition of C (900  $\mu\text{g}$ ) considering that all of the other treatments including the control increased. It is unlikely that an adequate supply of C was already available in the soil since  $\text{N}_2\text{O-N}$  emission levels were much lower than emissions from the other treatments which received the first addition of C.

In the soils which received effluent applications in the irrigation frequency study, over 100% of the applied  $\text{NO}_3^-$ -N was recovered as  $\text{N}_2\text{O-N}$  at the highest C



and  $\text{NO}_3^-$ -N additions. The  $>100\%$  N recovery is most probably due to denitrification of residual N in the soil. In contrast, the control soils, which did not receive any previous effluent applications, had only 47% of the applied  $\text{NO}_3^-$ -N recovered as  $\text{N}_2\text{O}$ -N. These results demonstrate that the soils which previously received effluent applications developed a very efficient denitrifying microbial population.

When comparing the denitrification capacity among the irrigation frequency treatments, higher  $\text{N}_2\text{O}$ -N emissions were observed from the soils which received the 3@ irrigation frequency of application (Table 41). These results suggest that an application rate with more alternating wetting and drying cycles produced conditions that are more favorable to denitrification. The effects of irrigation frequency on  $\text{N}_2\text{O}$ -N emission were more pronounced in the denitrification capacity study than the column study. The soils in the capacity study were placed in constant anoxic conditions while the emission rates from the column study would be influenced by the duration of anoxic conditions caused by applied rainwater and effluent. The  $\text{N}_2\text{O}$ -N emission rates from the denitrification capacity study were at least 10 times higher than the rates measured from the irrigation frequency study. This observation demonstrates that soils under effluent irrigation have a high potential for N loss through denitrification.

## **<sup>15</sup>N TRACER STUDY**

The data collected for the <sup>15</sup>N tracer study did not agree with measurements from the irrigation frequency study. In the gas samples removed during the tracer study, concentrations of <sup>15</sup>N<sub>2</sub> were near natural abundance which indicates that none of the applied <sup>15</sup>N denitrified. This was not expected since data in the irrigation frequency study showed large quantities of N<sub>2</sub>O-N emission immediately after effluent applications. These data would indicate that there were some problems with storage or analysis of the <sup>15</sup>N<sub>2</sub> gas samples. The samples were shipped to a private laboratory, which then shipped the samples to an overseas laboratory as a result of equipment problems. It is possible that the long lag time between sample removal and analysis as well as the extra shipping might have caused sample loss.

There was also inconsistency with the <sup>15</sup>N soil measurements (Table 42). Soil samples were taken from 3 depth increments in the columns. Even though the total soil weight was measured for each column, the total weight for each depth increment was not directly measured. Since the recovery calculations of <sup>15</sup>N are based on the total weight of each depth increment, indirect measurements of the weight were utilized. The individual depth increment weights were calculated from bulk density measurements for the soil. The indirect soil weight measurements may have caused errors in calculating the percent recovery in the soil (Table 42). The recovery rates indicate that large amounts of <sup>15</sup>N remained in the soil.

Recovery rates in the soil ranged between 64 and 130%. The high variability in the recovery percentages was most probably caused by the indirect weight measurements.

The  $^{15}\text{N}$  measurements in the column leachate agreed more closely with data from the irrigation frequency data (Table 43). The percent recovery of  $^{15}\text{N}$  in the leachate collection ranged between 1.6 and 4.9% (Table 43). These data demonstrated that leachate from the columns contained low amounts of N, indicating that the majority of applied N was in the soil or denitrified. However, a portion of the  $^{15}\text{N}$  loss from the soil water might have been caused by dilution in the MIT as reported Karamonas and Rennie (1981) and Delwiche and Steyn (1979). Even though the  $^{15}\text{N}$  tracer study contained inconsistencies, it supported observations from the irrigation frequency study and the Montgomery Co. and Alleghany Co. field studies which showed small N losses in leachate water.

Table 40. Soil N content with depth after effluent applications in the column experiments.

Effluent Application	Irrigation Frequency <sup>†</sup>	Column Soil Depth					
		0-7		7-14		14-20	
cm wk <sup>-1</sup>		NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N
-----µg g <sup>-1</sup> soil-----							
0	-	5.7 a*	51.2 a	1.0 a	37.8 a	0.0 b	32.6 a
1.25	1	5.9 a	34.2 b	0.8 a	21.3 b	0.0 b	12.4 a
1.25	2	1.9 a	29.7 b	0.1 a	16.9 b	0.0 b	8.1 a
1.25	3	4.1 a	37.6 b	0.1 a	19.1 b	0.1 b	11.8 a
2.5	1	4.5 a	31.6 b	0.8 a	18.3 b	0.4 a	11.7 a
2.5	3	4.2 a	34.3 b	0.4 a	14.1 b	0.1 b	7.6 a

\* Means followed by the same letter in the same column are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.  
<sup>†</sup> applications day<sup>-1</sup>

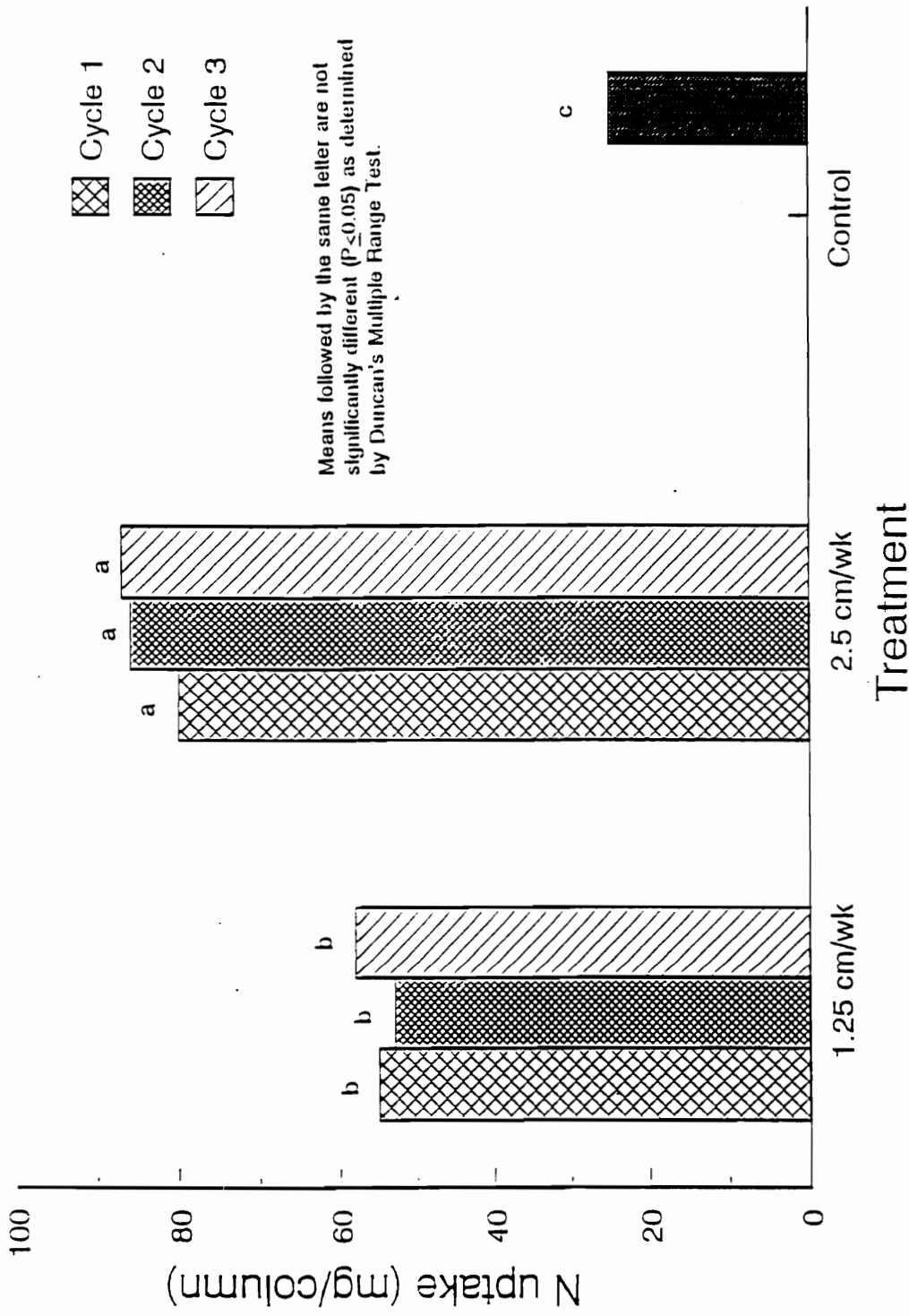


Figure 36. Nitrogen uptake by the fescue cover in the irrigation frequency study.

Table 41. Nitrous oxide emissions from column soils that received N and C amendments.

Amendments		Column Soil Treatments					
Carbon	Nitrogen †	3@ Frequency	2@ Frequency	1@ Frequency	Control		
		Effluent Loading Rate (cm wk <sup>-1</sup> )					
		1.25	2.5	1.25	1.25	2.5	0
----- µg g <sup>-1</sup> soil-----		-----µg N <sub>2</sub> O-N g <sup>-1</sup> soil-----					
		-----24 hours-----					
0	84	4.3 d*	3.9 b	5.7 b	3.6 cd	2.5 c	3.6 b
900	84	18.4 ab	19.7 a	14.9 a	9.4 b	4.4 bc	10.2 a
1800	84	22.1 a	20.5 a	16.2 a	15.6 a	19.6 a	9.5 a
0	0	1.7 d	1.7 b	2.8 b	0.2 d	3.1 c	1.4 b
900	0	15.6 bc	5.6 b	5.8 b	8.8 bc	9.5 b	8.6 a
1800	0	13.1 c	5.0 b	5.1 b	10.4 ab	6.6 bc	10.3 a
		-----48 hours-----					
0	84	6.2 b*	5.8 b	6.0 b	5.9 b	3.7 b	4.5 b
900	84	156.7 a	137.5 a	103.4 a	4.2 b	1.0 b	39.7 a
1800	84	140.5 a	133.5 a	93.8 a	114.9 a	130.9 a	39.5 a
0	0	2.3 b	1.6 b	1.0 b	0.2 b	2.9 b	1.1 b
900	0	13.3 b	4.2 b	4.2 b	7.1 b	8.1 b	6.9 b
1800	0	13.0 b	3.8 b	4.1 b	9.4 b	4.4 b	8.0 b

\* Means followed by the same letter in individual columns are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

† Carbon applied as D(+) Glucose.

‡ Nitrogen applied as NO<sub>3</sub><sup>-</sup>-N in KNO<sub>3</sub>.

○ Frequency indicates applications per day

Table 42. Soil <sup>15</sup>N levels and recovery rates in the tracer study.

Column	Depth	Soil weight	% N	%atom <sup>15</sup> N	Recovery	Total % Soil Recovery
	cm	g			-----%	
1	0-7	1019	0.62	0.58	87.3	
	7-14	1656	0.29	0.45	27.4	
	14-20	2420	0.18	0.42	15.8	130.5
2	0-7	1019	0.57	0.53	59.5	
	7-14	1656	0.30	0.40	10.4	
	14-26	2917	0.19	0.39	6.7	76.6
3	0-7	1019	0.59	0.58	80.4	
	7-14	1656	0.27	0.42	15.0	
	14-23	2101	0.16	0.40	7.0	102.4
4	0-7	1019	0.56	0.52	54.3	
	7-14	1656	0.25	0.40	8.0	
	14-22	1943	0.19	0.37	2.0	64.3

Table 43. Column leachate <sup>15</sup>N Recovery rates.

Column	DAY							Total
	1	2	3	4	5	6	7	
	-----% <sup>15</sup> N Recovery-----							
1	0.21	1.08	0.73	0.64	-	1.03	1.21	4.9
2	0.09	0.12	-	0.18	0.22	0.29	0.69	1.6
3	0.04	-	0.003	0.02	0.18	0.02	1.31	1.6
4	0.82	1.24	1.29	1.44	-	-	-	4.8



## DISCUSSION SUMMARY

In many respects the results from the column study agreed with observations from Montgomery and Alleghany Co. field studies. However, the more accurate  $N_2O-N$  emission measurements in the column study demonstrated much higher N losses from denitrification than observed in the field study. There were small differences between irrigation frequency treatments in  $N_2O-N$  accumulations from the columns, except in columns which were suspected to contain pockets of high denitrification activity. However, the irrigation frequency treatments exhibited different behaviors in the short term  $N_2O-N$  emission rates. Nitrous oxide emissions, where effluent was applied in a single application (1@ frequency), generally increased slowly over an extended period of time. While more frequent applications of lesser volumes of effluent, produced quick increases in  $N_2O-N$  emissions directly after effluent applications. The rapid emission rate increases lasted for shorter time periods when smaller volumes of effluent were applied. Denitrification emission rates behaved in spike-like functions due to changes in soil moisture. High rainfall applications combined with effluent applications appeared to reduce  $N_2O-N$  emission rates by enhancing  $NO_3^-N$  movement below the microbial active zone in the topsoil. The results suggest that there is a potential for optimizing N loss in spray irrigation by maintaining effluent in the microbial active topsoil through proper effluent application frequencies.

The plant N uptake, soil, and leachate data from the columns generally

agreed with the field data from Montgomery and Alleghany County sites. Plant N uptake was shown to increase in the columns with increased effluent application as in the field studies. There were no  $\text{NH}_4^+$ -N or  $\text{NO}_3^-$ -N accumulations in the soil due to effluent application. Similar results were also observed in the field studies during the growing seasons. For the majority of the leachate collections from the columns, the amount of N leached represented small percentages of what was applied in the effluent.

The denitrification capacity of the soils which received different irrigation frequency applications were all limited by both C and  $\text{NO}_3^-$ -N. Soils which previously received effluent applications were able to denitrify over 100% of the applied N during optimum conditions. The results from the denitrification capacity study suggested that soils which received more frequent effluent applications produced a more efficient denitrifying microbial population and thus higher potential loss of N.

The  $^{15}\text{N}$  tracer study agreed with the field and column studies in demonstrating low N amounts in leachate waters. However  $^{15}\text{N}_2$  gas samples collected from columns were inconsistent with data from the irrigation frequency study which was attributed to sample loss before analysis.

The column study supported results from the field studies which demonstrated that spray irrigation as an OSWTDS has a high potential for wastewater renovation on sites which contain soil limiting factors. The data indicated a high potential for N loss from these systems. However these losses,

especially from denitrification, are subject to seasonal fluctuations in environmental conditions, most importantly of which is soil moisture.

## CONCLUSIONS

1. The renovation capacity of surface applied effluent such as spray irrigation can be very efficient. The quality of leachate from the columns agreed with the field data which demonstrated high N removal rates. This result was also supported by a  $^{15}\text{N}$  tracer study.
2. Single daily doses of effluent in an irrigation system instead of split applications, produced extended periods of low  $\text{N}_2\text{O}$  emissions. However, high rates of  $\text{N}_2\text{O}$ -N emissions occurred directly (0.5 - 3 h) after effluent applications.
3. There were no differences between effluent application frequencies in the total  $\text{N}_2\text{O}$  accumulations except at the 2@ irrigation frequency at the  $2.5 \text{ cm wk}^{-1}$  rate of effluent application. Based on the column data, it is suggested that effluent applications via spray irrigation be applied in two applications 8 h apart for a greater potential of N loss.
4. Columns that received the 2@ irrigation frequency at the  $2.5 \text{ cm wk}^{-1}$  rate of effluent application were suspected to contain pockets of high denitrification activity which caused large amounts of  $\text{N}_2\text{O}$  production representing over 100% of the applied N.
5. Denitrification rates were reduced when triple the normal rainfall amount was applied to the columns which was attributed to movement of  $\text{NO}_3^-$ -N below the most active microbial zone.
6. Nitrogen loss from denitrification in the columns was calculated to be much

higher than losses in the field measurements. This was attributed to rate increases directly after effluent applications, which were not measured in the field. Losses in the column study average between 30 and 40% in the normal and double the normal rainfall conditions and 5% in the triple normal rainfall condition.

7. Different frequencies of effluent application did not change the amount of N taken up by the plant, however column plant tissue data agreed with field data showing that increased effluent loading rates increased plant N uptake.
8. Column data agreed with field data in that little residual  $\text{NO}_3^-$ -N remained in the soil under effluent irrigation.
9. The denitrification capacity of the column soils were limited by both C and  $\text{NO}_3^-$ -N. Prior effluent application enhanced the denitrification capacity of the soils, which, when under optimum conditions denitrified over 100% of the applied N.
10. An irrigation frequency with more alternating wetting and drying conditions (3@ frequency) increased the denitrification capacity of the soils microbial population.

## **PART III**

### **N BALANCE**

The N balance was determined using soil, plant, leachate, and N<sub>2</sub>O emission measurements from both the field and the denitrification column studies. Since the column study more accurately measured N<sub>2</sub>O emission rates, N<sub>2</sub>O values from the column study were utilized in conjunction with field measurements. The N balance was determined for June and November 1990 at the Montgomery Co. site to observe seasonal differences (Table 44). The N balance was also determined on an annual basis for both the Montgomery and Alleghany Co. sites (Table 45). Nitrous oxide emission measurements from the column study were used as estimates for denitrification loss in the annual N balances. The N balance was comprised of crop uptake, NH<sub>4</sub><sup>+</sup>-N and NO<sub>3</sub><sup>-</sup>-N in the upper 15 cm of the soil, leachate N and denitrification. All the N balance parameters were corrected by subtracting the control values from samples that received effluent. A missing component of the N balance was NH<sub>3</sub> volatilization. Volatilization measurements were not performed since the majority of N applied was expected to be present as NO<sub>3</sub><sup>-</sup>-N. However, at the Montgomery Co. site the largest N component was NH<sub>4</sub><sup>+</sup>-N. Ammonia loss probably occurred as effluent was applied. Ammonia losses from sprinklers has been reported by other researchers to be close to 10% (Henderson et al., 1961).

When the N balance for June at the Montgomery Co. site was considered,

the vast majority of N was present in the crop. Over 100 and 70% of the estimated N applied for the one month period is in crop uptake under the 1.25 and 2.5 cm wk<sup>-1</sup> rates, respectively. The high recovery in crop uptake was attributed to N accumulations in the soil profile during colder temperatures, which later mineralized and or nitrified and was available for crop uptake. The next highest sink for N was exchangeable soil NH<sub>4</sub><sup>+</sup>-N which accounted for 20 to 30% of the applied N in the effluent. This result was not unexpected at the Montgomery Co. site since over 90% of the applied N was in the organic or NH<sub>4</sub><sup>+</sup>-N form. The majority of NH<sub>4</sub><sup>+</sup>-N accumulations in the soil was probably held electrostatically by the 2:1 minerals as well as the sesquioxides present at the site. Accumulations of NH<sub>4</sub><sup>+</sup>-N in the soil during colder temperatures is supported by the monthly topsoil measurements (Figure 21).

Soil NO<sub>3</sub><sup>-</sup>-N accounted for a small percentage of the applied N, which was <1% for both loading rates during June. Low concentrations of NO<sub>3</sub><sup>-</sup>-N in the soil are attributed to rapid uptake by the growing grass cover. There was no evidence in leachate measurements of a NO<sub>3</sub><sup>-</sup>-N "pulse" after the accumulation of NH<sub>4</sub><sup>+</sup>-N and subsequent nitrification during warmer temperatures. Nitrogen loss via leaching was assumed to be nil in June since no net percolation should have occurred based on the hydraulic budget (Table 7). During this period, evapotranspiration was higher than the total amount of precipitation and irrigation, thus water loss below the root zone would be low. This was supported by observations in the field which showed that the sampling wells at the 60 cm depth

were dry.

During the winter the hydraulic characteristics of the system changed. In the winter, plant growth is minimal and evapotranspiration is greatly reduced (Table 7). The plant uptake component was not included in the N balance for November since plant growth was minimal. The accumulation of  $\text{NH}_4^+$ -N in the soil profile was rapid during the non-growing season compared to the growing season (Figure 21). However accumulations were not yet present in the soil samples collected in November. The amount of soil  $\text{NH}_4^+$ -N at the high effluent application rate was not included in the N balance for November since  $\text{NH}_4^+$ -N levels were lower than control values. Soil  $\text{NO}_3^-$ -N accounted for a small percentage of the applied N during November. These  $\text{NO}_3^-$ -N levels were attributed to the small amount of  $\text{NO}_3^-$ -N applied in the effluent and cooler temperatures which would limit nitrification.

There was percolation of water due to rainfall in November according to the hydraulic budget which was supported by field observations that noted soil saturation at the 60 cm depth. Nitrate concentrations in leachate water at the low effluent application rate in November averaged  $0.63 \text{ mg L}^{-1}$ . Since there was a high volume of leachate water due to low evapotranspiration, the estimated amount of N leached represented 12.4% of the applied N for that period. Nitrate concentrations in leachate at the  $2.5 \text{ cm wk}^{-1}$  rate of effluent application during November were low ( $0.17 \text{ mg L}^{-1}$ ), which accounts for the lower estimated N loss in leachate at that rate.



Field denitrification measurements accounted for a higher percentage of N recovery in June than in December. Recovery rates in June were approximately 5% of the effluent N applied to the soil-plant system (Table 44) while recovery rates in December were less than 2%. Denitrification measurements however, were collected several hours after irrigation and would not have included emission rate increases directly after irrigation.

The N balance at both sites between September 1989 and September 1990 is presented in Table 45. A larger amount of N was applied at the Alleghany Co. site due to the higher concentration of total N in the effluent. The total amount of N applied was adjusted to available N by estimating 30% of the organic N fraction was mineralized (U.S. EPA, 1981). Total recovery rates of the N applied in 1990 for the Montgomery Co. site were 93 and 90% at the low and high irrigation rates, respectively. Total recovery rates for the year in the Alleghany Co. site were 82 and 75% for the 1.25 and 2.5 cm wk<sup>-1</sup> rates, respectively. When crop uptake over the one year period was considered, N recovery in the fescue was approximately 50% in the Montgomery Co. site and 35% in the Alleghany Co. site. Differences in N recovery in the fescue at in both sites between loading rates were small, which suggests that the capacity for N removal by the fescue was not limited by the loading rates used.

The NO<sub>3</sub><sup>-</sup>-N and NH<sub>4</sub><sup>+</sup>-N concentrations in the soil profile at both sites were determined in November 1990. Since seasonal fluctuations of NH<sub>4</sub><sup>+</sup>-N were observed in the topsoil, the values used for the annual balance are probably not

indicative of  $\text{NH}_4^+$ -N accumulation in the soil for the year. The  $\text{NH}_4^+$ -N levels measured in November indicate that the accumulation observed in mid December (Figure 21) was not yet evident. However the amount of  $\text{NH}_4^+$ -N in the soil measured in November represent less than 10% of the N applied for the year in both sites. The N recovery in soil  $\text{NO}_3^-$ -N was less than 4% in both sites. The soil  $\text{NH}_4^+$ -N and  $\text{NO}_3^-$ -N data from both sites demonstrated that net storage of N in the soil was small.

The values for leachate N presented in the annual balance were corrected for dilution. The amount of dilution from the effluent was based on  $\text{Cl}^-$  data. The values were also based on accumulated losses. The accumulated losses of N in leachate was based on the volume of leachate from irrigation and rainfall at each site (Appendix Table A-4) for each month. Rainfall amounts were corrected for evapotranspiration using estimates from Tables 6 and 7. The loss of N in the Montgomery Co leachate was 10.2 and 4.6% in the 1.25 and 2.5  $\text{cm wk}^{-1}$  irrigation rates, respectively. Even though the recharge volume of water at the 2.5  $\text{cm wk}^{-1}$  irrigation rate was larger than the 1.25  $\text{cm wk}^{-1}$  rate of effluent application the N loss in leachate was less at the 2.5  $\text{cm wk}^{-1}$  rate of application. This was caused by the higher concentrations of N observed under the 1.25  $\text{cm wk}^{-1}$  rate compared to the 2.5  $\text{cm wk}^{-1}$  irrigation rate. The percent recovery in the Alleghany Co. leachate was 2.0 and 7.3% for the low and high rates, respectively. However the values presented may not reflect "pulse" of N from the profile that may have occurred during a large rainfall event.

The means of N<sub>2</sub>O emissions from the normal and double the normal rainfall conditions were used for estimating N loss from denitrification in the annual N balances. Values from the 1@ frequency application treatment were utilized since the application method had the most resemblance to the field effluent applications. The average N loss from denitrification for both rainfall conditions were 33 and 27% for the 1@ irrigation frequency at the 1.25 and 2.5 cm wk<sup>-1</sup> effluent applications, respectively. The N<sub>2</sub>O emission measurements from the column study were collected in a greenhouse environment where the temperature was approximately 24° C. Denitrification emissions in the field are most certainly lower when temperatures are cooler. Since denitrification estimates were extrapolated from 24 h measurements and since denitrification rates are highly influenced by environmental conditions, the yearly estimates most probably fall within a wide range.

Table 44. Nitrogen balances for Montgomery Co. site in June and November 1990. All values have been normalized by subtracting the values of the control plots.

	June 1990			November 1990		
	Irrigation Rate (cm wk <sup>-1</sup> )			Irrigation Rate (cm wk <sup>-1</sup> )		
	1.25	2.5		1.25	2.5	
	mg N m <sup>-2</sup>	%	mg N m <sup>-2</sup>	mg N m <sup>-2</sup>	%	mg N m <sup>-2</sup>
N applied†	1100		2200	1100		2200
wastewater						
N recovered						
crop	1282	116	1555	0	0	0
NH <sub>4</sub> <sup>+</sup> -N in soil‡	275	25	626	292	26	*
NO <sub>3</sub> <sup>-</sup> -N in soil‡	10	0.9	12	45	4.0	*
Denitrification	59§	5.4	112§	5§	0.4	44§
Leachate	¶		¶	137	12.4	106
Total recovered	1626	147	2305	474	42.4	150
Unaccounted	-	-	-	626	57.6	2050
						93.3

\* Value less than control value.

† N includes organic N fraction.

‡ Soil measurements include total in the upper 15 cm of the profile.

¶ no water leached according to hydraulic budget.

§ Based on field N<sub>2</sub>O emissions.

Table 45. Nitrogen balances for both field sites in a one year period (Sep 89 - Sep 90). All values have been normalized by subtracting the values of the control plots.

	Montgomery Co. site				Allegheny Co. site			
	Irrigation Rate (cm wk <sup>-1</sup> )		Irrigation Rate (cm wk <sup>-1</sup> )		Irrigation Rate (cm wk <sup>-1</sup> )		Irrigation Rate (cm wk <sup>-1</sup> )	
	1.25	2.5	1.25	2.5	1.25	2.5	1.25	2.5
N applied†	mg N m <sup>-2</sup>	%	mg N m <sup>-2</sup>	%	mg N m <sup>-2</sup>	%	mg N m <sup>-2</sup>	%
wastewater	7000		14000		17400		34800	
N recovered								
crop	3500	50	7486	53	6656	38	12151	35
NH <sub>4</sub> <sup>+</sup> -N in soil	*		351	2.5	1638	9.4	1501	4.3
NO <sub>3</sub> <sup>-</sup> -N in soil	20	0.3	507	3.6	*		702	2.0
NH <sub>4</sub> <sup>+</sup> -N in leachate‡	232	3.3	321	2.3	157	0.9	2095	6.0
NO <sub>3</sub> <sup>-</sup> -N in leachate‡	481	6.9	320	2.3	208	1.1	439	1.3
Denitrification		33\$		27\$		33\$		27\$
Total recovered		93		90		82		75
Unaccounted		7		10		18		25

† Available N includes 30% mineralization rate of the organic N fraction.

‡ Soil measurements include total in the upper 60 cm of the profile.

§ Leachate measurements determined from concentrations at the 60 cm depth with dilution and volume of water determined from precipitation + irrigation - evapotranspiration.

\* Value less than control value.

§ Based on average N<sub>2</sub>O emissions from the column study.

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## **APPENDIX**



Table A-1. Sandfilter effluent quality with sampling time from the Alleghany Co. site .

DATE	pH	E.C	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	PO <sub>4</sub> -P	CL <sup>-</sup>
		S m <sup>-1</sup>	-----mg L <sup>-1</sup> -----			
Feb 90	6.1	-	0.52	9.10	0.7	11.9
May 90	-	-	18.0	16.1	-	-
Jun 90	7.1	-	31.0	12.6	-	-
Sep 90	-	-	80.0	35.7	-	-
Nov 90	7.6	0.085	33.6	10.0	4.80	-
Jan 91	7.5	0.057	6.5	9.2	2.6	14.1

Table A-2. Electric conductivity of subsurface water from both field sites.

Site	Depth --cm--	Effluent Loading Rate (cm wk <sup>-1</sup> )		
		0	1.25	2.5
			-----S m <sup>-1</sup> -----	
Montgomery Co.	60	0.0111 b*	0.0145 b	0.0265 a
	150	0.0224 a	0.0232 a	0.0230 a
Alleghany Co.	60	0.0100 b	0.0132 b	0.0192 a
	150	0.0057 b	0.0058 b	0.0081 a

\* Means followed by the same letter for each depth and site that are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

Table A-3. Runoff water quality means for each collection date at the Montgomery Co. site.

Date	P†	Trt‡	Vol¶	pH	E.C	NO <sub>3</sub> <sup>-</sup> -N	NH <sub>4</sub> <sup>+</sup> -N	PO <sub>4</sub> -P	Cl <sup>-</sup>
	cm	cm/wk	L		S m <sup>-1</sup>	-----mg L <sup>-1</sup> -----			
8/19/89	4.6	0	1.6	8.0	-	0.27	0.41	0.13	0.5
		1.25	1.2	7.6	-	0.64	0.57	0.05	0
		2.5	1.0	7.8	-	0	0.15	0	-
9/12/89	3.1	0	3.0	8.2	-	2.60	0.24	0.37	1.1
		1.25	2.0	8.1	-	1.79	2.92	0.32	0
		2.5	1.0	8.2	-	3.18	0.11	0.36	0
9/18/89	3.6	0	>50	5.7	.0085	0.64	0.16	0.03	-
		1.25	>50	5.6	.0094	0.79	0.16	0.02	-
		2.5	>50	6.0	.0094	0.53	0.12	0.02	-
10/19/89	3.4	0	42	5.8	.0139	8.71	0.51	0.06	8.2
		1.25	>50	5.6	.0157	9.64	0.26	0.06	12.8
		2.5	>50	5.7	.0164	0.30	0.24	0.09	11.8
11/26/89	5■	0	20	6.0	.0127	0.66	0.27	0.05	13.1
		1.25	28	6.1	.0146	0.48	0.20	0.08	12.7
		2.5	>50	6.2	.0270	0.19	0.22	0.07	22.4
2/4/90	2.2	0	>50	6.0	.0106	0.18	0.09	0.02	0.9
		1.25	>50	6.4	.0195	0.17	0.09	0.03	2.8
		2.5	>50	6.6	.0257	0.29	0.11	0.04	9.3
8/7/90	5.8	0	2.6	7.4	.0179	0.44	1.16	0.10	-
		1.25	3.0	7.3	.0175	0.21	0.62	0.09	-
		2.5	2.0	7.4	.0229	0.47	2.25	0.09	-
9/22/90	0.9	0	1.3	7.3	.0242	3.11	0.20	0.32	3.5
		1.25	1.6	7.0	.0201	10.47	0.14	0.95	0
		2.5	1.0	7.3	.0343	8.88	0.36	0.60	4.9
10/19/90	3.5	0	-	7.3	.0088	0.28	0.49	0.21	13.4
		1.25	-	7.3	.0099	0.28	0.41	0.28	19.3
		2.5	-	7.5	.0153	0.17	0.35	0.38	30.6
2/7/91	1.4	0	-	7.5	.0124	1.31	0	0.08	12.5
		1.25	-	7.4	.0106	1.49	0.22	0.08	10.1
		2.5	-	7.3	.0346	1.12	0.37	0.21	23.4

† P indicates amount of precipitation causing runoff event.

‡ Trt indicates effluent loading rate.

¶ Vol indicated the volume of runoff collected.

■ Snow fall.

Table A-4. Precipitation amounts for each site during experiment.

		Montgomery Co. site†	Alleghany Co. site‡
		-----cm-----	
Jun	1989	17.9	
Jul		12.2	
Aug		11.4	
Sep		20.5	
Oct		8.1	
Nov		8.6	
Dec		0.7	
Jan	1990	8.3	7.5
Feb		9.4	7.4
Mar		8.1	6.0
Apr		6.0	8.1
May		18.4	12.3
Jun		2.3	5.5
Jul		10.2	13.5
Aug		10.4	7.9
Sep		4.6	1.9
Oct		18.0	15.1
Nov		3.4	4.1
Dec		9.3	7.5
Jan	1991	7.5	
Feb		5.5	
Mar		14.9	

† Measurements collected at VPI&SU Horticultural Farm Rt. 460

‡ Measurements collected at Hot Springs VA.

Table A-5. Nitrate-N and  $\text{NH}_4^+$ -N in soil column leachate for each collection date.

		Effluent Loading Rate					
		cm wk <sup>-1</sup>					
		1.25	2.5				
		Application Cycles					
		applications day <sup>-1</sup>					
Date	Control	1	2	3			
		mg L <sup>-1</sup>					
Mean SE‡		NO <sub>3</sub> <sup>-</sup> -N					
Mar 90	7.4±5.3	0.8±0.2	9.7±2.9	0.3±0.1	2.5±0.9	1.6±1.4	0.6±0.3
Apr 90	3.4±2.6	0.3±0.0	2.7±1.2	0.1±0.0	2.8±0.7	1.6±1.3	0.05±0.03
May 90	0.2±0.06	0.06±0.03	0.2±0.1	0.1±0.0	0.3±0.1	0.2±0.1	0.2±0.05
Nov 90†	2.6±0.8	4.4±2.0	¶	2.1±0.9	5.1±0.2	2.9±2.5	3.0±0.9
		NH <sub>4</sub> <sup>+</sup> -N					
Mar 90	0.2±0.1	0.04±0.01	0.2±0.08	0.1±0.04	0.1±0.02	0.1±0.01	0.20±0.05
Apr 90	0.05±0.05	0.01±0.01	0.14±0.11	0.1±0.1	0.04±0.04	0.08±0.05	0.3±0.03
May 90	0.3±0.1	0.3±0.14	0.4±0.3	0.3±0.14	0.2±0.03	0.3±0.19	0.3±0.1
Nov 90†	0.4±0.07	0.5±0.4	¶	0.5±0.06	1.0±0.2	0.8±0.3	0.7±0.2

§ Applications per day of applied effluent.

† Collected after a 3x application of the normal rainfall application.

‡ Standard error.

¶ No leachate collected.

Table A-6 Soil Cl<sup>-</sup> concentrations with depth at both field sites.

Depth	Montgomery Co.				Alleghany Co.			
	0	1.25	2.5	2.5	0	1.25	2.5	2.5
cm	-----cm wk <sup>-1</sup> -----							
	-----μg g <sup>-1</sup> -----							
0-15	31.7	139.0	88.7	50.7	67.6	44.8		
15-30	17.7	86.0	76.8	39.4	36.4	32.7		
30-45	17.5	98.5	87.0	47.6	84.9	30.7		
45-60	17.4	66.7	66.5	44.8	101.2	41.4		

Table A-7. Soil carbon levels with soil depth and effluent applications at both field sites.

Alleghany Co. site‡			
Effluent Loading	Soil Depth		
	0-15	15-30	30-45
0	1.46 b*	0.60 a	0.00 a
1.25	1.61 ab	0.66 a	0.00 a
2.5	1.62 a	0.75 a	0.00 a
Montgomery Co. site¶			
0	2.48 a	0.64 a	0.22 ab
1.25	2.51 a	0.63 a	0.04 b
2.5	1.89 b	0.46 a	0.38 a

\* Means followed by the same letter for each depth at individual sites are not significantly different ( $P \leq 0.05$ ) as determined by Duncan's Multiple Range Test.

‡ Samples collected after 12 months of effluent application (analysis by dry combustion).

¶ Samples collected after 18 months of effluent application (analysis by dry combustion).

## VITA

Gregory Thomas Monnett, son of Gregory Flagg and Marcella Regan Monnett, was born in Pittsburgh, Pennsylvania on July 3, 1962. He was raised to age 13 in Bloomfield, New Jersey, after which the family moved to Knowlton, New Jersey. In Knowlton he developed his interest in agriculture which lead to his B.S degree in Agronomy from Delaware Valley College of Science and Agriculture in 1984. In 1987 he finished his M.S. Degree in Agronomy from Clemson University, where he specialized in Soil Fertility. Mr. Monnett received his Ph.D. in Crop and Soil Environmental Science from Virginia Polytechnic Institute and State University in June 1992. Greg married Margaret Matt on August 31, 1991 in Richmond, Virginia.