

THE EFFECT OF DISTANCE BETWEEN ARTIFICIAL  
DRAINAGE FACILITIES AND DISPOSAL TRENCHES  
ON THE MOVEMENT OF BIOLOGICAL AND  
CHEMICAL POLLUTANTS FROM SEPTIC  
TANK EFFLUENT,

by

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(ABSTRACT)

A field study was conducted at a residence in Chesapeake, Virginia to determine the effect of setback distances from a drainage ditch on the disposal of septic tank effluent. The study was done from September 1979 to August 1981. The soil used is the Tomotley series belonging to the fine-loamy, mixed, thermic family of Typic Ochraquults. Four prototype trenches were installed at 1.5, 3, 6 and 21 m from the edge of a drainage ditch which was 1.5 m deep. The trenches were pressure dosed equally with 2.4 to 4 cm per day. Replicated nests of sampling wells at depths 90, 150 and 300 cm were placed with distance from the trenches. Continuous stage recorders were positioned 4.5, 29.1 and 60 m from the ditch to monitor water table behavior. Groundwater analysis included fecal coliforms, the  $\text{NH}_4$ ,  $\text{NO}_3$ ,  $\text{NO}_2$ , Na, Ca, Mg, Cl and  $\text{PO}_4$  ions, pH and EC.

Ammonium concentrations in the soil beneath the trenches indicated nonuniform effluent infiltration yet more uniform than with conventional gravity flow distribution.

High sodium absorption ratios of the septic tank effluent ranging from 18 to 45 did not significantly reduce infiltration rates as no

ponding of effluent in the trenches was observed.

The existing land surface was sloped greater than the water table gradient resulting in decreasing unsaturated depths with closeness to the ditch. The mean distances between the trench bottom and the water table were 64 cm at 3 m, 80.1 at 6 m and 90.4 cm at 21 m.

The accumulation of fecal coliforms,  $\text{NH}_4$ , P, Cl and total salts (EC) in the groundwaters at 120 to 150 cm depth was inversely proportional to the mean distances from the trench bottoms to the water table. The lack of denitrification occurring underneath the trench with the most extensive unsaturated zone suggests  $\text{NO}_3$  may accumulate under disposal systems that have mean unsaturated depths of  $\geq 90$  cm.

Effluent movement was mainly lateral and in the upper zone of the water table with limited vertical movement below the water table surface.

An equation applying D-F theory to infinitely deep soil was developed to describe flow for the given drainage system and for the inclusion of standard size drainfields.

The current practical setback distance of 21 m is considered counterproductive for effluent treatment for some situations.

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## INTRODUCTION

Human body and household wastes pose potential health hazards unless safely disposed. Waterborne disease outbreaks of salmonellosis, giardiasis, shigellosis and gastroenteritis caused by enteric bacteria and traced to fecal contamination of water supplies were cited by Cabelli, 1978. Infectious viral diseases, especially hepatitis, are spread by way of sewage polluted drinking and recreational waters (Metcalf, 1978, and Britton, 1978). Home sewage, consisting of feces and urine and bath, laundry and kitchen wastewaters, is discharged to either public or private disposal systems. Public sewage works are designed to be self contained and meet discharge standards set by governmental agencies and require full-time trained operators. Private systems, while designed for safe disposal based on current state of the art, are not as controlled as public systems and are more subject to environmental interaction. Probably the majority of homeowners (operators) are ignorant of the performance characteristics and maintenance requirements of their systems. The number of individual on-site sewage disposal systems in this category is significant. For example, Virginia has 27.5 percent (408,213 homes ) of its housing units served by on-site septic tank systems (U. S. Census, 1970).

The large number of individual sewage disposal systems currently in operation and the anticipated increase in demand for private sewage disposal systems in the future pose potential problems for maintaining the quality of ground and surface waters. Waters used for consumption and recreation and those projected for future use must be protected.

Studies and surveys have pointed to on-site disposal systems as contributing to eutrophication of surface waters and contamination of both surface and shallow groundwaters (Edmondston, 1968; Schmidt, 1972; Childs, et al., 1974; Quan, et al., 1974; Miller, 1975; Katz, et al., 1980; Porter, 1980; Carlile, et al., 1981). A review article by Gerba, et al. (1975), revealed 2-27 percent of groundwaters surveyed and used for drinking water supplies contained fecal indicator organisms.

The extent of public sewerage is limited by local economics and policies and usually serves only highly populated areas. Sprawling urban and rural communities rely on individual on-site wastewater disposal systems. A large number of housing areas and developments in the Coastal Plain region of Virginia have low living-unit density and consequently are not served by public sewerage. These homes, as an alternative, use on-site sewage disposal systems. Portions of these land areas are characterized by soils with high seasonal fluctuating water tables. These soils are also poorly to very poorly drained but otherwise have moderate to rapid permeability. These sites are not normally suited for conventional septic tank-soil absorption systems (ST-SAS) because of the hydraulic and effluent treatment problems associated with seasonal high water tables. Effluent is more efficiently renovated in the soil with time and distance under aerobic unsaturated flow conditions versus anaerobic saturated flow conditions which occurs when water table levels are near the soil-gravel interfaces of drainfields. Also additional hydraulic loading of an already satu-

rated soil can result in surface seepage of septic tank effluent (STE), increasing the potential for a public health hazard.

In order to overcome the limitations of restrictive drainage and high water tables and meet the increasing demand for additional housing, the Virginia State Health Department has allowed local governing bodies to provide artificial drainage to lower seasonally high water tables for housing area developments. The drainage facilities are usually surface trenches along property boundaries, streets and roads leading to a continuous outfall. Following construction of drainage ways and assurance of continued maintenance of the artificial facilities, individual on-site sewage disposal systems are permitted. Currently, as standard practice, the distance between the disposal field and the drainage facility ranges from 18-24 m. The objectives of this study are listed below.

1. Determine the shape of the water table with distance from a drainage ditch.
2. Evaluate the performance of soil absorption trenches and the effect of the water table levels on their operation.
3. Determine and evaluate the movement of bacteria and chemical constituents of septic tank effluent from identical absorption trenches placed varying distances from a drainage ditch.
4. Develop an equation to describe the observed slope of the water table and to account for the addition of a disposal drainfield within the drainage system.

## REVIEW OF LITERATURE

### Introduction

Past studies conducted on ST-SAS have considered their effect on the surrounding environment and visa versa. In the same tradition, this review will discuss the processes and factors that affect STE movement from the subsurface absorption trench into the adjacent natural soil and movement and attenuation of biological and chemical components away from the SAT. Significant individual STE components will be discussed separately.

### Effluent Infiltration

A conventional SAT that receives effluent from the septic tank for disposal by soil absorption is 1) partially filled with gravel to maintain a soil-gravel interface over the entire bottom area and part of the sidewall area and 2) contains distribution piping enclosed in the gravel (USPHS, 1967). The first phase of safe effluent disposal is infiltration of STE through this soil-gravel interface. Distribution of STE along the trenches may determine success or failure of a ST-SAS.

Converse (1974) reported STE distribution by gravity flow via 10-cm perforated piping resulted in effluent exiting the perforation at the lowest elevation. Nonuniform STE distribution creates localized overloading which can result in system failure either by effluent surfacing or gross contamination of shallow groundwaters (Otis, et al., 1974a).

A biological mat or crust resistant to flow just under the soil-gravel interface has been observed in field installations and laboratory studies. Development of this mat was attributed to suspended solids

and exudates of microbial organisms as a result of excessive STE loading rates (Winneberger, et al., 1960; Bouma, et al., 1972; Jones and Taylor, 1965; Mitchell and Nevo, 1964). Anaerobic conditions at a portion of the soil-gravel interface, inherent in trickle, gravity flow, perpetuate and accelerate the biological matting process (Avnimelech and Nevo, 1964) that eventually seals or partially seals the entire soil-gravel interface and may result in surfacing of STE. Kristiansen (1981) observed that surface clogging of a fine sand advanced at a rate of  $.33 \text{ m}^2/\text{month}$  when loaded at a rate of 4-6 cm/day by gravity flow in a laboratory system designed to simulate field trenches. He further indicated higher temperatures more than increased loading rates facilitated clogging.

Introduction of STE into the SAT by pressurized perforated pipes provides greater uniformity of effluent distribution than gravity flow systems and lessens localized overloading (Converse, 1974). Pressurized systems also afford dosing and rest periods which allows reaeration of the soil-gravel interface and subsequent oxidation of a developing biological crust thereby increasing the longevity of the system (Bouma, et al., 1974). Hargett, et al. (1982), while testing STE loading strategies, observed ponding occurred first with simulated gravity trickle flow (8 doses/day) before single doses, independent of loading rates. They further concluded clogging at the soil-gravel interface increased with increasing loading rates.

In the event a significant biological crust develops, and effluent is continuously ponded in the SAT, steady state unsaturated flow results

and uniform soil moisture tension (SMT) will develop below the crust and results in a unit hydraulic gradient (Hillel and Gardner, 1969; Baver, et al., 1972). For this condition the resulting SMT is dependent on the crust resistance and pore distribution of the soil (Bouma, et al., 1974). A subsequent rise in the water table to levels sufficiently close to the trench bottom (dependent on the prevailing constant SMT) would lower the SMT and reduce the equilibrium hydraulic gradient necessary to overcome the crust resistance and slow down or cease STE infiltration (Bouma, 1975).

Water tables in shallow unconfined aquifers or perched water tables have been observed to fluctuate widely and rise rapidly in response to significant rainfall during the winter and spring (Palkovis, et al., 1975; Bianchi and Haskell, 1966; Hallock, 1978; Stewart and Reneau, 1981b). Drainage ditches with outfalls depress the water table level near the ditch and exert less influence with distance away from the ditch. The water table surface would therefore have greater slope as the ditch is approached. The shape of a water table drawdown curve caused by parallel drains and local precipitation can be calculated by exact analysis given uniform hydraulic conductivities ( $K$ ) of soil layers and the depth to an impermeable barrier (Kirkham and Powers, 1972). For infinitely deep soils, Van Demeter (in Childs, 1969) developed an equation for the maximum rise of the water table for rainfall on an infinitely deep soil with uniform  $K$  and parallel drains.

Dupuit-Forchheimer (D-F) analysis has been shown to give good approximation of water flow analysis, in spite of paradoxical assumptions

for shallow uniform soils overlying an impermeable barrier (Childs, 1969; Freeze and Cherry, 1979). Childs (1969) suggested using D-F theory also for infinitely deep soil when the drainage system involved orthogonal sets of drains. The reasoning involves mathematically transforming the rectangular drains into equivalent parallel drains which can be done for both D-F and Van De Meter equations because neither relationship depends on a depth to a barrier regardless of the dimensions of the rectangular configuration.

The sodium absorption ratio (SAR) of the effluent can also influence STE infiltration because soil structure will be altered and the hydraulic conductivity reduced. The deleterious effects of a high exchangeable sodium percentage (ESP) have long been recognized (Scofield and Headly, 1921). An exchange constant for each soil determines the relationship between ESP and SAR. The U. S. Salinity Laboratory Staff (1954) states an SAR of 15 can adversely alter soil structure. Other work in the areas of leaching soils with varying NaCl solutions (McNeal and Coleman, 1966; McNeal, 1968; Lagerwerff, et al., 1969) have shown the combination of low salt concentrations and high SAR's caused soil swelling and reduction of hydraulic conductivity while high salt-low SAR caused no reduction. The presence of montmorillonitic clay minerals is closely linked to soil swelling and dispersion (McNeal, et al., 1966) although iron oxide coatings precipitated on montmorillonite will substantially reduce swelling (Rowell, 1965, and El Rayeh and Rowell, 1973).



Gardner, et al. (1959) experimenting with the Yolo loam and Pachappa sandy loam soils observed a one thousand fold decrease in weighted-mean water diffusivity when the ESP was near 25 and salt concentration was decreased from 300 to 3 meq/l.

Corey, et al. (1977) determined theoretically, using McNeal's (1968) empirical model for interlayer swelling, that the use of water softeners in conjunction with ST-SAS have no necessarily irreversible damaging effects. They indicated the natural soil at the soil-gravel interface would not be permanently altered provided the NaCl recharging backwash was discharged to the septic tank to increase the salt concentration and replenish the exchanged Ca and Mg. They also suggested soil swelling may occur upon introduction of foreign dilution water (i.e. rainfall, interflow and runoff waters) into the drainfield area.

#### Effluent Movement in Soils

Once effluent has infiltrated past the soil-gravel interface, its movement and renovation varies with the soil moisture status and pore geometry. The significance of unsaturated flow in purification of STE was emphasized by Otis (1976). In summarizing the extensive studies conducted under the auspices of the Small Scale Waste Management Project at the University of Wisconsin-Madison, he reiterated the enhancement of soils treatment capacity via filtration, biochemical reactions, and absorption during slow effluent movement under unsaturated flow conditions. Effluent moves at a much slower rate during unsaturated flow in smaller pores (Bouma, et al., 1974) affording greater intensity of the above processes. On the other hand, STE movement through soil

macropores is rapid in comparison and results in reduced treatment with distance. Movement of effluent bacteria is considered to be shortened by a factor of 5 to 10 under unsaturated flow compared to movement when all the soil pores are saturated (Romero, 1970).

Macropore flow occurs in two situations pertinent to STE disposal and treatment. One above and the other below the water table. In well-structured soils with large void spaces between aggregates rapidly applied effluent can pond in the trench gravel and ultimately flow rapidly downward through macropores for the length of pore continuity. Rapid movement, greater than molecular dispersion and/or average solution velocity, is termed hydrodynamic dispersion and results in less efficient treatment and increased travel distances for STE components (Anderson and Bouma, 1977).

Hydrodynamic dispersion takes place in macropores in either water saturated or unsaturated soil. More rapid short-range movement is likely to occur when effluent is dosed onto a drained soil than the same soil completely saturated. Ziebell, et al. (1975) observed greater bacteria movement through drained soil columns containing strong-structured soils when dosed with STE than when the columns were initially saturated, then dosed and allowed to drain. Thomas and Phillips (1979) attested, in general, to the importance of macropore flow and specifically to increased salt movement.

Below the water table surface, soil solution flow is primarily through the largest soil pores as they are paths of least resistance. The movement of STE after reaching a completely saturated zone depends

on the prevailing hydraulic gradient of the diluted STE soil solution. Reneau (1978a and 1979) attributed distant movement of both biological and chemical components of STE to saturated macropore flow influenced by a strong hydraulic gradient towards tile drains and additionally to uneven distribution of STE. The absence of a discernible water table gradient resulted in downward STE movement below the water table in a field study by Stewart and Reneau (1981b) whereas a mean water table gradient of 2.8 percent resulted in mostly lateral STE flow and limited vertical movement in a similar soil. Harkin, et al. (1979) reported limited STE movement in perched water tables below mound systems because of the static nature of the temporary perched water.

#### Bacteria Movement

Bacteria have limited active mobility and move in mass with the soil solution. The removal of pathogens in soils has been considered a function of many soil parameters such as temperature, pH, soil absorption, biological soil clogging, soil moisture, nutrient availability and biological competition and antagonism (Gerba, et al., 1975). Another key factor as pointed out by Otis (1976) is the soil solution flow regime. Unsaturated flow retards bacteria movement by physically filtering and absorbing the organism in the smaller pores and by affording more time and more aerobic conditions for pathogen dieoff as a result of biological competition and antagonism. Saturated flow which uses the large macropores, results in less opportunity for soil adsorption and filtration and is generally faster and provides a more suitable environment for enteric bacteria.

In a series of articles in the late 1930's, Caldwell (1937, 1938a, b) and Caldwell and Parr (1937), studied the movement of bacteria from latrines placed in and above groundwater in soils of varying depths. They reported the rate and direction of groundwater flow as the most influential factors in bacteria movement and that bacteria movement varied in direction as the water table depth changed but with little bacteria movement normal to the flow path. The migration of B. coli from a pit latrine below the water table were initially found at distances up to 23 m but within 7 months the B. coli were present only at the edge of the latrine. This drastic reduction was attributed to what Caldwell called "soil defense" which apparently corresponds to the modern term "biological clogging". In a dry latrine above a water table, B. coli were limited to 30 cm penetration into a sandy soil. Subsequent daily additions of 400 liters of water caused B. coli to move only 2 m downward and 30 cm laterally.

Romero (1970), reviewed previous research and concluded that for an ideal system, biological pollutants should travel no more than 15-30 m with the groundwater. He concluded that biological organism penetration in unsaturated conditions would reach approximately 3 m.

Laboratory studies with 55-cm undisturbed soil columns by Ziebell, et al. (1975) showed that fecal coliform bacteria moved rapidly down planar voids and channels in a well structured soil when the columns were drained prior to dosing. The bacteria were completely attenuated when the columns were saturated before dosing.

In a field study Hagedorn, et al. (1978) used freshly dug pits,

below a two percent ground surface gradient, to inject antibiotic resistant E. coli at 15- and 45-cm depths. They observed initial rapid (1 day) movement of fecal indicator organisms to 50-cm deep radially placed wells at a 50-cm distance when the water table was initially at a 45-cm depth below the ground surface. The authors attributed this rapid movement to diffusion. Gradual peaks were observed in wells at distances of 1500 cm zone 2-3 days after rainfall when the bacteria was injected into a coarser textured, soil horizon at 15-cm depth.

At the 45-cm injection site in the finer textured and weaker structured horizon, the peaks were lower and had a 8-day lag period following rainfall. In a subsequent study by Rahe, et al. (1978), E. coli were introduced into the A and B horizons of two hillside soils with artificially maintained perched water tables. Coliform movement was rapid laterally (15 m in 1 hour) for all injection depths on a 14 percent slope site (Dixonville series) and slower and more attenuated at a 10 percent slope site (Hazelair series). Both soils had the same texture in the A horizons. Coliform movement was severely restricted in the dense clay B horizon of the Hazelair series. The authors attributed the rapid coliform movement on saturated macropore flow. A third study (McCoy and Hagedorn, 1980) using three distinguishable strains of E. coli injected into the A, B and C horizons, upslope from a soil transitional zone (a footslope), observed translocation patterns that were more consistent with injection depths near the introduction points but converged downslope at the transition zone. Large volumes of water with  $10^5$  E. coli per 100 ml were observed to surface from a downslope rodent burrow. The

authors concluded flow was converging from diffuse flow regions near the injection points into mainly macropore or pipe flow as water moved downslope.

Studies conducted in Virginia coastal plain soils with restrictive Plinthic and Fragic horizons by Reneau and Pettry (1975) showed virtually no fecal coliform penetration into the restrictive horizons and moderate to large reductions in coliforms detected in perched and continuous water tables with horizontal distance from the drainfield. Reneau (1978 b) working in wet tile-drained soils reported fecal coliforms moved laterally in a predictable manner as influenced by distribution of STE throughout drainfield, distance from the drainfield and the water table gradient.

Harkin, et al. (1979) monitored the performance of sand-filled mounds in Wisconsin and observed high total coliform concentrations in saturated soil at abrupt textural boundaries and in the surface zone of the water tables. They attributed this distribution to the movement of bacteria with percolating water. Subsequent downward movement of coliforms was restricted if unsaturated conditions existed or continued when the lower zone was saturated.

Stewart and Reneau (1981a) working with conventional gravity flow systems in poorly drained coastal plain soils (Typic Ochraquults) observed maximum movement of fecal coliforms when the water table rose to the vicinity of the drainfield trench bottoms. Coliform movement was lateral over a relatively impermeable B horizon and in the presence of a water table gradient (albeit only averaging 2.8 percent) and was

vertical to a measured depth of 3 m in absence of a discernible water table gradient.

Carlile, et al. (1981) monitored 16 sites including both conventional and alternative systems in the North Carolina coastal plain region. They observed the greatest lateral coliform movement in systems continually operating under saturated conditions. Limited improvement in bacteria attenuation was observed by the use of pressure distribution systems. In comparing all sites, they reported median fecal coliform most probable number (MPN)/100 ml in groundwaters were 6 times as great in a clayey vs. sandy soils at 1.5 m horizontal distance and twice as great at 30 meters. This is probably another example of the importance of macropore flow in strongly structured soils.

The development of a crusted or clogged layer at the soil-gravel interface is effective in removing bacteria from the flow path (Tyler, et al., 1977). Typical soil conditions surrounding older ST-SAS in Wisconsin where crusting had formed at the soil-gravel interface were effective in eliminating fecal coliform bacteria past a 60-cm distance from the SAT (Ziebel, et al., 1974).

### Nitrogen Movement

The potential toxicity of nitrate ( $\text{NO}_3$ ) and nitrite ( $\text{NO}_2$ ) ions and nutrient enrichment are causes for concern with respect to nitrogen (N) sources and movement. The U. S. Public Health Service and the State of Virginia have set the drinking water limits for  $\text{NO}_3$ -N and  $\text{NO}_2$ -N at 10 and 1 ppm, respectively. Although N is not usually the limiting factor in surface water eutrophication, it can sustain the algae and

bacteria blooms once increased growth has commenced (Odum, 1971).

Nitrate contamination of groundwater has been reported to result from subsurface disposal of domestic sewage. In East Portland, Oregon, 30280-37850 cubic meters/day (8-10 MGD) is introduced into soil from individual on-site systems in a 78 sq. km (30 sq. mile) unsewered area. Disposal of this STE has resulted in  $\text{NO}_3\text{-N}$  levels of 5-12 ppm in shallow groundwaters which eventually reached a surface drain while  $\text{NO}_3\text{-N}$  levels in deeper aquifers and upgradient shallow groundwaters were <1 ppm (Quam et al., 1974). Nitrate concentrations of several hundred ppm in localized areas at the surface of a 21-m (70-feet) deep water table in a California study was attributed to septic tank use (Schmidt, 1972).

An investigation conducted in the Delaware Coastal Plain (Miller, 1975) contrasted two study areas where on-site disposal systems and home water wells were located on the same site. One characterized by soils with varying permeability (Elkton soil series, 25 percent of the area; Fallsington, 30 percent; Matapeake, 5 percent and Woodstown, 40 percent) and normally high seasonal water table levels with a population density of 77.2 people  $\text{km}^{-2}$  (200 people/sq. mile). The other areas was composed of the Sassafras series with a water table at 4.5-7.5 m (15-25 feet) and a density of 1029 people  $\text{km}^{-2}$  (2666 people/sq. mile). Samples collected from 37 shallow dug wells (3.9-10.5 m deep) in the first area showed a  $\text{NO}_3\text{-N}$  range of 0.01 to 11.3 ppm. The second area had deeper wells (10.5-21 m) that ranged from 5 to 30 ppm  $\text{NO}_3\text{-N}$  in 49 samples and averaged 13.3 ppm. The Delaware study indicated  $\text{NO}_3$  accumulation in groundwaters in well drained soils used for STE disposal.

A study by Walker, et al. (1973b) in Wisconsin showed essentially



complete nitrification of STE-N was occurring and little or no denitrification occurred upon reaching the groundwater in deep sandy soils. They also estimated that 0.2 ha (1/2 acre) was needed down gradient for dilution of  $\text{NO}_3\text{-N}$  below the 10 ppm level.

Nitrogen present in soils in relatively small amounts (Brady, 1974) is a dynamic element and varies in availability through process of soil organic matter mineralization and immobilization, soil absorption, nitrification and denitrification (Alexander, 1961). Sikora and Corey (1975) in a review article stated the phenomena most pertinent to sewage disposal are soil absorption of ammonium ( $\text{NH}_4$ ) and nitrification ( $\text{NH}_4 \rightarrow \text{NO}_2 \rightarrow \text{NO}_3$ ) and denitrification ( $\text{NO}_3 \rightarrow \text{N}_2$  gas). The kinetics of biological nitrogen transformations favor the less toxic  $\text{NO}_3$  as the conversion from  $\text{NO}_2$  to  $\text{NO}_3$  is faster than the transformation of  $\text{NH}_4$  to  $\text{NO}_2$  (Brady, 1974).

Several factors influence nitrification, among which are  $\text{NH}_4$  availability, pH, amount of oxygen ( $\text{O}_2$ ) present, temperature, soil moisture and population of nitrifying bacteria. The foremost of these are the availability of  $\text{NH}_4$  and sufficient oxygen ( $\text{O}_2$ ) as the nitrifiers are mainly obligate autotrophic aerobic organisms. Denitrifiers, on the other hand, are heterotrophic facultive bacteria requiring the absence of free  $\text{O}_2$  and an energy source (Alexander, 1961). Sikora and Corey (1975) stated pH and organic matter can affect N transformations but are secondary in importance to the aeration status of soil systems. The effect of soil moisture on nitrification/denitrification is important as soil moisture directly affects the movement of air in the soil.

Optimum moisture tension range for nitrification in a silt loam soil is reported by Parker and Larson (1963) to be from 40-50 to 275 cm water. Pilot and Patrick (1972), working with soils of varying textures discovered a critical tension for each soil between 20-40 cm of water corresponding to 11-14 percent air-filled porosity above which denitrification would not occur and that a very small amount of  $O_2$  prevented denitrification. Nitrification can take place at low moisture content (high moisture tension) but at a slower rate. Justice and Smith (1962), recovered as  $NO_3$ , 100 percent of  $NH_4$  added after 21 days in a silt loam soil at seven bars tension and recovered 60 percent at the wilting point, 15 bars tension. Stewart, et al. (1979) working with nonperforated soil columns observed conversion of total N in STE to  $NO_3$  to be 83-100 percent complete in a loamy sand at 90-cm depths.

Carlile, et al. (1981), investigated 16 on-site sewage disposal sites in a wide range of coastal plain soils including both conventional and alternative systems. They observed at the 1.5- m horizontal distances greater mean groundwater concentrations of  $NO_3$ -N (3.3 to .9 ppm) in seasonally saturated vs. continually saturated, conventional gravity flow system vs. pressure distribution systems and sandy soils vs. clay soils. Mean  $NH_4$ -N levels (4.3 to 3.1 ppm) were higher for the continually saturated systems, pressure systems and sandy soils.

Russell (1961) suggested nitrogen loss due to denitrification during the rise and fall of the water table. Patrick and Wyatt (1964) showed  $NO_3$  loss by denitrification during the wetting and drying cycles of rice paddy operations. Denitrification decreased with increased

cycling and was restored to the original rates upon additions of rice straw as a carbon energy source.

Harkin, et al. (1979) evaluating Wisconsin mound performance observed 44 percent of the N was nitrified in the sand fill (30-60 cm deep) and subsequently denitrified in the saturated zone at the interface with the underlying natural soil.

Reneau (1977) reported evidence of nitrification/denitrification cycles related to fluctuations in water tables. In another study, Reneau (1978b) observed the highest levels of  $\text{NO}_3$  in groundwaters occurred as the water table was rising and subsequently decreased with time.

#### Phosphorus Movement

The importance of phosphorus (P) as a pollutant is realized for its role in eutrophication of surface waters. As a limiting nutrient, P can trigger algae blooms and jeopardize the stability of fresh water ecosystems. Odum (1971) attests the sensitivity of even large ecosystems to P. Washington Lake, Seattle, was enriched with P (30-40 ppb) via sewage effluent resulting in oxygen depletion and fish dieoff and yet recovered when P levels were suppressed (Edmondston, 1968). Agronomists have long been concerned with P since it is a major plant nutrient but in short supply in soil solutions. The limited availability of P is attributed to soil fixation involving absorption on clay minerals and precipitation. The capacity of soils to fix P causes concern in the agriculture community but is regarded as an asset from an environment point of view. Yuan, et al. (1960) and Hsu (1965), reported iron and

aluminum to be the key agents in precipitation and fixation of added P in acid soils. Sikora and Corey (1975) concluded in a review study that only in soils with little sorption capacity would the pollution potential of P from STE be considerable.

Using packed laboratory columns under saturated conditions, Sawhney (1977) discovered P breakthrough after reaching the Langmuir-determined P sorption capacity with a leaching solution of STE-P levels and concluded ST-SAS placement in high water tables may result in undesirably high levels of P in groundwaters. Other research by Adriano, et al. (1975) and Sawhney and Hill (1975) show P to be absorbed in excess of the Langmuir absorption maxima. Further, Sawhney and Hill (1975) suggested P absorption sites to be regenerated by alternate wetting and drying. Reneau (1979) in a field study in Virginia Coastal Plain soils was able to predict horizontal movement of P from STE as a function of distance from the drainfield in the direction of a distinct water table gradient and emphasized the increased sorption of P in unsaturated flow and the more extensive P movement under saturated conditions.

Harkin, et al. (1979) suggested organic coatings on soil or sand fill surfaces prevent P absorption and/or lack of adequate reaction time to account for P movement in natural soils at 55-cm depth where orthophosphates averaged 0.99 ppm.

Stewart, et al. (1979) observed soil solution P levels at the 95-cm depth of loamy sand textures to be 2.2-2.8 ppm whereas a mixture of topsoil and loamy sands reduced the P concentration to .6 to .8 ppm range, indicating increased P absorption for finer textures.

Large amounts of active iron and aluminum and acid soil were cited by Carlile, et al. (1981) as factors restricting movement of P in a variety of soils used for disposal of STE. Phosphorus concentrations in ground waters  $>1$  mg/l were not detected past 3-m lateral distances.

## MATERIALS AND METHODS

An experimental area representing typical soil and site conditions of housing area development using artificial drainage was located in the Coastal Plain physiographic province at a private residence in Chesapeake, Virginia. The study area has flat topography and is 4.5 m above sea level. The soil, the Tomotley series, belongs to the fine-loamy, mixed, thermic family of Typic Ochraquults and is classified as poorly drained based on the presence of low chroma soil colors within 38 cm of the surface. A description of the Tomotley soil is located in the Appendix. Selected physical and chemical soil properties are listed in Table 1. The saturated hydraulic conductivities in Table 1 was determined by the constant head method (Klute, 1965). Bulk densities for the horizons of the Tomotley soil were determined by the core method (Blake, 1965). The total pore space was determined by the "difference method" where the totally saturated soil core is oven dried and 1 g water is assumed equal to 1 cm<sup>3</sup> (Vomocil, 1965). The pore size distribution was determined by the equation for capillary rise (Vomocil, 1965) using volume of water released at given soil moisture tensions determined by the pressure plate method (Richards, 1965). The cation-exchange capacity was determined by the ammonium saturation procedure (Chapman, 1965). Organic matter was done by the Walkley-Black method (Allison, 1965).

The 1.22-hectare home site had a perimeter drainage ditch ranging in depth from 1 to 2 m that emptied into a large canal, the Portsmouth Ditch, which is a major drainage way for the Dismal Swamp. A portion

Table 1. Selected physical and chemical properties for the Tomotley series.

Soil Property	Horizon						
	A1	B1	B21tg	B22tg	C1	C2	C3g
Depth (cm)	0-18	18-38	38-73	73-103	103-128	128-183	183-208+
Sand (%)	61.1	64.2	52.7	65.8	-	95.0	91.5
Silt (%)	30.9	21.7	26.3	12.2	-	0.0	2.5
Clay (%)	8.0	14.1	21.0	22.0	-	5.0	6.5
Hydraulic Conductivity (cm/day)	182	132	98.4	76.8	300	-	-
(GPD/sq. ft.)	45	33	24	19	74	-	-
Bulk Density (g/cc)	1.11	1.70	1.71	1.77	1.67	-	-
Pore Space (%)	76.2	64.1	47.3	49.0	54.3	-	-
Pore Size Distribution (microns)							
>108 (%)	11.1	27.6	12.3	15.3	15.4	-	-
108-52 (%)	5.1	4.4	7.8	5.0	7.9	-	-
52-27 (%)	12.4	12.3	10.7	7.6	22.6	-	-
27-4 (%)	9.9	14.3	17.0	15.8	18.4	-	-
<4 (%)	61.5	41.4	51.3	56.3	35.6	-	-
Organic Matter (%)	8.0	2.3	0.4	0.3	0.2	0.1	0.4
CEC (meq/100 g)	23.1	10.6	10.0	9.6	2.0	3.0	14.8

of the drainage ditch 1.3-m deep and paralleling the road serving the housing area was chosen as a representative drainage facility.

Four prototype-SAT were installed at distances of 1.5, 3, 6 and 21 m from the edge of the drainage ditch (See Figure 1). A profile of the study area is given in Figure 2. The construction details and dimensions of the prototype-SAT are shown in Figure 3.

The SAT were excavated with a trenching machine equipped with 20-cm tines. The initial trenches were longer than six m to allow for depth adjustment and backfilled and tamped with original soil to obtain the design 6-m length. Pea gravel (6 mm in diameter) was placed directly on the freshly dug trench bottom to a depth of 15 cm. The perforated PVC pipe was positioned on top of the gravel and covered with the original soil. The 3 mm holes of the PVC pipe were directed downward. The trenches were mounded slightly to prevent surface water entry and seeded with 'KY 31' fescue (Festuca arundinaceae). Surface swells and shallow drainways were constructed at Systems 1 and 2 some 11 months after installation to divert surface waters.

Replicated nests of groundwater sampling wells were placed at depths of 90, 150, and 240 cm and equal distances from each mini-SAT provided sufficient available space between the prototype trench and the edge of the drainage ditch (See Figure 4). The sampling depths were selected to facilitate determination of vertical STE movements by providing distinct sampling zones (ie, prevention of overlapping sampling zones). The distances of the sampling wells from each absorption trench were selected to facilitate statistical comparisons. Four of the systems



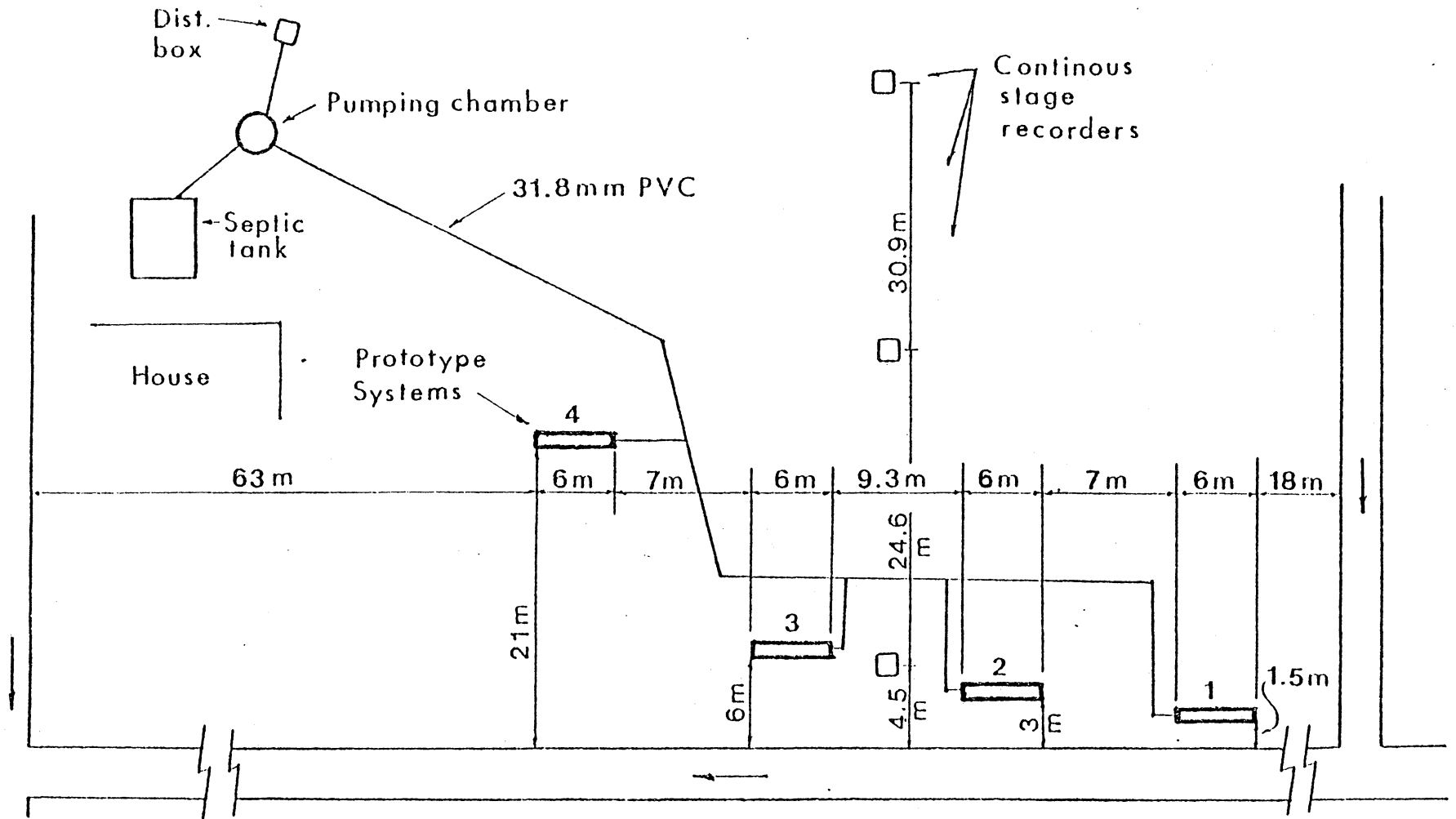


Figure 1. Site plan view of experimental area and relative locations of prototype trenches, water level recorders and drainage ditches.

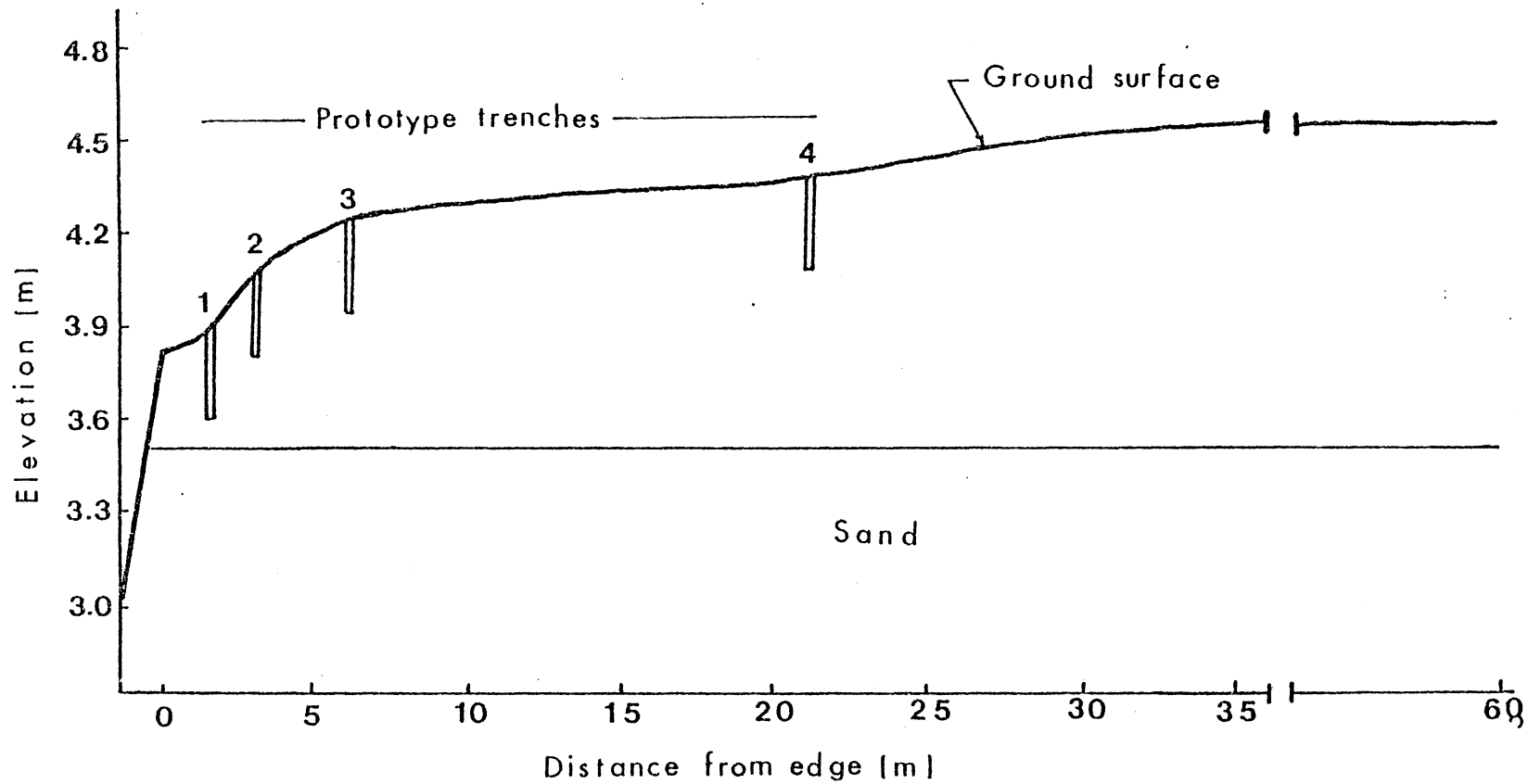


Figure 2. Profile of study area including the ground surface, the sand substrata and prototype trenches.

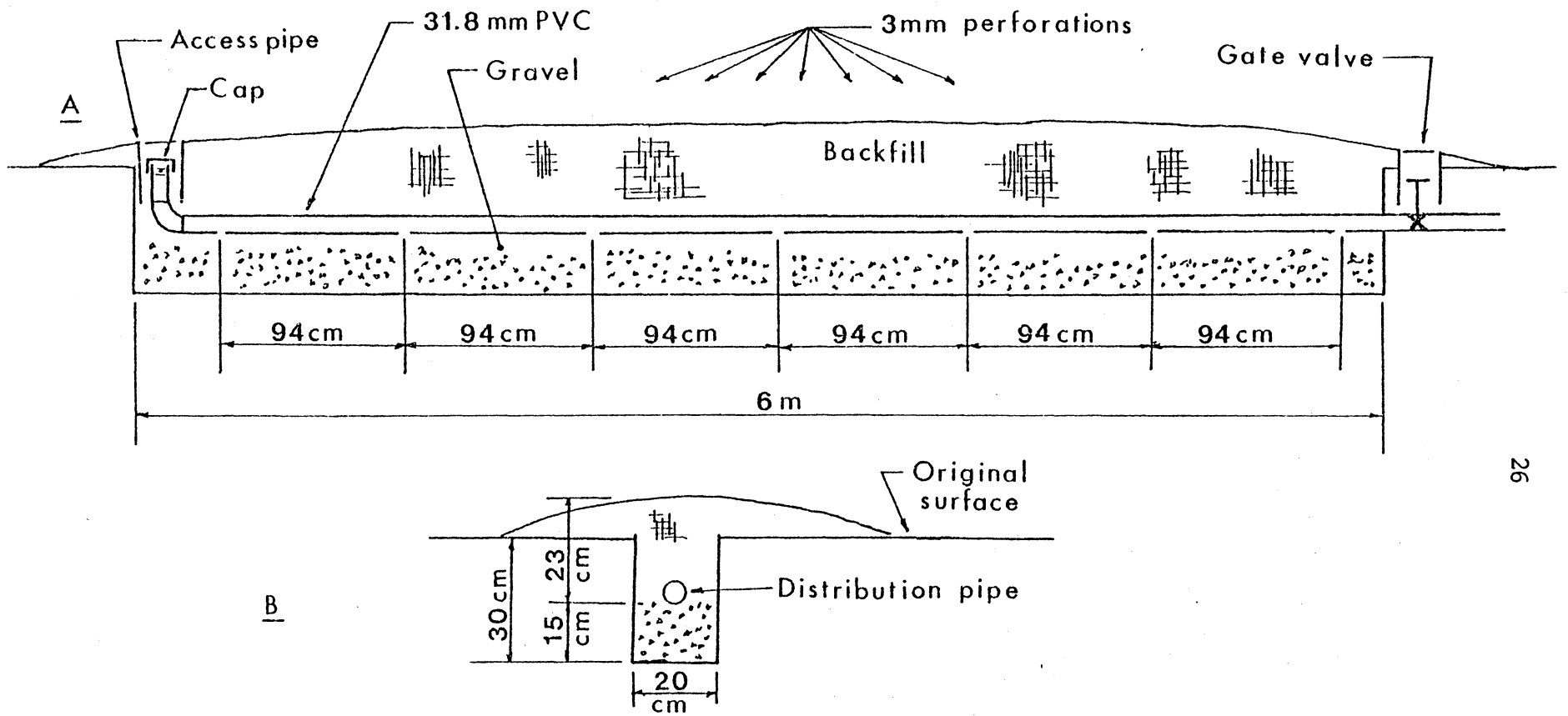


Figure 3. Cross sections of prototype trenches, A, longitudinal and B, transversal.

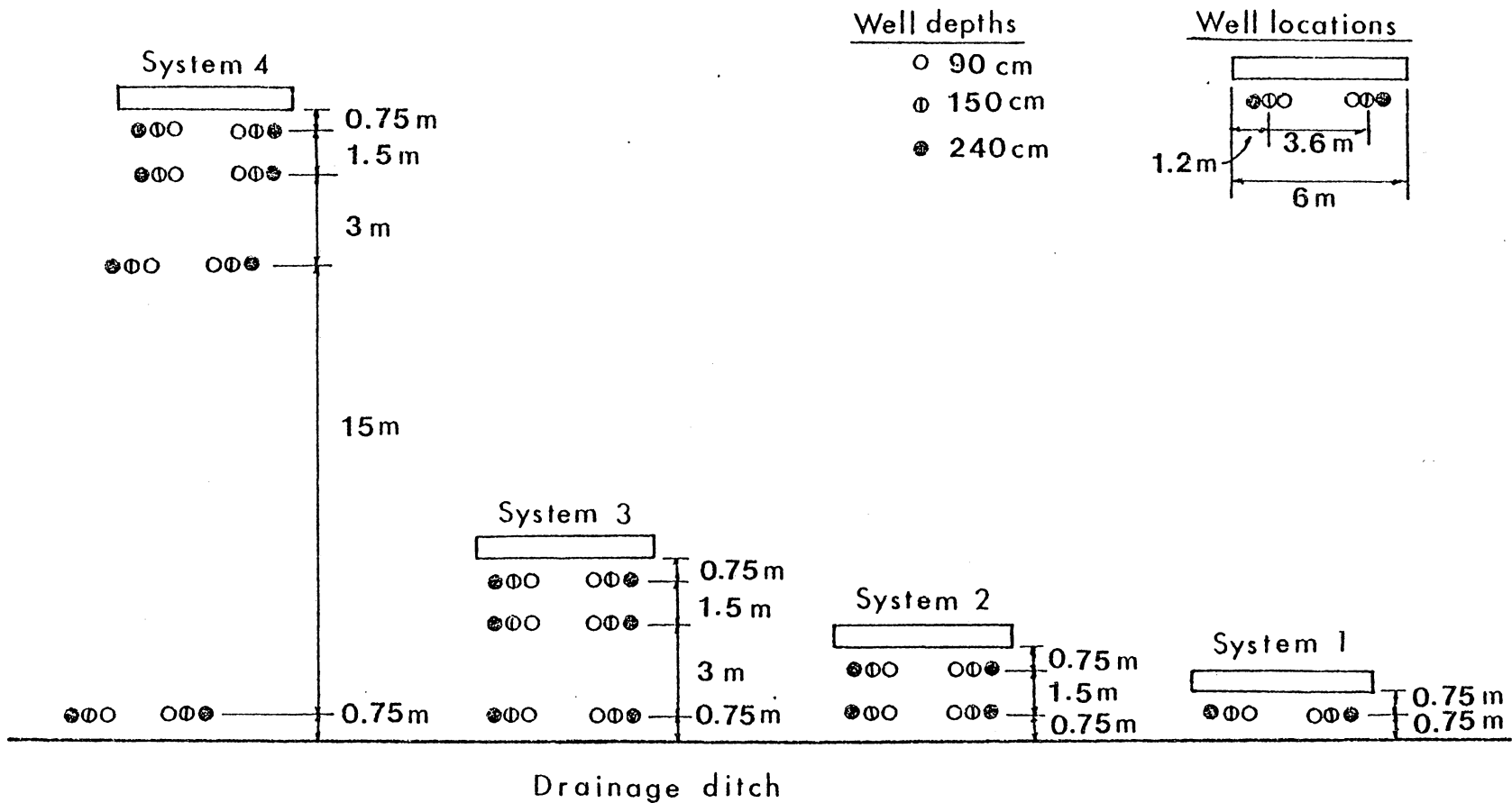


Figure 4. Relative locations and depths of ground water sampling points.

have wells at 0.75-m distance from the absorption trench and also 0.75-m distance from the edge of the drainage ditch. Three of the systems (System 2, 3 and 4) have wells at 2.25 m, two (Systems 3 and 4) have wells at 5.25 m and one (System 4) has wells at 20.25 m.

Monitoring well installation methods and construction prevented surface water contamination and pipe seepage and allowed removal of groundwater with minimum sediments from depth zones of 75-90 cm, 120-150 cm and 210-240 cm (See Figure 5). Normal well installation consisted of augering an 8-cm diameter hole to the desired depth, placing coarse sand around the perforated portion of the sampling pipe, adding several cm of sodium-saturated bentonite, backfilling, and tamping the original soil to within 20 cm of the surface. This was followed by cement-sand grout near the surface into which a short section of a 10-cm protective pipe was embedded.

When fluid sands were encountered, the sampling wells were pushed to the desired depth followed by addition of bentonite, backfilling, tamping, and grouting as above.

Background water quality levels were determined from remote ground waters collected 60 m from the drainage ditch at 1.5 m deep.

Sampling collection points for drainage water were established up and down gradient from the experimental area.

Three large wells 20 cm in diameter, approximately 200-cm deep, and grouted to exclude surface waters were located 4.5, 29.1 and 60 m from the edge of the drainage ditch and equipped with stage recorders (Model 5, Belfort Instrument Company) to continuously monitor water

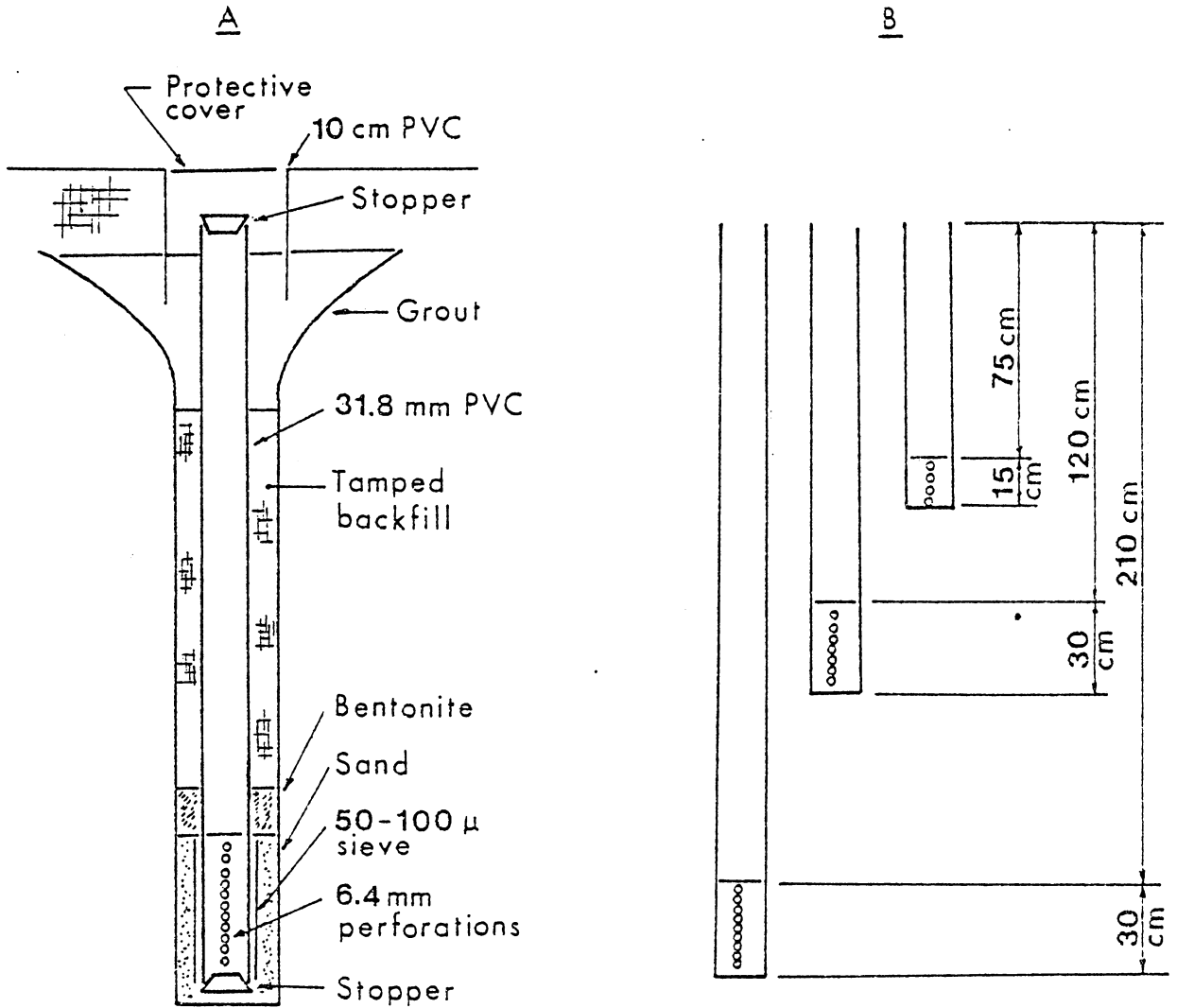


Figure 5. Sampling wells, A, construction and B, sampling depth zones.

table levels (See Figure 1). A tipping bucket rain gauge (Model 110, Weatherford Co.) was also installed to collect precipitation data.

Septic tank effluent was supplied from a flow-through chamber which had a 300 liter pump down capacity situated between the home's septic tank and conventional distribution box and pumped to each prototype-SAT. Effluent delivery was controlled using an adjustable normally-open timer connected in series with the pump. A standing pressure head at the distal end of each distribution pipe was set and maintained using 31.8 mm gate valves for throttling the STE flow. Weighted check valves (laboratory calibrated) were placed ahead of each mini-system to shut off flow simultaneously with the end of the pumping cycle. A check valve in the pumping chamber prevented backflow and allowed, along with the weighted check valves, the supply piping to remain full following the first pumping. The check valve arrangements provided quick fill up of the distribution pipes in the trenches. No gravity flow followed the end of a pumping cycle thereby maximizing uniform STE distribution among the SAT. The time differential for STE to reach positive pressure at each system was compensated by increasing standing pressure heads. Pressure heads and the timer were set to result in a 4-cm ( $42 \text{ liters/m}^2$ ) loading for each SAT per pumping cycle, which was scheduled for a single daily operation for the duration of the experiment.

The recommended loading rates based on estimated percolation rate of 18 min/cm (45 min/in) for the sandy clay loam soil is  $18.48 \text{ liter/m}^2$  (Va. State Board of Health, 1982). Measured percolation rates at the designed SAT depth averaged 4 min/cm (10 min/in) corresponding to a

34.86 liter/m<sup>2</sup> loading rate. The percolation test procedure (Va. State Board of Health, 1982) was modified slightly to include a three hour soaking period at a constant level 20 cm above the bottom of the hole. The percolation rate was taken as the steady rate of fall of the water level as the level dropped to the surface of the gravel filled bottom. Although the design loading rate was greater than even the rate based on actual percolation tests, it was not considered excessive because of the anticipated uniform application of the STE in the SAT expected from the pressure method of distribution.

Operation of the experimental system was started immediately after installation in September 1979 and continued until January 1982 with interruptions in February 1980 for a period of 1 to 14 days due to equipment failure and 30 days in July-August 1981 for resting the prototype trenches coinciding with the resident family's vacation.

Difficulty in adjusting and maintaining the standing pressure head at the distal end of each mini-SAT distribution pipe was experienced for the first six months of operation. During this time the standing wastewater levels at the distal ends of the distribution pipes had to be readjusted each site visit. The problem resulted from the necessarily small opening in the throttling gate valves. Closing the throttling valve sufficiently to create friction head loss with relatively low, 10 liter/min (2.67 gal/min), flow rates resulted in an extremely narrow slit at the gate valve seat. The narrow opening was observed to trap STE particulate matter, both organic and mineral, thereby reducing pressure and flow.



The problem was corrected by replacing the 31.8-mm gate valves at each system with 12.7-mm gate valves and the addition of 31.8-mm gate valve in the main header line. The combination of an overall pressure reduction caused by the main line valve adjustment and the change in shape (a wider slit) of the adjusted seating of the smaller valves eliminated clogging in the throttling valves. This resulted in easier and more accurate setting of the standing pressure heads which did not deviate with time more than 5 cm. Bimonthly observation of the standing effluent levels at each trench for the three months following the gate valve replacement and quarterly thereafter confirmed the proper functioning of the STE delivery system.

Monitoring wells (1.8-cm conduit) were driven to the soil-gravel interface to determine ponding of effluent in the trenches. Collection and analysis of groundwater samples and measurements of piezometer heads in the individual wells was begun on September 24, 1979 and was performed 33 times at 2-4 week intervals until August 15, 1981. The continuous water table and rainfall recorders were placed in operation September 1979 and monitored until January 1982.

Water samples were collected via permanent individual 6.3-mm I.D. flexible tubing under applied suction. Wells were completely evacuated before collecting samples in containers fitted with adapters. Containers and adapters were washed with 6N HCl and thoroughly rinsed with distilled water prior to use. Samples were placed on ice within one hour of collection and bacterial analyses for fecal coliforms begun within 24 hours after collection. Bacteria were enumerated by the

multiple-tube dilution technique at 44.5 C and recorded as the MPN/100 ml (APHA, 1971). These analyses were performed by the Virginia State Consolidated Laboratory Services-Bureau of Microbiological Science.

Chemical analysis included  $\text{NH}_4$ ,  $\text{NO}_3$ ,  $\text{NO}_2$ , Na, Ca, Mg, pH, EC, P, and Cl. Analysis for Na, Ca and Mg was done only every other sampling date. All samples were filtered stepwise using glass microfiber filters followed by 0.45 micron membrane filter to remove soil particles. Ammonium was determined by an ammonia ( $\text{NH}_3$ ) electrode (Banwart, et al., 1972). Nitrite was determined colorimetrically by a modified diazotization technique (USEPA, 1979). Nitrate was reduced to  $\text{NO}_2$  in a cadmium reduction column and the concentration determined colorimetrically as described above. Sodium, Mg and Ca were determined by atomic absorption specttoscopy, pH by glass electrode and EC by a Wheatstone-bridge type conductivity meter (USEPA, 1979). Orthophosphate was determined colorimetrically by a single solution technique (Murphy and Riley, 1962). Chloride was determined using a Cl-selective electrode (Selmer-Olsen and Oien, 1973).

Soil samples below each trench bottom were collected at experimental system termination at three equally spaced location and at midpoints between pipe perforations at depths of 0-4, 4-9, 29-34 and 59-63 cm. Subsamples were shaken in 2N KCl solution (~10 ml per gram) for 24 hours and filtered as above. The extracted solutions were analyzed for  $\text{NO}_3$  and  $\text{NH}_4$  as described above.

Statistical Analysis System (SAS), (1976) in conjunction with Virginia Tech's computer facilities was used for data reduction and

analysis.

Precipitation data to supplement the on-site rain gauge data was obtained from the USGS weather station at Lake Drummond which was located approximately 16 km from the experimental site.

## RESULTS AND DISCUSSION

### System Performance

Following final adjustments and a few modifications, the experimental setup and the prototype systems appeared to function successfully with the exception of System 1 where effluent reached the ground surface.

Evidence of surfacing effluent, black deposits (probably ferrous sulfides) were observed for System 1 as early as July 1980, approximately 300 days after system start-up. The volume of STE reaching the surface appeared small and did not flow continuously overland to the drainage ditch except possibly during heavy rainfall in a diluted solution. The probable reason for STE reaching the surface since the water table level was well below the trench bottom was continuously ponded effluent in the trench. It was thought that storage was not available in the gravel pore space and the effluent surfaced from positive pump pressure. In efforts to rectify the problem at System 1 and prevent a similar occurrence at the other systems, the loading rate was reduced by 20 percent to  $33.6 \text{ liters/m}^2/\text{day}$  ( $0.8 \text{ gal/ft}^2/\text{day}$ ) in July of 1980.

The failure of System 1 continued in similar fashion for two or three additional months. Evidence of STE on the surface disappeared in February 1981, only to reappear in May 1981 at which time the loading rate for all the systems was again reduced, this time to  $25.2 \text{ liters/m}^2/\text{day}$  ( $0.6 \text{ gal/ft}^2/\text{day}$ ). There was no evidence of STE reaching the surface at the other three sites. Two months later the black deposits at Site 1 had expanded. At this time, the entire experimental system was

shut down to coincide with the residence family's vacation and to rest the SAT's. Pipes were placed in the trench gravel to determine the depth of ponding at this time in each SAT. No ponding of effluent was observed in any of the mini-SAT's nor was any present for the duration of the experiment.

In August 1981, the supply pipe to the mini-SAT at Site 1 was severed and capped. Daily dosing of the other three SAT's was resumed at the original loading rate of 42 liters/m<sup>2</sup>/day (1 gal/ft<sup>2</sup>/day).

The SAT at Site 1 was excavated and the distribution pipe and gravel uncovered. Cross sections at these locations revealed no dark gelatinous layer at or beneath the soil-gravel interface which represents biological clogging (Bouma, et al., 1972). No evidence of biological buildup was actually expected in view of no measured effluent ponding. All the perforations in the distribution pipe were open except one which was partially closed by a piece of drill shaving. In the trench below the area of effluent surfacing, the interface between the gravel and the distribution pipe was glazed over with soil particles including the area beneath the perforations. This sealing of the gravel surrounding a point of STE application accounts for the presence of effluent on the surface even though effluent was not ponded in the trench. The pressurized effluent followed the path of least resistance which happened to be upward through the disturbed backfill. The presence of large quantities of soil particles at the pipe-gravel interface could have resulted from the combination of the lack of a suitable cover (i.e. straw, construction felt, etc.) over the distribution pipe and gravel

and the landscape position of the SAT.

The position of the mini-SAT's near the drainage ditch were in the path of surface runoff because open drainage ditches are designed to remove both surface and groundwaters. The two nearest systems (1 and 2) were initially adversely affected by surface waters. Small gullies in the mounded fill for these systems resulted from erosion caused by rapidly moving surface water during high intensity rainfall.

Surface swells and swallow drainage ways were constructed in August 1980 to prevent surface water intrusion into the two mini-SAT nearest the drainage ditch (Systems 1 and 2). The other sites being further away from the drainage facility and not situated in a concentrated surface water flow path were not affected by surface runoff. The erosion process which resulted from large quantities of water was also undoubtedly accompanied by saturated zones in the backfill material above the distribution pipe and gravel. The cohesive forces binding finer soil particles are greatly lessened by saturation conditions especially in disturbed soil thus affording movement of soil sediments with soil water. Soil particles in water would naturally seep along the smooth surface of the distribution pipe and be deposited at the pipe-gravel interface. In this case, bridging by soil sediments between individual gravels during the movement and deposition phase may have been possible because of the small diameter gravel which resulted in narrow pore spaces. The use of larger diameter gravel may have prevented or delayed the above sealing phenomena.

The movement of disturbed soil materials overlying SAT is supported

by Harkin (1980) who credited the movement of fine materials to trench bottoms, following extreme ponding which saturated the backfill material, as the cause for nonrestorability of infiltration rates of trench bottom surfaces by hydrogen peroxide treatment.

The same sealing phenomena was also observed at another research site in Virginia under similar conditions except the medium was nongraded builders sand containing fine sand particles.

### SAR Effect

In spite of a high STE-SAR values (See Table 2), the increased sodium levels had no obvious adverse effect on the infiltration of STE into the natural soil surrounding the SAT's. Although the hydraulic conductivities of the soil near the soil-gravel interfaces was not measured, the lack of measurable ponding in any of the trenches indicates no significant reduction of hydraulic conductivity as a result of the high SAR's in the STE.

The elevated SAR of the STE results from the use of a household water softener which discharges the exchanged divalent ions plus a high salt (NaCl) content into a surface drain rather than the septic tank.

The apparent lack of detrimental effect from the high SAR's on the hydraulic functioning of the SAT is probably due to a combination of factors. The low amount of montmorillonitic swelling clays in the Tomotley soil is one factor. Another is the generally high amounts of iron and aluminum in Ultisols which according to Lagerwerff, et al., (1969) lessens the deleterious effect of high SAR-low salt content on soil systems. Reneau (1978b) suggested preferential absorption of Ca

Table 2. Mean selected chemical and biological concentrations in household water, septic tank effluent and remote groundwaters at the 1.5-m depth.

	Household Water	Septic Tank Effluent	Ground Water
<u>E. coli</u> (MPN/100 ml)	1.6	$5.72 \times 10^5$	1.6
pH	7.62	8.0	7.9
EC ( $\mu\text{mhos/cm}$ )	365	641	61.7
P (ppm)	0.188	13.2	<.001
Cl (ppm)	61.8	101.1	10
NH <sub>4</sub> -N (ppm)	0.146	30.7	<.01
NO <sub>3</sub> -N (ppm)	0.015	0.116	<.001
NO <sub>2</sub> -N (ppm)	0.01	0.054	<.0001
Na (ppm)	171.2	315.8	14.5
Ca (ppm)	4.4	6.45	60.4
Mg (ppm)	0.68	1.19	2.84
SAR (meq/l) <sup>-1/2</sup>	39.77	31.72	0.58



and Mg under acidic soil conditions that would lower the ESP which is conducive to reducing soil swelling. A combination of the above factors and primarily the low amounts of montmorillonitic clay minerals could account for the lack of an adverse effect of the high SAR in the STE on hydraulic conductivity.

The effects, if any, were delayed by the presence of relative high levels of Ca indicated by the background levels which averaged 60.4 ppm compared to the mean groundwater Ca concentrations of 3.05, 1.16, 1.79 and 6.74 ppm for System 1, 2, 3 and 4, respectively at distance of 75 cm and depth of 1.5 m. The increase of SAR's with time can be seen in Figure 5 where after 15-16 months the SAR in soil solution near the SAT's were similar to SAR values in the STE and approaching steady state conditions for System 2 and 3. The absence of data during the period April to January, is due to the combination of analyzing samples for Ca, Mg and Na every other sampling date and lack of sufficient groundwater in the 120 to 150-cm sampling zone on many occasions. The original large amounts of Ca on the soil exchange complex was exchanged with time for Na by mass action and leached from the area.

Since no adverse effects were observed as evidenced by the absence of continuous ponded effluent in the SAT's, there is no reason to suspect that the elevated SAR in the STE will cause system failure. This is supported by other investigations on the same soils (Stewart and Reneau, 1981b) and similar soils (Reneau, 1978b).

#### Distribution of STE

The ideal operation of an SAT incorporating perforated low pressure

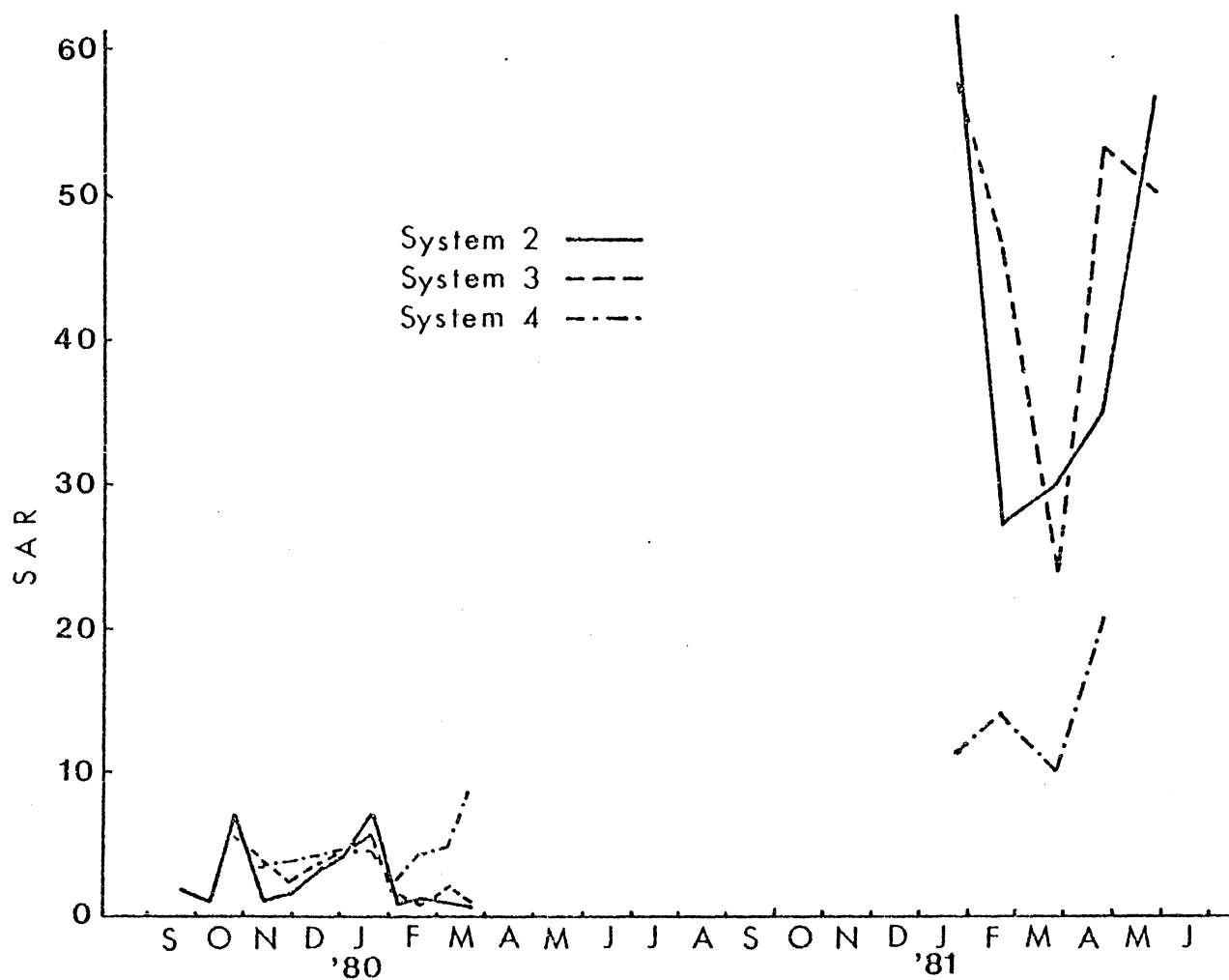


Figure 6. SAR of ground waters at 0.75-m distance and 1.5-m depth for each prototype system with time.

piping is to spread STE evenly over the trench bottom and maintain uniform infiltration on a volume basis through the soil-gravel interface. Because effluent is pumped into the gravel at a rate greater than the soil infiltration rate, the levelness of the soil-gravel interface at the trench bottom is paramount to maximizing uniform application of STE to the entire trench bottom area.

Untreated STE is necessarily pumped at greater rates relative to soil infiltration rates because of the large diameter holes (3 mm) needed to prevent clogging of the pipe perforations from the organic and mineral particulate matter contained in STE. The large holes and the associated hydraulics to maintain positive pressure in the piping network dictate large pumping rates. A sloping or uneven trench bottom would cause the effluent to transverse the gravel and result in hydraulic overloading of the lower parts of the trench bottom.

During the pumping cycle, effluent quickly flows through the non-resistance gravel to flood the trench bottom. The depth of ponding for an ideal system would be the same at all points but otherwise the depth of standing effluent at any point depends on the elevation of the trench bottom at that position. At any instance during the flooded stage, a potential energy difference in effluent filling the gravel pore space does not exist. Infiltration of effluent into the soil results from a potential difference caused by soil matrix suction across the soil-gravel interface which will result in practically vertical streamlines. The volume of effluent stacked above a given area of the trench bottom during the pumping cycle then reasonably represents the STE

loading for that area.

In order to obtain uniform infiltration on a volume basis per unit area ( $L^2$ ), regardless of the dimensions of  $L$  the soil must be homogeneous in space. In natural soils, homogeneity of  $K$  and infiltration is relative to the amount of soil area and surface characteristics of the soil area. The change in  $K$  within an area ( $L^2$ ) is very erratic when  $L$  is very small because of soil microstructure (soil solids and pore space) but approaches a constant value as  $L$  increases because of incorporating a variety of soil pore sizes and shapes (King, 1974). For purposes of this discussion,  $L$  is considered sufficiently large and  $K$  constant.

The STE pumping dose, trench length and slope of trench bottom affect the distribution of effluent in the trench gravel relative to the bottom area. The design of SAT using pressure distribution should specify level trench bottoms to achieve maximum use of the trench area and eliminate spot overloading. The levelness of the trench bottom is the most difficult to achieve compared to the pumping dose and trench length. Initial leveling of trenches with backhoes and trenching machines is dependent on the equipment operators. Careful reworking of the trench bottom grade and the use of survey instruments is generally not practiced by ST-SAS contractors.

The effect of factors that control the distribution of effluent in the gravel and STE loading of the trench bottom can be seen in Table 3. The calculations used for Table 3 were based on the assumption of 40 percent pore space in the gravel. The values of the pumping dose,

Table 3. The interaction of SAT dosing, length and slope on STE loading of trench bottom.

Dose	Length	Slope	Trench loaded with half dose	Total trench used
(cm)	(m)	%	%	%
2	20.5	0.1	44.9	100
2	20.5	0.2	40.1	100
2	20.5	0.4	32.1	100
2	20.5	0.7	24.5	83.7
2	20.5	1.0	20.4	69.8
2	20.5	2.0	14.5	49.4
2	1.5	0.4	48.5	100
2	7.5	0.4	42.7	100
2	13.5	0.4	37.4	100
2	20.5	0.4	32.1	100
2	27.5	0.4	27.9	95.3
2	30.0	0.4	26.8	91.3
.4	20.5	0.4	14.5	49.4
1.2	20.5	0.4	25.0	85.5
2.0	20.5	0.4	32.1	100
2.8	20.5	0.4	36.4	100
3.6	20.5	0.4	39.1	100
4.0	20.5	0.4	40.1	100

length and slope when held constant, represent the medians of what could be expected in actual field installations. Clearly, short trenches, large doses and slight slopes represent the conditions that provide the most uniform infiltration on a volume/area basis. Even slopes as slight as 0.7 percent can create an STE overload averaging twice the design amount in half of the trench for the median values of a 2-cm dose (.5 gal/ft<sup>2</sup>) and 20.5-m trench length. The concern with hydraulic overloading is the development and spreading (creeping failure) of a biological crust at the soil-gravel interface which could eventually cause the failure of the SAT to infiltrate STE (Bouma, et al., 1972 and McGaughy and Winnberger, 1964a, b).

The undesirable effects of a sloping trench bottom is shown to be lessened by shortening the trench length and increasing the amount of STE dosed per pumping cycle. This suggests the addition of vertical physical barriers in the trench gravel at intervals coinciding with distribution pipe perforations would be highly cost-effective and contribute to the longevity of the SAT. Additional benefits could probably be derived also from increasing STE storage to result in a larger dose per pumping cycle.

For this study, during pumping, effluent was added at a rate of 8 mm/min compared to 1 mm/min soil infiltration rate based on the saturated K of the horizon encompassing the design depth. The fast pumping rate undoubtedly caused temporary flooding of each SAT.

The conditions and construction procedures under which the SAT's in this study were operated and installed were believed adequate to

insure uniform infiltration of STE through the soil-gravel interface. The short 6 m trenches were placed on the contour and dosed heavily at 2.4 to 4 cm. The soil at the trench bottom depth was observed to be uniform across the length of the trench and the effluent was introduced through seven equally sized and spaced perforations.

In spite of the design, controlled installation and operational procedures, nonuniform STE movement out of the SAT was apparent. The concentrations of STE constituents in the groundwaters near the SAT's and  $\text{NH}_4$  analysis of soil samples taken immediately below each SAT indicated uneven distribution of STE. The replicated sampling wells should contain similar concentrations of STE components provided effluent is uniformly infiltrating into the natural soil. A Duncan's multiple range test (Steel and Torrie, 1960) was used to test the difference in the means between Rows 1 and 2 for Cl, EC and  $\text{NH}_4$  concentrations in the most adjacent groundwaters at a distance of 75 cm and 1.5-m deep for each system. The above STE components were selected because of the large differences in concentrations between background groundwaters and STE (See Table 2). Chloride was also chosen because of its unrestricted movement in soils (Cl is not absorbed by negatively charged soil particles and is not subject to biological transformations). Significant differences between groundwater concentrations in Rows 1 and 2 existed for Systems 2 and 3 for all selected variables as can be noted in Table 4. No significant differences between Row 1 and 2 for System 4 were indicated for any of the variables. The above results agree with  $\text{NH}_4$  concentrations in the soil below the trench bottoms (Table 5) where

Table 4. Duncan's multiple range test for differences between rows of sample well means at the 75-cm distance and 1.5-m depth for each system for selected variables.

Variable	Row	System		
		2	3	4
		- - - - - ppm - - - - -		
Cl	1	53.8 a <sup>†</sup>	17.4 b	104.5 a
	2	21.8 b	82.6 a	97.7 a
NH <sub>4</sub> -N	1	1.13 a	0.04 b	0.09 a
	2	0.19 b	1.08 a	0.06 a
EC	1	181.8 a	45.9 b	206.6 a
	2	74.8 b	217.7 a	212.2 a

<sup>†</sup>Different letters represent a significant difference at the 5% level.



Table 5. Ammonium-nitrogen concentrations in soil below prototype trenches midway between distribution pipe perforations at both ends and the middle of the trench and the mean of control points at experiment termination.

System	Depth	Sampling Location		
		1	2	3
	(cm)	ppm		
2	0-4	110.50	40.27	24.87
	4-9	7.42	58.76	3.89
	9-14	17.62	13.10	1.56
	24-29	13.12	3.93	2.83
	59-64	24.55	14.56	82.32
3	0-4	3.40	4.55	16.86
	4-9	1.72	3.55	10.18
	9-14	3.08	4.55	1.57
	24-29	2.36	7.46	5.09
	59-64	15.13	7.57	0.91
4	0-4	4.84		5.47
	4-9	8.17		2.66
	9-14	3.93		2.69
	24-29	1.71		3.67
	59-64	.86		1.74
Control	0-10	—————	2.43	—————
	25-30	—————	2.77	—————

Sampling Location 1 and 3 correspond with Row 2 and Row 1. The middle sampling location for the SAT at Site 4 was mistakenly not sampled. The disparity between  $\text{NH}_4$  concentration at sampling locations for the 0 to 4-cm depth immediately at and below the soil-gravel interface is contributed to the unevenness of the trench bottom.

The difference in groundwater concentrations between rows of wells and soil concentration may also result from soil heterogeneity particularly of soil macropores. The presence of macropores would allow uneven infiltration on a volume/area basis from the trench and provide a more direct and quicker route to the groundwater affording less time and soil contact for soil "sorption" processes. Macropore flow could also account for the increased  $\text{NH}_4$  concentrations with depth below the trench bottoms (See Table 5) in reverse of the normally expected decrease with depth. Ammonium concentration increases at deeper depths was prevalent at System 2. The lack of significant differences in the groundwaters of Rows 1 and 2 for concentrations of Cl, EC and  $\text{NH}_4$  at System 4 and the small difference in  $\text{NH}_4$  accumulated in the soil below and at the ends of the trench suggest uniform effluent distribution and infiltration for this system.

Even though there is an apparent difference in uniformity of STE infiltration across the soil-gravel interface between systems, the fact that  $\text{NH}_4$  concentration in the soil immediately below the trench (0-4 cm) at each sampling location for each system was greater than the control indicates that effluent was infiltrating over the entire trench bottom. Each system received STE more uniformly than would be possible

with a conventional gravity flow arrangement.

### Saturated Zone

This section will discuss water table behavior during the study period as saturated conditions at or near the absorption trench bottoms resulting from high water tables can affect the normal operation of SAT and the movement of biological and chemical components from STE.

The water table levels were generally lower during the study period than in previous years. The average water table level at 60 m from the drainage ditch during the period January through April, during which time the evaporation-transpiration potential is low and water levels are highest, are listed in Table 6. The differences in levels between years are attributed to differential rainfall. The highest mean daily rainfall resulted in the shallowest water table levels. There appears to be a water storage carryover effect from the high rainfall year 1979 to 1980 which resulted in a relatively shallow water table levels for the first four months of 1980 in spite of low rainfall during this period. Rainfall deficits throughout the remainder of 1980 and continuing into 1981 resulted in deeper water table depths.

The abnormally low water table levels during the experimental period enhanced the overall operation of the prototype systems and soil treatment capacity resulting from the increased amount of unsaturated flow beneath the trench bottoms.

The seasonal variation and general trend of water table levels are illustrated in Figure 7. The water table reaches its highest level for any given year in the mid to late winter (January-March), falls

Table 6. Mean water table depths at 60 m from drainage ditch and mean rainfall for January through April and total rainfall for the years 1978-1981.

	-Year - - - - -			
	78	79	80	81
	- - - - - cm - - - - -			
Mean water table depth (cm)	60 <sup>†</sup>	56.3 <sup>†</sup>	63.3	98
Mean daily rainfall (mm)	3.8	4.9	2.8	1.8
Total annual rainfall (cm)	128.2	151.2	90.5	100.1

<sup>†</sup>From Stewart and Reneau (1981b).

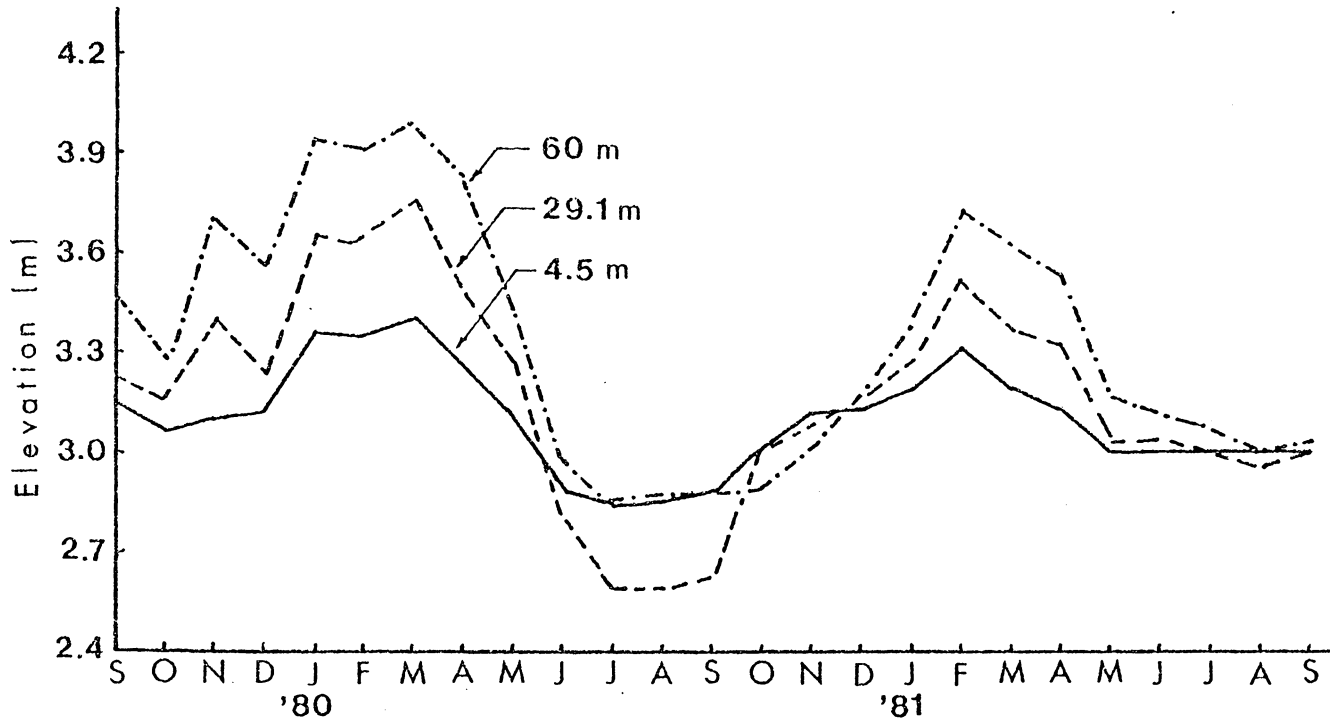


Figure 7. Monthly mean water table elevations above sea level with time for 4.5, 29.1 and 60 m from the drainage ditch.

during the spring (May-June) to a summer low and rises again in the fall. The effect of the drainage ditch is evident during periods of high water levels as water table elevations decrease with closeness to the drainage ditch. The influence of the ditch diminishes as the water level declines to lower positions as noted by small elevation differences with distance. When water table levels are near and below the drainage ditch invert elevation, 3 m, the water table gradient was directed inward from the ditch towards the 29.1-m distance. This indicates the presence of an additional and orthogonal gradient. The perpendicular gradient probably results from the large and deep drainage canal, the Portsmouth Ditch, adjacent to the property and results in channelized flow from the area of the continuous recorder at the 29.1-m distance.

The periodically measured water levels in the individual sampling wells as well, as the continuous recording discussed above, revealed a water table gradient away from the perimeter ditch when the water table was at or below the 3-m elevation.

Water table fluctuations are dampened by the influence of the drainage ditch as can be noted in Figure 7 and are detailed in Table 7. The percentage of time the water tables peaked at the highest levels was less for the 4.5-m distance. Neither at the 4.5-m distance nor the 29.1-m distance did the water table rise to the 45-cm depth. On the other hand, the water table remained above the 105-cm depth at the 4.5-m distance for a longer period of time, 51.7 percent compared to 16.8 percent for 29.1--m distance and 33.9 percent for the 60-m distance. The additional

Table 7. Percent of time of water table levels remained in given depth zones with distance from the drainage ditch for a 762-day period interval.

Distance — m —	Depth Zone (cm)									
	0- 15	15- 30	30- 45	45- 60	60- 75	75- 90	90- 105	105- 120	120- 135	135+
4.5	0	0	0	0.4	5.4	13.4	32.5	22.6	23.9	1.8
29.1	0	0	0	1.1	1.3	6.5	7.9	15.1	9.4	58.7
60	0	0	1.0	6.5	6.1	10.8	9.5	10.9	5.6	49.5

gradient towards the canal mentioned before is believed to effect lower water tables at the 29.1-m position and to account for the low percentage above the 105-cm depth, which is very close to the 3-m elevation of the ditch invert.

For the landscape position near the ditch, the advantage of less water table fluctuations and less peak time at shallower depths is offset by the greater amount of time at intermediate depths. The intermediate depths in this case were 60-105 cm below the trench bottom which could affect the long term operation of the trench if biological crusting develops. A crust resistant to flow would create ponding in the trench and result in an equilibrium soil suction below the trench bottom (Bouma, 1975). The steady state soil moisture tension (SMT) below drainfield trench bottoms ranges between 20-100 cm of water according to Bouma, et al. (1972) and varies with effluent quality and soil texture. Under trench ponding conditions, the driving force to move the STE has two main components, the standing head above and the SMT below the trench bottom. Encroachment of the water table into the unsaturated zone could lessen the necessary SMT and reduce effluent flow. This could ultimately cause failure of the trench.

The analysis and discussion of water table fluctuations and percent time in depth zones below the surface is based on the ground surface as shown in Figures 2 and 8. The slope of the ground surface between the 29.1-m distance and the 60-m distance was .02 percent and 1.5 percent between 4.5 and 29.1 m for an average slope of 0.7 percent from 60 m to 4.5 m. Had the ground slope been flatter and sloped only



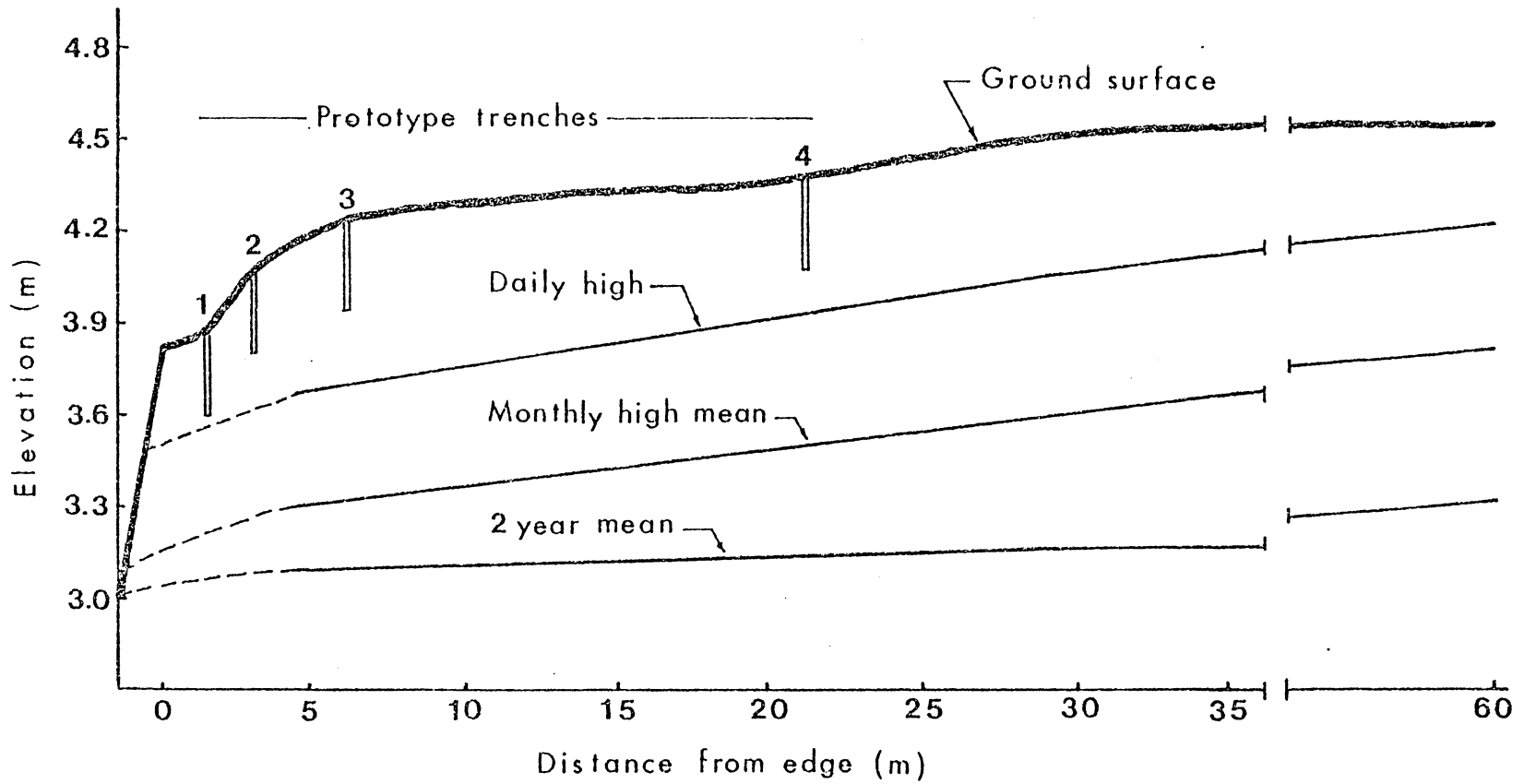


Figure 8. Profile of study area including the prototype trenches, water table levels and ground surface.

0.17 percent, the amount of time the water levels remained in given depth zones would have been relatively unchanged at 29.1 m but greatly changed at 4.5 m. The effect of a gentler ground slope (0.17 percent) would result in the water table level rising above the 105-cm depth at 4.5-m distance only 5.8 percent of time compared to 51.7 percent with 0.7 percent average slope.

The anticipated increase in the depth of the unsaturated zone with decreasing distance from the drainage ditch resulting from the water table drawdown did not take place. The 2-year average depth of the water table below the trench bottoms as measured periodically from the individual wells at the 75-cm distance from the prototype systems was 59.1 cm for System 2, the closest system to the drainage ditch, 71.4 cm for System 3, and 77.1 cm for System 4, the system most distant from the drainage ditch. The depth of the unsaturated zones below each system was positively correlated with distance from the drainage facility rather than decreasing with distance away from the trench.

The reasons for the reversal of the anticipated trend are the gently sloping water table drawdown curve in conjunction with the excessive sloping of the ground surface. A profile of the study area in Figure 8 shows the slope of the ground surface to be greater than the slope of the water table, especially within 10 m of the ditch. The ground surface slope between the edge of the ditch and 6 m is 8.1 percent and the 2-year mean water table slope for the same distance is 1.0 percent. The average ground slope between 4.5 to 29.1 m from the

drainage ditch was 1.5 percent and 0.7 percent between 4.5 and 60 m. The water table at its highest monthly mean level was gradually sloped (0.8 percent) from the 4.5 to 60-m from the drainage ditch.

The sloping of the ground surface is necessary for surface drainage that lessens the load of the underground drainage pathways. The amount of surface runoff depends on the surface infiltration capacity which varies with the porosity of the upper soil horizon and the antecedent moisture conditions (Baver, et al., 1972). Generally speaking runoff coefficients for 0-5 percent slopes are .10 to .30 for sandy loam to silt loam surface textures (Schwab, et al., 1965). Surface runoff is decreased by good grassland or woodland cover, high soil infiltration, depressional areas and relatively flat terrain (U.S. Soil Conserv. Svc., 1953). Schwab, et al. (1965) reported that depressions of 3-6 cm (.1-.2 ft.) can greatly reduce runoff on slopes of 1-2 percent of loam surface textures.

The areas where drainage would most likely be exercised in conjunction with on-site sewage disposal is in the Coastal Plain region or otherwise relatively flat terrain and would have woodland or grass vegetative cover which minimizes surface runoff. Considering that any slope towards a drainage facility decreases the unsaturated zone above the water table, the value of surface grades greater than 0.1-0.2 percent when on-site sewage disposal systems are involved would be negative. The importance of preventing surface runoff from entering the drainfield and adding to the hydraulic load on the trenches is of course recognized and merits extra precaution and expenses to limit depress-

ional areas in the immediate vicinity of the drainfield trenches.

### Effluent Movement

This section will consider movement of STE away from the drainfield trenches first in general terms, secondly the extent of effluent movement using the free moving chloride (Cl) ion as an indicator and salt accumulation which is denoted by electrical conductivity (EC) and thirdly, movement of STE constituents that are directly related to ground and surface water pollution, namely bacteria, N, and P.

Soil solution chemicals and microbes move with the soil water in response to a hydraulic gradient. Diffusion is recognized but considered short-ranged and is a much slower process than mass movement. Effluent introduced into a relatively dry soil will flow and fill the smallest soil pores and continue to move under the influence of gravity and soil suction until an equilibrium is reached, that is, when capillary forces equal gravitational forces.

During effluent flow through soil pores, interaction occurs between the soil solids and components of soil solution and may result in absorption of ions on soil exchange sites, precipitation, biochemical transformations and physical filtration. Slow movement in small diameter pores enhance the soil "sorption" process.

The potential for soil "sorption" of effluent components is greater above the water table than in the saturated zone since flow at least partially takes place in smaller pores where the soil contact area per volume of soil solution is large compared to increased macropore flow below the water table and where the soil solution is more diluted.

The nature of operation of an on-site sewage disposal system tends towards steady state conditions only when considering long periods of time or when effluent is continuously ponded in the drainfield trench. A fairly constant amount of STE is applied on a daily basis. Within a 24-hour period, however, the effluent is not uniformly applied with time but dosed rapidly within minutes, a uniform amount at a specific interval in the case of pressurized distribution and varying amounts and intervals in the case of gravity distribution.

The daily effluent movement routine involves flooding the trench bottom, flow into the soil, and distribution within the soil. After system start up, the soil below the trenches is wetted and effluent movement downward is carried by the smallest pores that will accommodate the rate of the added STE. The effluent takes the pathway of least resistance and travels in the large pores immediately following the dose. The length of travel in the large pores depends on the continuity and the moisture status of the soil surrounding these pores. Effluent initially and momentarily ponded in the trench will flow mainly through macropores and continue to flow until absorbed laterally by soil adjacent to the macro pore or until the end of the macro pore is reached or until the effluent arrives at a water table where flow rate is necessarily reduced. The process of rapid movement through macropores has been termed "hydrodynamic dispersion" by Anderson and Bouma (1977). Ziebell, et al. (1975) demonstrated rapid movement of STE through drained laboratory columns containing a strongly structured soil following a dose that flooded the soil surface.

The climate also affects the movement of STE. During the periods of high evaporation-transpiration (E-T) downward movement is reduced because the overall drier soil status results in absorption of the effluent at shallower depths. If the rate of effluent application together with rainfall is less than the E-T rate, upward movement of the soil solution occurs. Rainfall in excess of E-T results in downward movement of effluent. The low intensity rainfalls of long duration would have the greatest effect on the downward movement of STE components as the flow pathway would be through the smaller pores and along the walls of the larger pores. High intensity rainfall of short duration should have less effect on leaching since flow would occur in the interior of the large pores bypassing zones of high STE component concentrations.

#### Chloride and Electrical Conductivity

Chloride is a reliable indicator of effluent movement since it is not readily absorbed by the soil and is not subject to biological transformations and hence moves freely in mass with the soil solution. Also the average Cl concentrations in STE is ten times greater than mean levels in the native groundwaters rendering the nonreactive Cl an effective tracer of STE movement.

The seasonal pattern of Cl concentrations measured in wells closest to the SAT and the depth which most closely approximates the level of the water table surface is shown in Figure 9. The highest concentration of Cl in groundwater is shown to be when the water table is highest during the wet winter months. The difference between the years is attributed to the large amount of rainfall during January-March

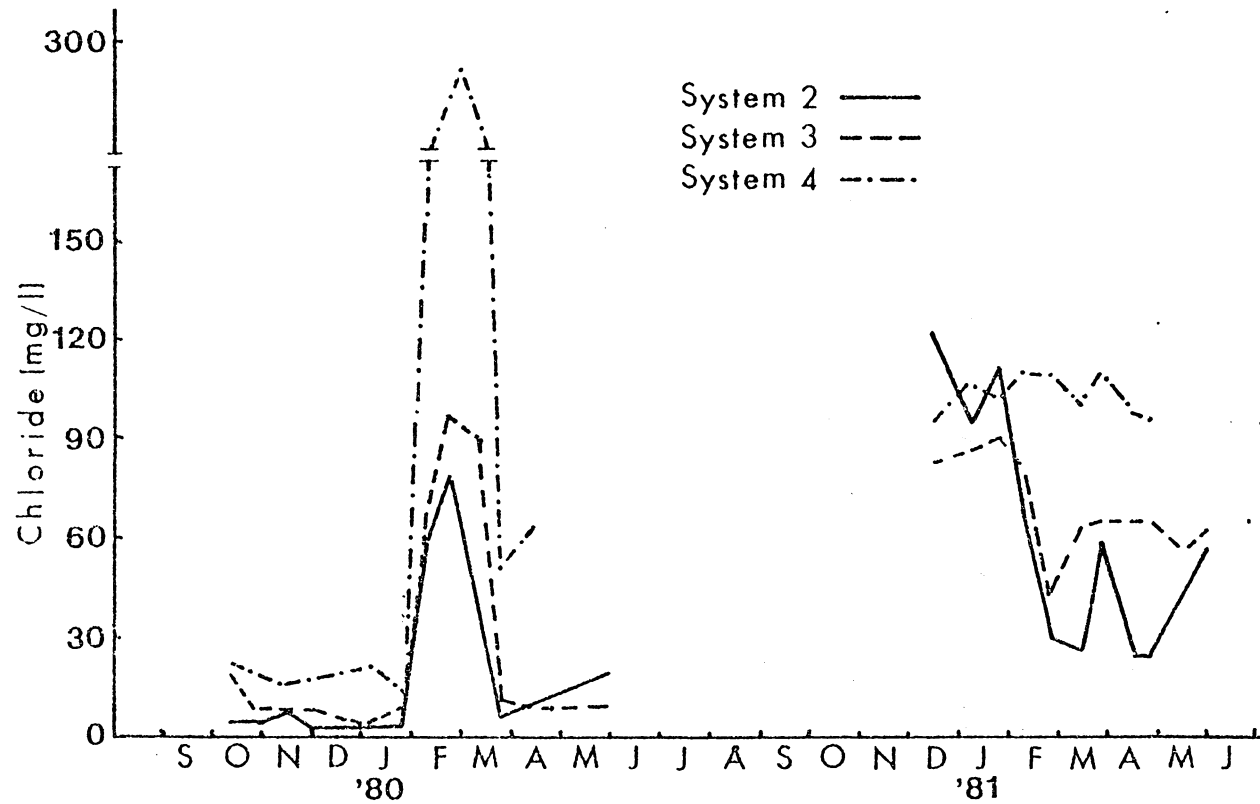


Figure 9. Chloride concentrations in ground waters at 0.75-m distance and 1.5-m depth for each system with time.

1980, 28.2 cm, compared to 16.7 cm for the same period in 1981. Additionally, the water table rose sharply and to greater heights in the winter of 1980 and gradually in 1981. The rapid rise in the water table could possibly cause a quick flush of the Cl accumulated in the vadose zone during low water table periods.

In considering the difference in groundwater Cl concentrations between years, rainfall and rate of water table rise in this case is more important since the mean difference in water table heights for the January-March period between 1980 and 1981 was only 16.8 cm at 4.5 m distance from the ditch and 33.8 cm at 29.1 m. The effect of the relatively high rainfall in the winter of 1980 is apparent when observing vertical movement of Cl (See Figure 10). Effluent movement as indicated by Cl is mainly horizontal in the upper portion of the water table and vertical only during periods of high rainfall and high water tables.

Lateral chloride movement as seen in Figure 11 again emphasizes the effect of high rainfall and a rapidly rising water table in the winter of 1980. The lateral Cl movement to a distance 2.5 m from the prototype system along the upper saturated zone was rapid for each system and reached peak concentrations in a short time. In 1981, peak concentrations for each system were reached over a longer time span and not simultaneously. The pattern in 1981 is influenced by a gradual rise of the water table and relatively low rainfall. The notable differences in 1981 in time of peak concentrations between systems is attributed to slope of the water table. The water table gradient near the drainage ditch in the area of System 2 was the greatest and diminished with



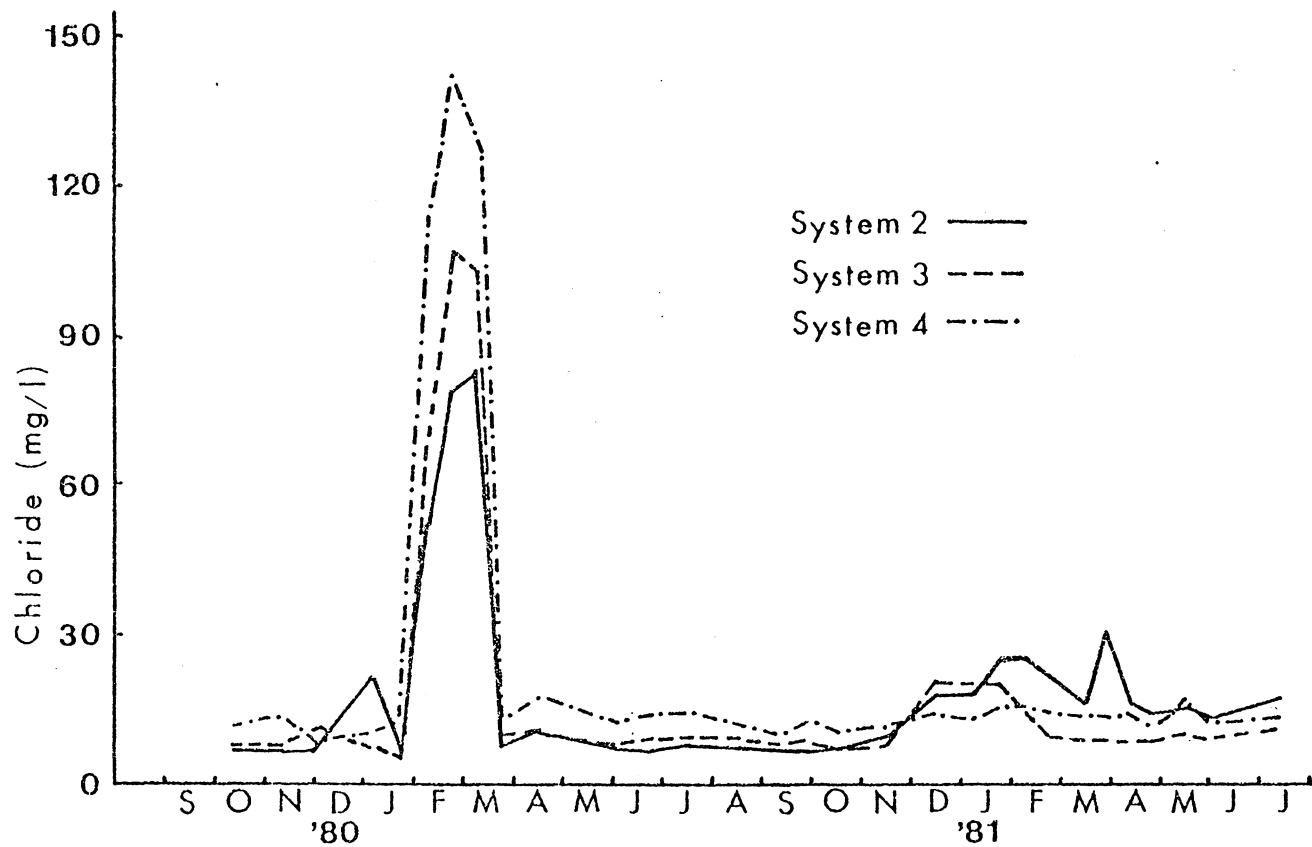


Figure 10. Chloride concentrations in ground water at 0.75-m distance and 2.4-m depth for each system with time.

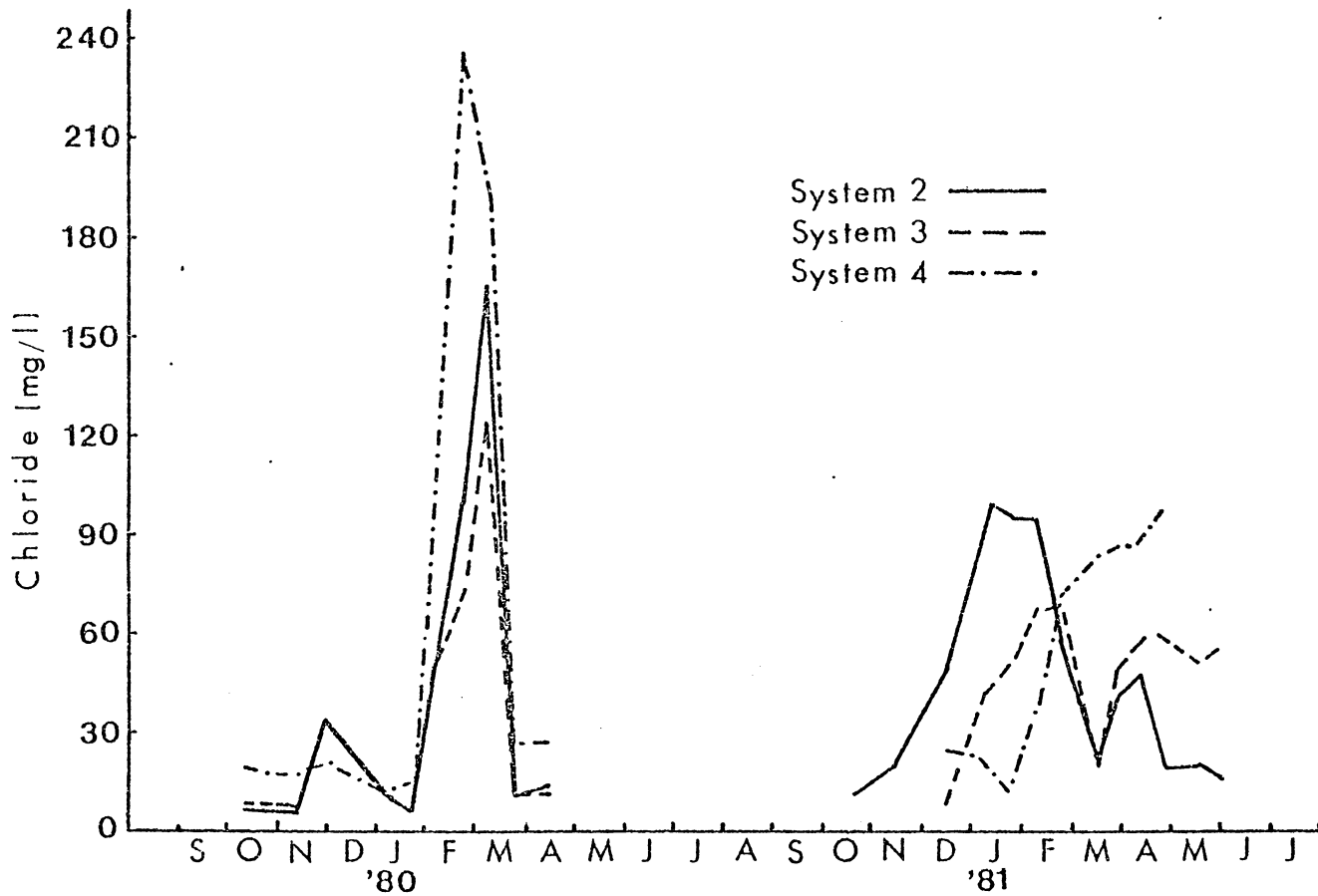


Figure 11. Chloride concentrations in ground water at 2.25-m distance and 1.5-m depth for each system with time.

distance from the drainage ditch. The differences in water table slopes (1 percent at System 4, 2.5 percent at System 3 and 3.2 percent at System 2) resulted in a lag time of approximately 1 month between Systems in lateral Cl movement in the upper saturated zone to the 2.25-m distance.

The EC of the soil solution/groundwater is indicative of total salt content within practical limits (U. S. Salinity Lab Staff, 1954). Since the mean EC of STE is several times higher than the native groundwaters (See Table 2) it can also be used to follow STE movement. The means of Cl concentrations and EC levels in groundwater leading towards the drainage ditch for sampling days when all wells were sampled are listed in Tables 8 and 9.

The samples from which the means were calculated do not include the start up period during which time the sampling wells were adjusting to the soil environment and the dosing rates were variable and not accurately known. The higher Cl concentrations and EC levels in groundwaters near and below System 4 could be due to a larger unsaturated zone (as discussed earlier) beneath the drainfield which acts as a storage area, especially when E-T rates are high, during low water table periods and subsequently mobilized and flushed when the water table rises. Another contributing factor to higher Cl levels associated with System 4 is the larger water table fluctuations which result in greater movement of effluent with the more frequent rise and fall of the water table surface into the vadose zone underlying the absorption trench.

Movement of STE from the three systems as indicated by Cl and EC is predominantly lateral along the upper zone of the water table with

Table 8. Mean chloride concentrations in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
		----- ppm -----			
2	1.5	56.5 b <sup>†</sup>	43.4 a		(a) <sup>‡</sup>
	2.4	15.3	15.3		
3	1.5	64.3 b	43.6 a	13.6 b	(b)
	2.4	11.4	8.9	9.9	
4	1.5	101.7 a	53.9 a	23.5 a	14.8 (b)
	2.4	13.3	13.5	15.0	19.9

<sup>†</sup>System means with different letters are significantly different at the 5% level in accordance with Duncan's multiple range test for the given depth and distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

Table 9. Mean electrical conductivities in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.75
		-----µmhos/cm-----			
2	1.5	184.5 b <sup>†</sup>	185.2 ab		(a) <sup>‡</sup>
	2.4	69.4	66.2		
3	1.5	178.7 b	117.9 b	55.6 b	(b)
	2.4	63.1	54.9	56.1	
4	1.5	283.1 a	198.2 a	133.9 a	87.7 (b)
	2.4	66.5	68.3	78.6	90.8

<sup>†</sup>System means with different letters are significantly different at the 5% level in accordance with Duncan's multiple range test for the given depth and distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

limited vertical movement. Lateral movement appears to be a function of flow through unrestrictive macropores and the rate of travel determined by the water table gradient. The decreases in Cl and EC levels with distance away from Systems 3 and 4 results from dilution and possibly dispersion. Dispersion would be contrary to Caldwell's (1937) study which shows virtually no movement of bacteria normal to the flow path of point discharges from pit latrines. Chemical constituents, however, have higher dispersion coefficients than bacteria because of ion size and resultant mobility differences and can be expected to diffuse normal to the flow path. Crane and Gardner (1961) illustrated transversal dispersion equals longitudinal dispersion but only at very low velocities around  $10^{-5}$  cm/sec. The velocity of the groundwater at Systems 3 and 4 during 1981 are estimated from the lag times in Figure 11 to be  $2.8 - 5.9 \times 10^{-5}$  cm/sec. (1-2 in/day). The low groundwater velocities increase the possibility of transversal dispersion which could partially account for decreases in Cl concentration with lateral distance. However, the partitioning of dispersion and dilution is unknown. The main cause of Cl decreasing concentrations with distance is believed to be dilution.

The lesser reduction in Cl and EC with distance at System 2 as compared to Systems 3 and 4 may result from the faster movement of the groundwater as a result of the steeper water table gradient allowing less time for rainfall dilution.

In summary the movement of STE as indicated by Cl, a conservative ion, and EC, practically conservative since electroneutrality must be

maintained, shows flow primarily occurs laterally towards the drainage ditch in the upper regions of the water table during the winter months when water tables are highest. Vertical movement below the water table is slight and is limited to periods of heavy rainfall in conjunction with high water table levels. The greater unsaturated storage area and greater water table fluctuations underneath System 4 as compared to Systems 2 and 3, resulted in greater concentrations of Cl and salts when water table levels increased. The lack of reduced Cl and EC levels with distance for System 2 is attributed to the rapid movement of effluent as influenced by a larger water table gradient affording less time for rainfall dilution.

Chloride and salt content are not considered environmentally important but discussed here only to indicate the potential movement of STE. The movement of more pollution sensitive constituents of STE will be discussed next.

### Bacteria

Escherichia coli bacteria is the most widely used organism to indicate fecal contamination and is the main component of the fecal coliform group (Cabelli, 1978). Because of the large numbers of enteric E. coli bacteria present in domestic sewage, fecal coliforms (FC) are a convenient and reasonably reliable indicator of pollution and health hazard potential for a given set of disposal conditions. In this study, fecal coliform bacteria are considered an excellent indicator of fecal bacteria movement in view of the great contrast between FC densities in STE and native groundwaters.

The movement of the indicator organism away from the prototype systems was greatly restricted as seen from the means in Table 10 and the maximum densities and number of positive samples in Table 11. Five fold decreases in mean FC concentrations from STE mean of  $5.72 \times 10^{-5}$  MPN/100 ml were observed in the groundwaters nearest the prototype trenches.

The limited movement of the FC bacteria away from each of the systems is attributed to two factors, uniform distribution of STE along the bottom of the absorption trenches and the relatively deep depth of the unsaturated zone beneath the trenches. Although the effluent was not spread in the trenches with complete evenness with respect to trench bottom area as discussed earlier, the distribution was much more uniform than can be expected from gravity flow systems. The large unsaturated zones resulted, in part, from landscape position, being close to the drainage ditch and also from low water tables due to less than average rainfall during the study period. Much greater concentrations and movement of FC have been observed when STE was introduced close or into saturated zones (Caldwell, 1937; Reneau and Pettry, 1975; Reneau, 1978a; Hagedorn, et al., 1978; McCoy and Hagedorn, 1979; Stewart and Reneau 1981a). The beneficial effects of uniform distribution in the treatment and disposal of STE have been noted by Converse (1974) and Otis, et al. (1974a). Otis (1976) in review, emphasized the importance of unsaturated flow in removing bacteria from the effluent flow.

Naturally, the highest densities of FC in groundwaters should occur when the water table is at its highest level. The maximum values listed



Table 10. Mean fecal coliform densities (log normalized) in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
----- MPN/100 ml -----					
2	1.5	10 a <sup>†</sup>	1.6 a		(a) <sup>‡</sup>
	2.4	1.6	1.1		
3	1.5	5.0 a	0.0 a	0.0 a	(a)
	2.4	1.0	1.0	1.0	
4	1.5	1.5 b	0.0 a	0.0 a	1.1 (a)
	2.4	0.0	0.0	1.2	0.0

<sup>†</sup>System means with different letters are significantly different at the 5% level in accordance with Duncan's multiple range test for the 1.5-m depth and given distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

Table 11. Maximum densities observed and number of positive samples for fecal coliform bacteria from 26 samples at the 1.5-m depth and 36 samples at the 2.4-m depth for each system, depth and distance for both rows of sampling wells from May 1980 to July 1981.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
----- MPN/100 ml -----					
2	1.5	11,000 (14) <sup>†</sup>	230 (5)		
	2.4	4,600 (5)	43 (2)		
3	1.5	4,600 (10)	0 (0)	0 (0)	
	2.4	3 (1)	3 (1)	9 (2)	
4	1.5	9 (4)	0 (0)	0 (0)	9 (1)
	2.4	0 (0)	0 (0)	530 (5)	0 (0)

<sup>†</sup>Number of positive samples.

in Table 11 were observed during the winter season of 1981 but none were detected during the high water period of 1980. This occurred in spite of higher water tables during 1980 and is illustrated in Figure 12. The lack of FC detection in groundwaters at the closest sampling wells during the winter of 1980 and subsequent high levels during the 1981 wet season suggests a long range time dependency on the extent of STE-bacteria movement. The observed short range delay with lateral distance is expected and is dependent on the mass movement of FC with the movement of the upper zone of the water table. The apparently low survival of FC during the first high water table season (1980) is contributed to a more hostile environment immediately below the absorption trenches. Although members of the FC group are facultative with respect to oxygen, they are not well adapted to continuous life in the outside (aerobic) world (Frobisher, 1968). Fecal coliforms thrive in more anaerobic conditions as in a septic tank. With time and continued introduction of STE, the soil environment beneath the absorption trenches and in the lateral effluent flow path become less aerobic and subsequently less hostile to fecal bacteria. Given the conditions of this experiment, that is nonsignificant biological clogging at the soil-gravel interface, the extent of movement of large numbers of FC may be anticipated to increase with time. In the event a biological mat/crust develops at the soil-gravel interface, FC movement away from the absorption trench should be retarded as a result of bacteria entrapment as suggested by Tyler, et al. (1977) and observed by Ziebell, et al. (1974).

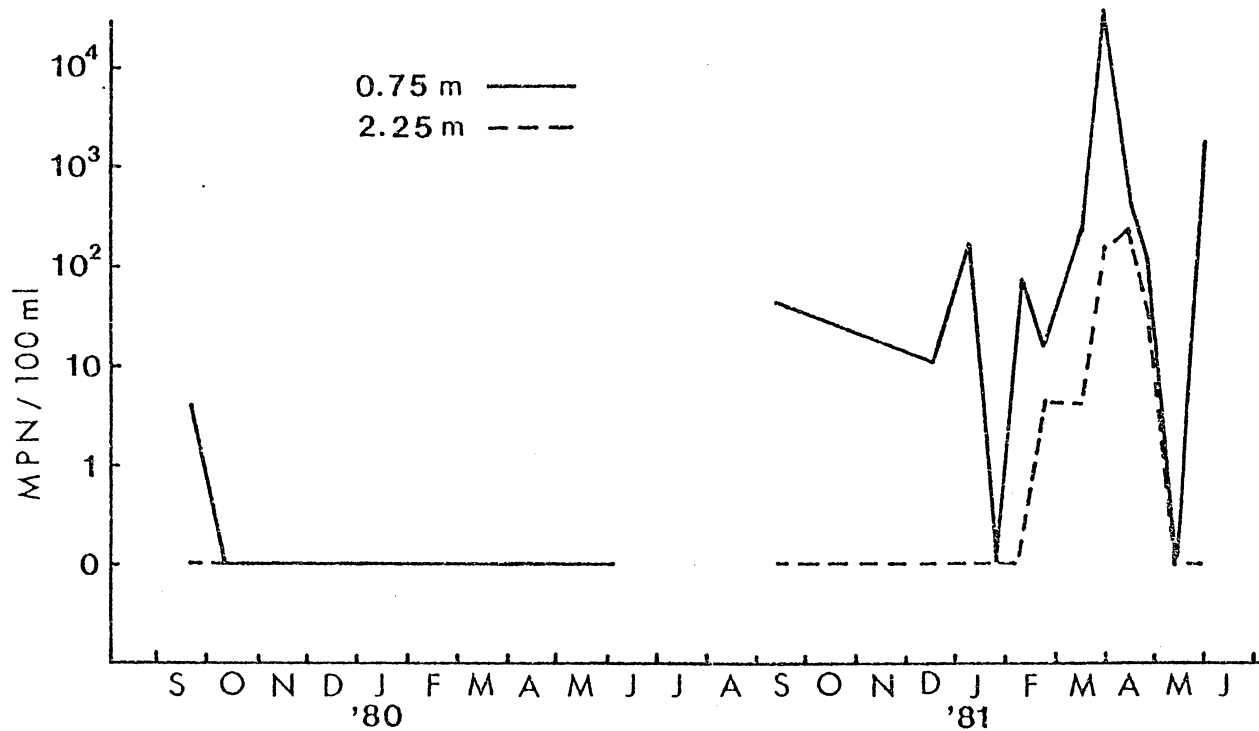


Figure 12. Fecal coliform densities for row 1, System 2 at 1.5-m depth for distances 0.75 and 2.25 m with time.

The seasonal pattern of FC movement following the first year is similar to Cl movement, high in winter and low in the summer. But unlike Cl which is stored in the unsaturated zone and later mobilized as the water table rises, bacteria are trapped/filtered and die under the unfavorable and competitive conditions when the water table is low.

The greater number of FC during high water table periods is attributed to increase downward macropore flow. As the water table rises, pores that otherwise readily accept STE are filled as a result of capillary action and allow more effluent to flow in the larger pores. Additionally and probably more important, is the increased continuity of macropores between the trench bottom and the water table created by the elimination of some of the noncontinuous and less active pore space as the water table rises therefore shortening travel time and distance. As the water table rises, the opportunity for hydrodynamic dispersion (rapid movement through macropores) is enhanced. Ziebell, et al. (1975) attributed the rapid penetration and recovery of E.coli from 60-cm column of well structured soil dosed with STE to hydrodynamic dispersion.

Once bacteria reach the water table, further movement is by macropores with decreased physical filtration taking place and velocity of water flow and survivability of the organism becoming important factors in distance of travel.

The differences in FC concentration in groundwaters near the trenches between systems as indicated statistically in Table 10 and maximum observed densities in Table 11 is believed to result primarily from the differences in depth of the unsaturated zones beneath the trench. The

mean water table level determined from periodic measurements below the trenches during the sampling period corresponding to Tables 10 and 11, were 64 cm for System 2, 80.1 cm for System 3 and 90.4 cm for System 4. These mean depths of unsaturated soil vary inversely with the mean densities of FC in groundwater at the 1.5-m depth and 0.75-m distance from each system. The significantly lower densities at System 4 may also result from the greater uniformity of STE volume/area infiltrating the trench bottom than the other systems as discussed in earlier sections. However, the differences in uniformity of STE spreading along the trench between systems is not considered to be as important as the differences in unsaturated zones beneath the trenches.

### Nitrogen

Nitrogen enters the drainfield in both inorganic and organic forms. The inorganic fraction is almost exclusively ammonium ( $\text{NH}_4$ ). The ratio of  $\text{NH}_4$ -N to organic-N in STE is approximately 3:1 (Walker, et al., 1973a and Stewart, et al., 1979). The total nitrogen loading for each of the prototype absorption trenches, based on operated STE flow, mean STE- $\text{NH}_4$  and the above ratio is equivalent to 365 kg/ha/year (326 lbs./acre/year). This is considerably larger than the 41 kg/ha/year naturally available from mineralization of soil organic matter and rainfall (Keeney, 1970). In view of the negligible background levels of nitrogen in the native groundwaters (See Table 2), any nitrogen subsequently detected in the groundwater is considered to have resulted from the introduction of STE. Nitrogen in the soil is usually in either the  $\text{NH}_4$  or  $\text{NO}_3$  form or organic fraction. Ammonium is cationic and subject to soil absorption and can

also undergo biological nitrification and be converted to anionic  $\text{NO}_3^-$ . Nitrate moves freely in the soil like Cl but is subject to denitrification resulting in the escape of nitrogenous gases. The organic nitrogen is mineralized by microbial action and is available for adsorption or nitrification. The mineralization process takes place in both aerobic and anaerobic states (Stanford and Epstein, 1974).

Nitrification takes place under aerobic conditions while denitrification mainly occurs in the absence of free oxygen provided an energy source, usually reduced carbon, is available.

Since Cl is not subject to soil absorption nor biological transformation and has been shown to exhibit similar movement patterns to  $\text{NO}_3^-$  (Cassel, et al., 1974), the  $\text{NO}_3^-$ -N/Cl ratio can indicate the occurrence of nitrification and denitrification with time at a given position or with distance. Similarly, the  $\text{NH}_4^+$ -N/Cl ratio, although not as definitive as the  $\text{NO}_3^-$ -N/Cl ratio because of soil absorption and organic mineralization, is used in describing the behavior of  $\text{NH}_4^+$ . When the  $\text{NH}_4^+$ -N/Cl ratio is decreasing,  $\text{NH}_4^+$  is being removed from soil solution either by adsorption or nitrification and when increasing,  $\text{NH}_4^+$  is being released into soil solution either by mineralization or cation exchange.

Ammonium is selectively absorbed at particular exchange sites because of its ionic size. Since  $\text{NH}_4^+$  is nearly equal in diameter to the potassium (K) ion (Rich, 1964), they compete for adsorption. Indeed, ammonium acetate is used as an extractant to determine soil-K availability (Pratt, 1965). The space limiting exchange sites have been shown to be well suited for K because of its unique fit into the interlayer

structure of interstratified clay minerals (Dowdy and Hutcheson, 1963 and Reichenbach, 1972). Ammonium should be retained/absorbed in the soil near the STE introduction site and advance only after  $\text{NH}_4$  exchange sites have been saturated. This was reported by Reneau (1979) and Stewart and Reneau (1981b).

The temporal variations in amounts of  $\text{NH}_4$  and  $\text{NO}_3$  and their accumulation relative to Cl in groundwaters near each system at the 0.75 m distance and 1.5-m depth are shown in Figures 13, 14 and 15. The lateral and vertical movement of  $\text{NH}_4$  and  $\text{NO}_3$  will be discussed by considering the means of each sampling distance and depth in Tables 12 and 13.

In considering the three systems in Figures 13, 14 and 15, some similarities and dissimilarities of N behavior and movement exist between systems.

During the period Oct. '79 to June '80 very low levels of  $\text{NH}_4$  reached the 1.5-m depth at 0.75-m distance from the prototype trenches. This results from soil adsorption of the applied  $\text{NH}_4$  during unsaturated flow. Any  $\text{NH}_4$  that did reach the 1.5-m depth during this period was apparently rapidly nitrified as aerobic conditions prevailed. The lack of any appreciable  $\text{NH}_4$  in the groundwaters at 1.5-m depth during the initial operation stages indicates no significant direct rapid pathways (macropores) existed between the trench bottoms and the water tables in the area of and to the depths of the sampling well perforations. Hydrodynamic dispersion to the 1.5-m depth did not appear to be an operative mechanism at this time. The adsorption of  $\text{NH}_4$  was taking



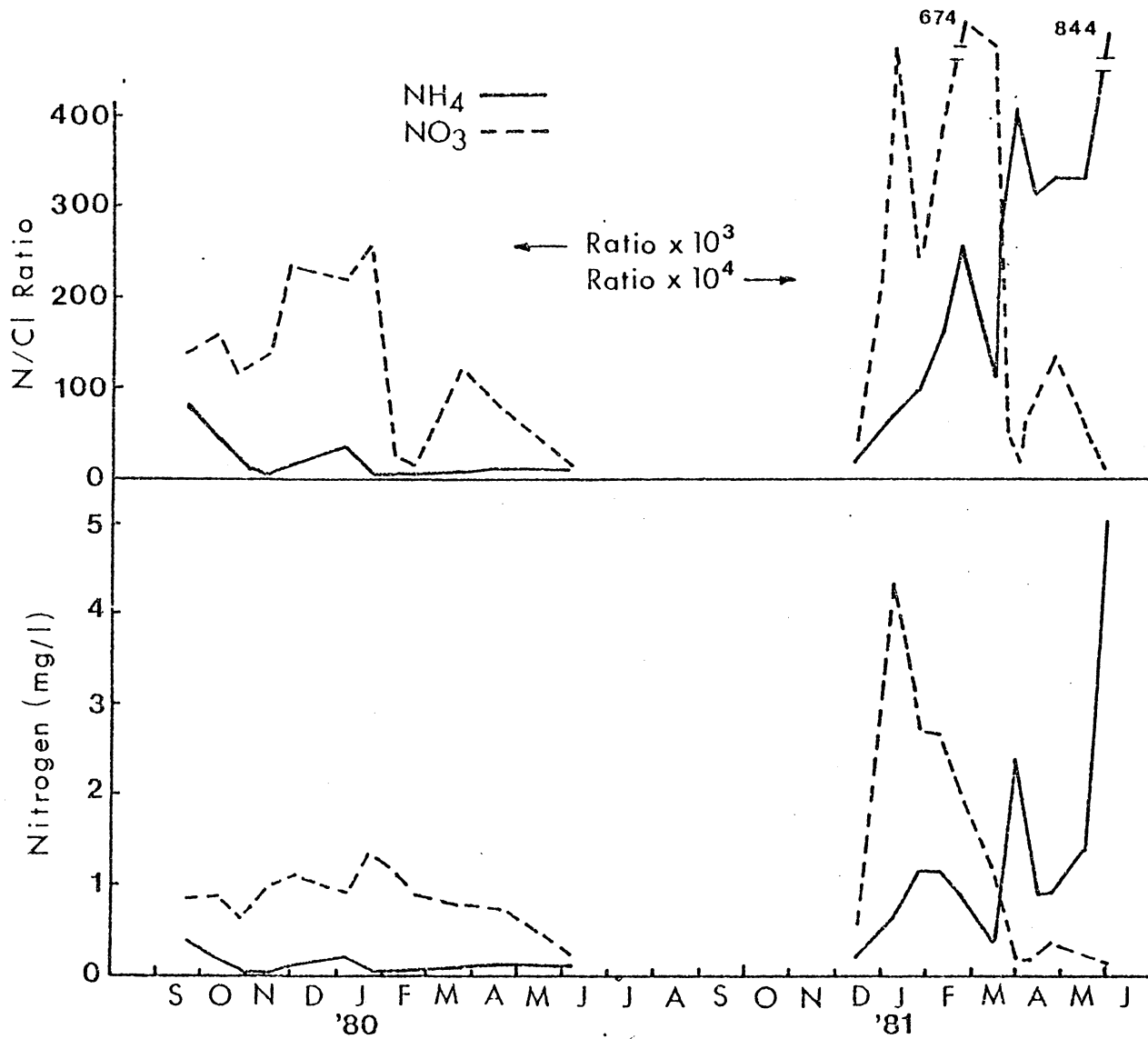


Figure 13. Ammonium-N and NO<sub>3</sub>-N concentrations and NH<sub>4</sub>-N/Cl and NO<sub>3</sub>-N/Cl ratios in ground waters at the 0.75-m distance and 1.5-m depth for System 2 with time.

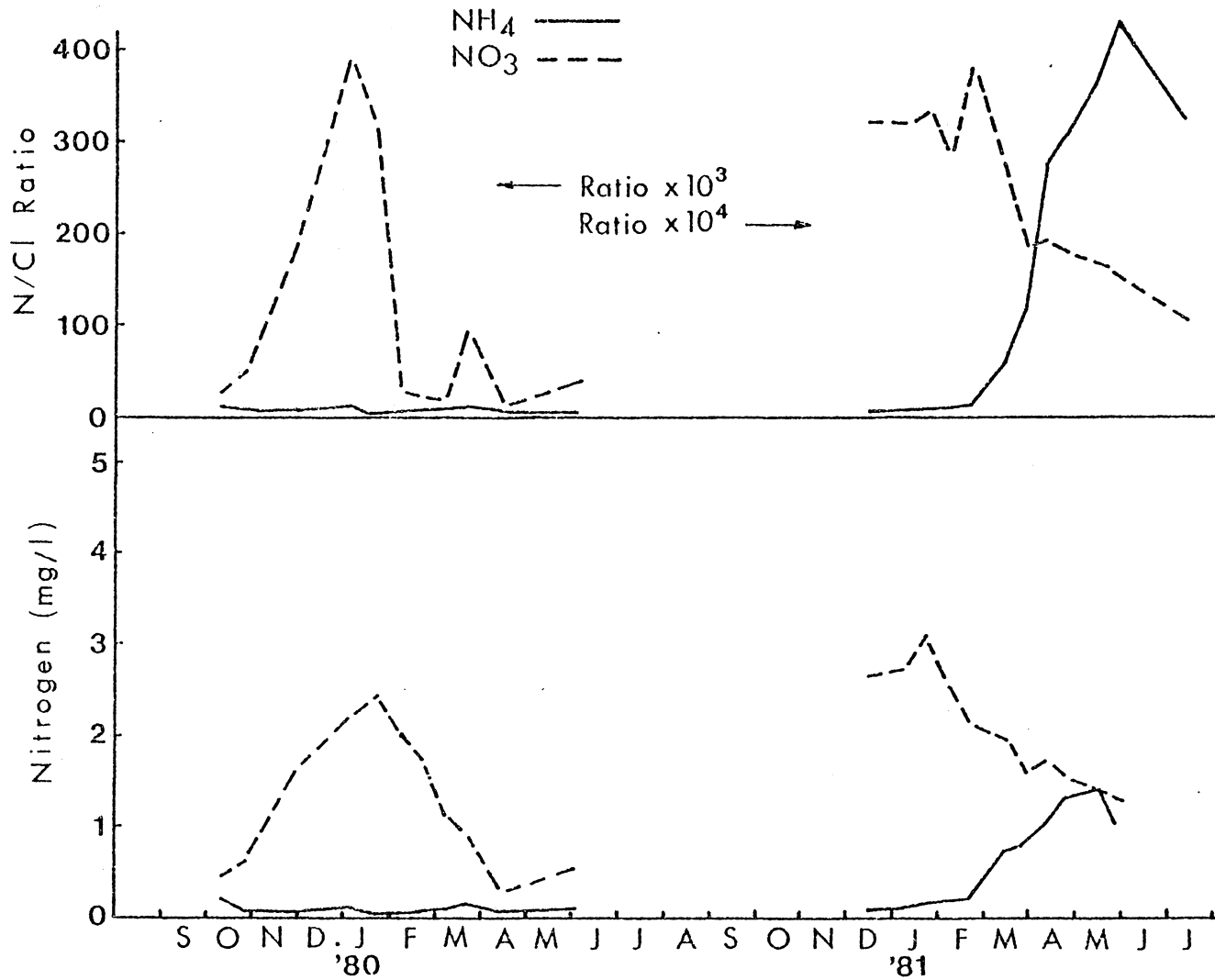


Figure 14. Ammonium-N and  $\text{NO}_3$ -N concentrations and  $\text{NH}_4$ -N/Cl and  $\text{NO}_3$ -N/Cl ratios in ground waters at the 0.75-m distance and 1.5-m depth for System 3 with time.

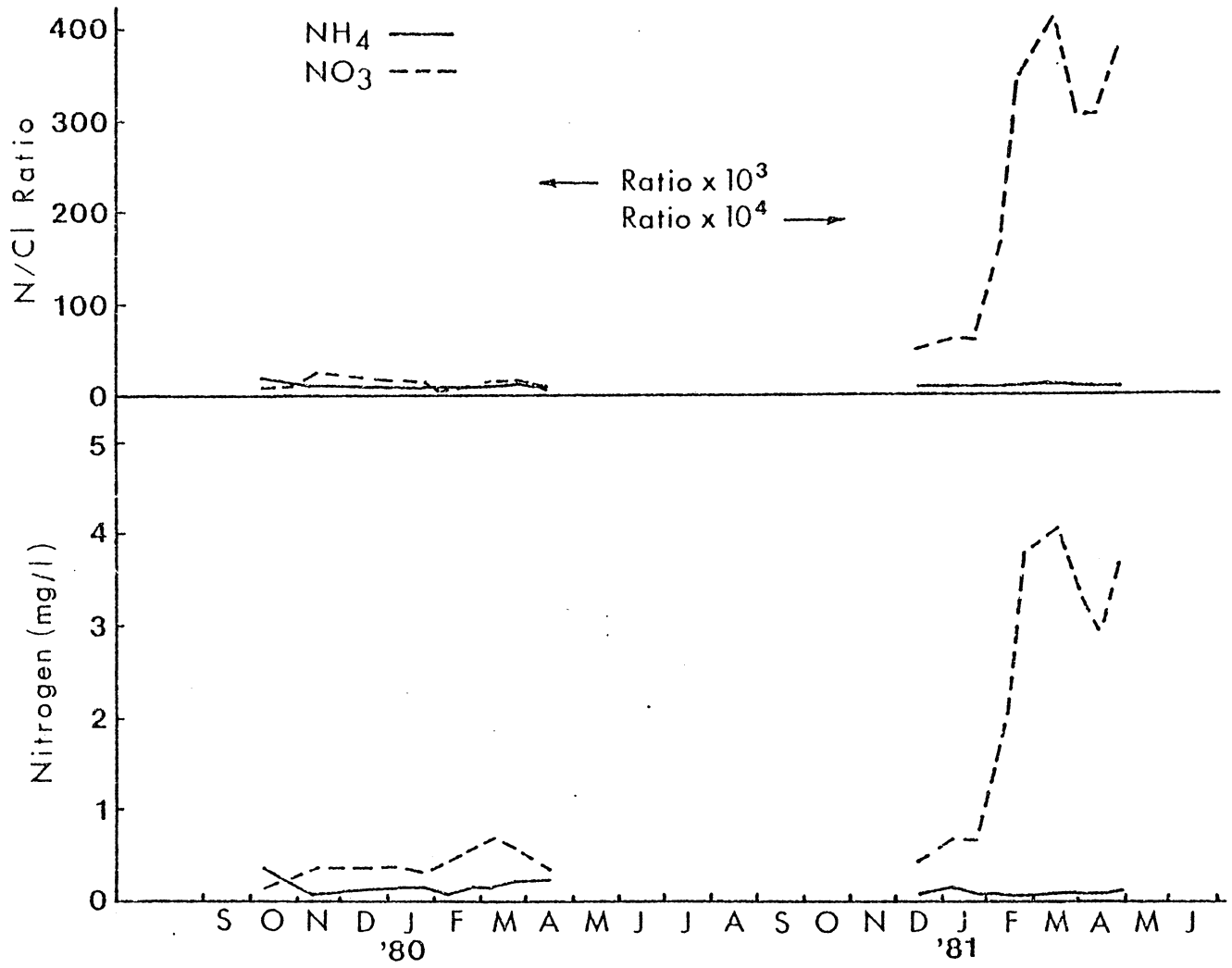


Figure 15. Ammonium-N and NO<sub>3</sub>-N concentrations and NH<sub>4</sub>-N/Cl and NO<sub>3</sub>-N/Cl ratios in ground waters at the 0.75-m distance and 1.5-m depth for System 4 with time.

Table 12. Mean  $\text{NH}_4\text{-N}$  concentrations in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
		----- ppm -----			
2	1.5	1.10 a <sup>†</sup>	0.15 a		(a) <sup>‡</sup>
	2.4	0.32	0.09		
3	1.5	0.91 ab	0.04 b	0.05 a	(a)
	2.4	0.08	0.07	0.09	
4	1.5	0.04 b	0.03 b	0.23 a	0.12 (a)
	2.4	0.06	0.06	0.16	0.19

<sup>†</sup>System means with different letters are significantly different at the 5% level (Duncan's test) for the 1.5-m depth and the given distances.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

Table 13. Mean NO<sub>3</sub>-N concentrations in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
		----- ppm -----			
2	1.5	1.32 a <sup>†</sup>	0.99 a		(a) <sup>‡</sup>
	2.4	0.06	0.07		
3	1.5	1.70 a	1.27 a	0.33 a	(b)
	2.4	0.10	0.06	0.06	
4	1.5	2.22 a	0.86 a	0.24 a	0.23 (b)
	2.4	0.09	0.09	0.12	0.10

<sup>†</sup>System means with different letters are significantly different at the 5% level in accordance with Duncan's multiple range but for the 1.5-m depth at the given distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

place at a faster rate than the movement of STE.

Nitrification was an active process during the fall of 1979 at the sampling points closest to Systems 2 and 3 as indicated by the increased  $\text{NO}_3\text{-N/Cl}$  ratio in Figures 13 and 14. The nitrification period was followed by denitrification as the water table remained in its highest position during the winter. Conditions sufficiently anaerobic for denitrification were created by saturation of the sampling zone and higher biological oxygen demand resulting from lesser aerobic decomposition of STE-organic components because of the reduced travel time to the water table and lower winter temperatures. Nitrification also occurs below the trenches for Systems 2 and 3 during the spring but it is only after continued and extended unsaturated (aerobic) conditions that  $\text{NO}_3$  levels peak as shown in the early winter of 1981 for Systems 2 and 3. Reneau (1979) observed the highest  $\text{NO}_3$  concentrations in groundwaters near STE disposal fields as the water table rose at the beginning of the wet season.

The nitrification-denitrification process was not active for System 4 at the 1.5-m depth during the year 1980 (See Figure 15). This is indicated by the relatively flat  $\text{NH}_4\text{-N/Cl}$  and  $\text{NO}_3\text{-N/Cl}$  curves during this period. Apparently  $\text{NH}_4$  was not reaching the 1.5-m depth to be subsequently nitrified. Neither was large quantities of  $\text{NO}_3$  being moved downward by rainfall as only low levels, < 1 ppm, were present during the winter-spring of 1980. The low levels of  $\text{NH}_4$  and  $\text{NO}_3$  in the saturated zone beneath System 4 as compared to Systems 2 and 3 is attributed to the greater absorption of  $\text{NH}_4$  resulting from the greater depth of the

unsaturated zone and the more uniform distribution of STE at System 4.

It is only after 18 months that appreciable levels of  $\text{NO}_3^-$  appear below System 4. This occurred in the late winter-early spring of 1981 at which time limited denitrification was taking place. While nitrification was occurring below System 4, denitrification was taking place below Systems 2 and 3 with a simultaneous increase in the levels of  $\text{NH}_4^+$ . This indicates septic-like conditions at the 1.5-m depth below Systems 2 and 3.

Denitrification and N movement were distinctly different for the three systems as shown by the  $\text{NO}_3^-$ -N/Cl curves in Figures 13, 14, and 15. Nitrate levels were decreased sharply below System 2 during the winter of 1981 and more gradual below System 3 while only just beginning to decrease 1-2 months later at System 4. The varying intensity of denitrification is apparently related to differences in available energy for the denitrifying organism. Stewart, et al. (1979) and Sikora and Keeney (1974) reported that natural soil organic matter (even peats and organic horizons) is not an ample supply of ready available energy to sustain denitrification at the rate of  $\text{NO}_3^-$  availability resulting from nitrification of STE- $\text{NH}_4^+$ . Poorly drained soils were observed by Gambrell, et al. (1975) to afford greater denitrification of residual  $\text{NO}_3^-$  from fertilization than moderately well drained soils. Initial high denitrification rates were restored only after the addition of an energy source following soil wetting and drying cycles in a study by Patrick and Wyatt (1964). Based on the above studies, for other than initial and short-lived denitrification, the energy source for sustained removal of  $\text{NO}_3^-$  by denitrify-

ing bacteria comes from the organic matter in the STE.

The differences in denitrification in the saturated zone beneath the three systems can be attributed to varying energy sources resulting from the degree of STE-organic matter oxidation. System 4 had the most extensive unsaturated zone which resulted in more complete oxidation of STE-organics and lower denitrification. System 2 was just the opposite and System 3 intermediate. Walker, et al. (1973b) reported little or no denitrification taking place in deep saturated zones underlying a STE drainfield but complete nitrification of STE-N which supports this study.

The increased levels of  $\text{NH}_4$  in the saturated zone at System 2 (the 0.75-m distance and 1.5-m depth) are statistically different from Systems 3 and 4 (See Table 12). The mean  $\text{NO}_3$  concentrations in groundwaters below the systems were not statistically different. However, the trend of increasing  $\text{NO}_3$  levels below System 4 is clear and the differences should increase with time as more  $\text{NH}_4$  becomes available for nitrification.

Lateral movement of  $\text{NH}_4$  as noted in Table 12 decreases with distance from the trenches. This is in agreement with information (Reneau, 1979) showing that decreases in  $\text{NH}_4$  concentrations in groundwaters could be expected with increased distance from STE disposal fields. No significant differences in  $\text{NH}_4$  levels were present at the 0.75-m distances and 1.5-m depths between Systems 2 and 3 but were statistically different at the same depth and the 2.25-m distance. The greater decrease of  $\text{NH}_4$  levels with distance at System 3 probably results from greater absorption of  $\text{NH}_4$  with decreased travel time as the water table gradient is



not as steep as at System 2.

Nitrates which tended to accumulate in the zone beneath the trenches decreased with distance for each system while the  $\text{NO}_3\text{-N/Cl}$  ratios remained relatively constant (Tables 13, 14). This situation indicates limited denitrification with distance and that the decrease was due to dilution. The zone of septic influence that increases denitrification is apparently restricted to the immediate absorption trench area allowing the  $\text{NO}_3$  to move freely with the upper layer of the saturated zone. Less travel time and lowered dilution account for the significant differences in  $\text{NO}_3$  concentrations between System 2 and Systems 3 and 4 at the 1.5-m depth at the edge of the drainage ditch.

Nitrate levels as well as  $\text{NO}_3\text{-N/Cl}$  ratios decreased with depth at all sampling locations. The 2.4-m depth was constantly saturated rendering an unsuitable environment for nitrification of any mineralized or percolated  $\text{NH}_4$ . Apparently enough soluble carbon from natural sources was available to denitrify the small quantities of  $\text{NO}_3$  that did penetrate to deeper depths.

In summary, N behavior and movement in the zone immediately below the prototype trenches was influenced by the depth of the water table. Variations in N behavior were observed even though the mean distances from the trench bottoms to the water table varied only by 26.4 cm. Nitrate accumulation was largest under System 4 which had the most extensive unsaturated zone. Ammonium accumulation and intensity of denitrification were the greatest underneath System 2 which had the shallowest mean water table. Limited denitrification at System 4

Table 14. Mean  $\text{NO}_3\text{-N/Cl}$  ratios ( $\times 10^3$ ) in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
2	1.5	24.4 a <sup>†</sup>	26.1 a		(a) <sup>‡</sup>
	2.4	3.7	4.2		
3	1.5	27.7 a	37.1 a	22.5 a	(a)
	2.4	8.6	7.1	6.2	
4	1.5	21.8 a	15.0 a	12.1 a	16.2 (a)
	2.4	6.3	6.0	10.3	5.1

<sup>†</sup>System means with different letters are significantly different at the 5% level in accordance with Duncan's multiple range but for the 1.5-m depth at the given distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

suggested that N losses resulting from water table fluctuations observed at Systems 2 and 3 and in other studies, Reneau (1979) and Stewart and Reneau (1981b) may not be as great when mean unsaturated depth zones are  $\geq 90$  cm. Nitrates moved laterally in the upper zone of the water table and decreased with distance as a result of dilution and dispersion. Denitrification prevented any downward  $\text{NO}_3$  movement below the water table. Lateral movement of  $\text{NH}_4$  was limited but advanced towards the drainage ditch in greater concentrations at System 2 where the water table gradient was steepest.

In groundwater at the edge of the drainage ditch, there were no significant differences in  $\text{NH}_4$  levels between the systems. However, the trend indicated that with time System 2, the absorption trench nearest the drainage ditch would supply appreciable amounts of  $\text{NH}_4$  to the drainage waters.

Nitrate level in groundwaters at the edge of the drainage ditch from System 2 were significantly higher than from Systems 3 and 4. The increased  $\text{NO}_3$  levels occurred in spite of System 2 having the smallest unsaturated zone. In the event of an extended unsaturated zone at 3-m distance from the drainage ditch which would be the case if the ground surface had been more gently sloped, undesirable levels of  $\text{NO}_3$  would have entered the drainage waters because of limited denitrification as experienced at System 4 and more rapid movement of water the upper saturated zone.

The overall low levels of N ( $< 5$  mg/l) in groundwaters beneath any of the systems is due in part to short life of the system not yet in

equilibrium with the environment and uniform distribution of STE along the absorption trench bottoms.

### Phosphorus

The P content averaged 0.188 ppm in household waters and 13.2 ppm in STE. The mean background levels in groundwaters was less than .001 ppm. The P loading rate to the bottom of the absorption trenches was 121 kg/ha/year (108 lbs./acre/year).

Phosphorus movement is shown by the means of each sampling point listed in Table 15. Only groundwaters adjacent to System 2 had a mean P concentration > 10 ppb. System 2 had the shallowest mean depth to the water table and subsequently the least amount of P sorption potential of the three systems. The absence of P in groundwaters adjacent to System 4 as well as the low mean levels at Systems 2 and 3 compared to the P loading rate are attributed to soil sorption which includes both direct adsorption and precipitation. Carlile, et al. (1981) and Harkin (1979) attest to limited P movement from STE disposal system as a result of P sorption. The P sorption process is enhanced by the uniform distribution of STE within the trench and the large amount of aluminum and iron oxides present at the experimental site.

These systems are relative young and are not yet in equilibrium with the environment. With time, the P levels in the groundwaters near and away from the trenches will probably increase. Reneau (1979) and Stewart and Reneau (1981b) reported P concentrations as high as 6.88 and 5.4 ppm in groundwaters near older (6 and 15 years) fields which were also shown to have very nonuniform distribution of STE. In the

Table 15. Mean P concentrations in groundwaters with depth and distance from the prototype systems.

System	Depth (m)	Distance (m)			
		0.75	2.25	5.25	20.25
		----- ppm -----			
2	1.5	0.018 a <sup>†</sup>	0.001 a		(a) <sup>‡</sup>
	2.4	0.005	0.001		
3	1.5	0.001 b	0.003 a	0.003 a	(a)
	2.4	0.001	0.004	0.000	
4	1.5	0.000 b	0.001 a	0.000 a	0.000 (a)
	2.4	0.007	0.000	0.005	0.001

<sup>†</sup>System means with different letters are significantly different at the 5% level for the 1.5-m depth and given distance.

<sup>‡</sup>System means with different letters are significantly different at the 5% level at the 1.5-m depth at the edge of the drainage ditch.

above two studies, P concentration in groundwaters > 1 ppm were limited to 1.5 and 7 meters but P was observed to move extended distances through saturated macropores under the influence of a strong hydraulic gradient without regard to the soil's P sorption capacity.

An additional process, the regeneration of P adsorption sites by alternate drying and wetting suggested by Sawhney and Hill (1975) may add to the soil capacity to restrict P movement as the water table fluctuates between a winter high and a summer low.

The results of this study relative to P movement supports Sikora and Corey (1975) conclusion that only in soils with little sorption capacity would the pollution potential of P from STE be considerable.

#### Effect of Drainfields On The Shape of Water Tables

For this study the volume of STE introduced through the widely spread prototype trenches was considered insignificant compared to the volume of rainfall for the drainage area. The ratio of STE to rainfall for each system was 0.001 and thus the presence of the four systems should have little effect on the shape of the water table. A full size system consistent with soil evaluation, percolation tests and residence size (Virginia State Board of Health, 1982) would result in a ratio of STE to rainfall thirty times greater than the above ratio. Also the STE would be concentrated in 30 x 3.6 m (108 m<sup>2</sup>) area with a loading rate of 1.5 cm/day within the drainfield area. Thus the need arises to compare the relationship between a full scaled disposal system and artificial drainage since the additional volume of STE would affect the shape of the water table drawdown curve.

Before considering the effect of a large volume of effluent on water table shape an equation of flow to describe the observed shape must be made available. An approximate method for matching the observed shape of the water table is sought in view of the uncertainty of soil parameters and the lack of detailed spatial soil hydraulic conductivities. In support of approximate methods, Childs (1969) stated approximations are often preferred without detailed information. van Schilfgaarde (1974) attested to the greater accuracy of drainage formulas more than the evaluation of pertinent parameters including both climatic and soil factors.

The classic approach to approximation of an equation of flow uses the Dupuit-Forchheimer (D-F) theory which was first developed by Dupuit in 1853 and used extensively by Forchheimer (Kirkham and Powers, 1972). Childs (1969) extended the use of the D-F equation for local precipitation and equaled spaced parallel ditches that extended downward to a shallow impermeable barrier underlying homogeneous soil in two dimensions to the case of an infinite deep soil with rectangular shaped drains in three dimensions. The resulting equation from Childs (1969) is

$$(z_m^2 - z_o^2) \frac{1}{L_1^2} + \frac{1}{L_2^2} = q/K \quad (1)$$

where  $z_m$  = maximum height of water table,  $z_o$  = height of water in ditch,  $L_1$  and  $L_2$  are one-half the separation distances of the orthogonal sets of parallel ditches,  $q$  = rainfall rate, and  $K$  = saturated soil hydraulic conductivity (assumed isotropic). If  $L_1 = L_2 = L$  and  $z_o$  is assumed to

be zero, then Eq. (1) takes the form:

$$z_m^2 \left( \frac{2}{L^2} \right) = q/K \quad (2)$$

Eq. 1 differs only by the multiplier "2" on the left hand side from Childs (1969) equation

$$z_m/L^2 = q/K \quad (3)$$

which gives the maximum rise in the water table for parallel drains placed  $L$  distance apart. Eq. 3 evolved from a Childs (1969) equation for the shape of the water table

$$(z^2 - z_0) / (2Lx - x^2) = q/K. \quad (4)$$

Because of the symmetry of a square-shaped system of ditches when  $L_1 = L_2$  in Eq. 1, the multiplier "2" can be logically included in Eq. 4 to render the shape of the water table for the special three dimensional case. Setting  $z_0 = 0$ , and rearranging,

$$z^2 / (2Lx - x^2) = .5 q/K \quad (5)$$

should approximate the shape of the water table for infinitely deep soil drained by a square-shaped drainage system.

The study area has a perimeter drainage ditch and is very nearly a 120 x 120-m square lot. The soil has a sandy sublayer beginning at the 103-cm depth and extends to a minimum of 300 cm. The sandy substrata is assumed to be infinitely deep.

The effective  $K$  ( $K_e$ ) for the system is initially unknown but can



be determined from Eq. 5 using the observed height at the center of the drained area.

To test Eq. 5 for description of the water table, the bottom of the drainage ditch was taken as the zero datum level and the 2-year means of daily water level elevations during the high water table period January to April at 4.5, 29.1, and 60-m horizontal distance from the ditch were used as the observed heights.

Using the average daily rainfall, the  $K_e$  was determined from Eq. 5 to be 748 cm/day. The calculated  $K_e$  was considerably higher than the horizontal  $K$  (173.4 cm/day) determined by methods of King (1974) for the composite of soil horizons above the drainage ditch invert. The large  $K_e$  falls within the range of conductivities for sand (Bouma, et al. 1972) and indicates that the major zone of water movement is in the sand substrata. Rogowski, et al. (1966) noted the separation distance of 60 m between drainage ditches that penetrated sand lenses were just as effective as a 30-m spacing. Raadsma (1974) states the common practice in Europe is to fill in drainage ditches used for initial drainage that are located within the area outlined by boundary ditches (300 to 400 m apart) that penetrate past clay layers into coarse sand. Such particles and observations suggest rapid water movement in large volumes in sandy sublayers.

The inclusion of the calculated  $K_e$  in Eq. 5 produced water table heights of 27.9 cm at 4.5 m, 62.7 cm at 29.1 m and 74.5 cm at 60 m. The calculated heights compare favorably with the observed heights of 27 cm (4.5 m), 51 cm (29.1 m), and 74.5 cm (60 m) with a maximum vari-

ance of 18.7 percent.

These results were considered reasonable for the given parameters.

It was also necessary to develop an equation (similar to Eq. 5) that also accounts for the addition of a drainfield as diagrammed in Figure 16. Using D-F assumptions, the flow per unit width ( $Q_x$ ) through the cross section bounded by the elevation of the ditch bottom and the water table surface following Darcy's Law is:

$$Q_x = K Z \frac{dz}{dx} \quad (6)$$

where  $K$  = saturated hydraulic conductivity and  $Z$  = height of water table above the ditch invert. Also, considering mass balance across the water table surface

$$Q_x = q_r (L-x) + q_d u_{(2)} [ Ld_2 - Ld_1 + u_{(1)} (Ld_1 - x) ]$$

where the unit step functions

$$u_{(1)} = \begin{cases} 0, & x < Ld_1 \\ 1, & x > Ld_1 \end{cases} \quad \text{and}$$

$$u_{(2)} = \begin{cases} 0, & x > Ld_2 \\ 1, & x < Ld_2 \end{cases} \quad (7)$$

$q_r$  = rainfall rate,

$q_d$  = drainfield loading rate,

$L$  = one half of the ditch spacing,

$Ld_1$  = distance from ditch to near edge of drainfield and

$Ld_2$  = distance from ditch to far edge of drainfield.

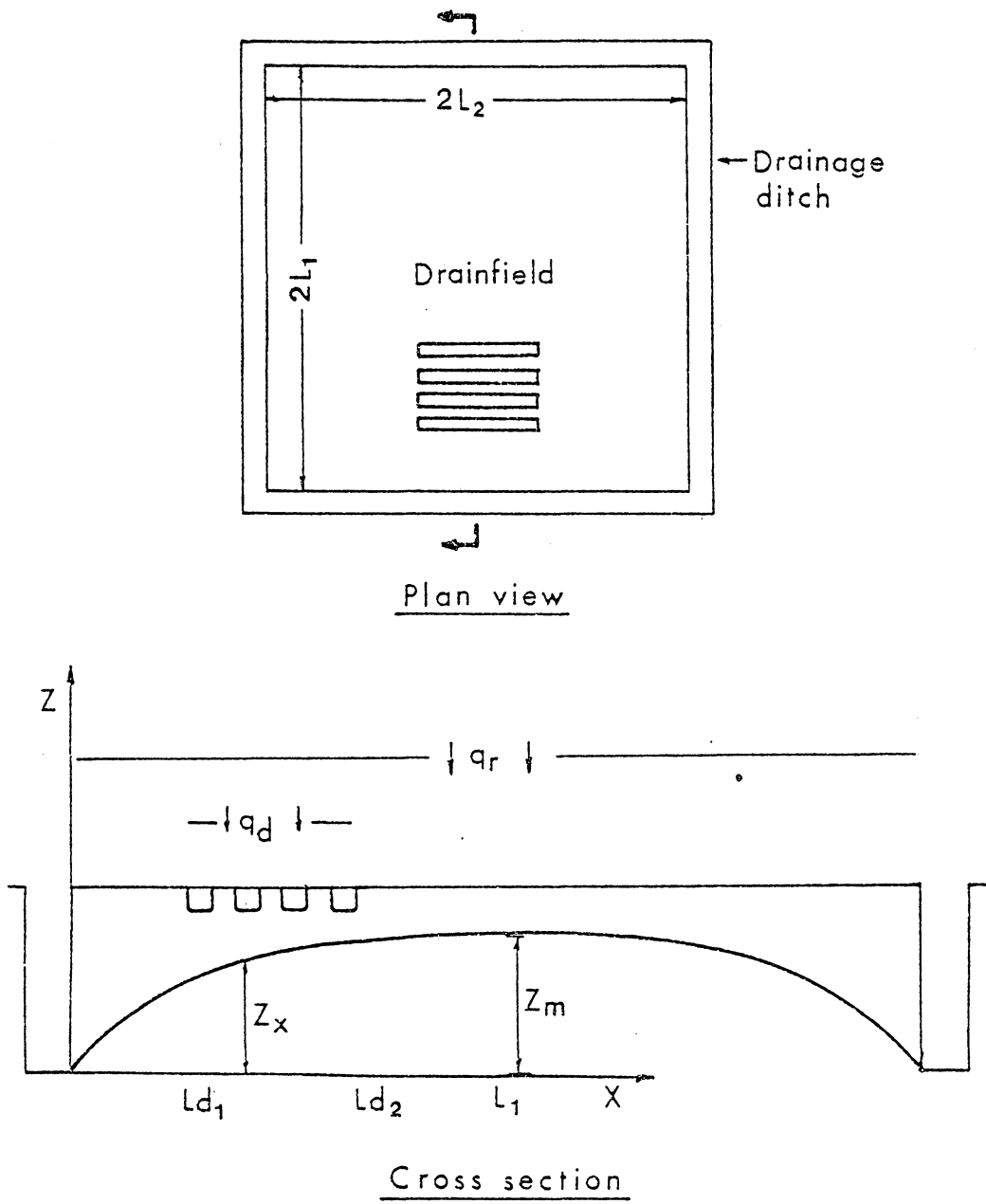


Figure 16. Plan view and cross section of local rainfall and septic tank effluent flow to drainage ditches in an infinitely deep soil.

The boundary conditions to be imposed are that the maximum height of the water table will be in the center of the drained area and the water level in drainage ditches is zero. These conditions are represented by

$$\begin{aligned} z_{(0)} &= 0 \\ z_{(L)} &= z_m . \end{aligned} \quad (8)$$

Equating 6 and 7 and integrating between the limits set by 8 produces

$$z_m^2 = \frac{q_r}{K} L^2 + \frac{q_d}{K} (Ld_2^2 - Ld_1^2) . \quad (9)$$

Letting  $Ld_2 = Ld + Lw$  where  $Lw =$  width of drainfield and  $Ld = Ld_1 =$  distance from ditch to edge of drainfield, Eq. 9 becomes

$$z_m^2 = \frac{q_r}{K} L^2 + \frac{q_d}{K} (2Ld Lw + Lw^2) . \quad (10)$$

Eq. 10 applies to a drainage system with a single set of parallel drains.

To include the three dimensional effect of rectangular drains and also not unduly complicate the mathematics, the rectangular drainage system is converted to a two dimensional system by considering the average flow path of the drains as suggested by Childs (1969). The three dimensional drainfield was converted to two dimensions using similar procedures. It was assumed that the distance from the drainage facility to the edge of the drainfield ( $Ld$ ) remains constant (before and after conversion) and the longer sides of both the drainfield and drainage ditches are parallel. Upon conversion from three dimensions to two

dimensions Eq. 10 becomes

$$Z_m^2 = \frac{q_r}{K} \frac{L_1^2 L_2^2}{L_1^2 + L_2^2} + \frac{q_d}{K}$$

$$\left[ 2Ld \left( \frac{Lw^2 Lf^2}{Lw^2 + Lf^2} \right)^{1/2} + \frac{Lw^2 Lf^2}{Lw^2 + Lf^2} \right] \quad (11)$$

$L_f$  and  $L_w$  are mutually dependent and determined by the area required for the drainfield according to regulations. The maximum loading rate of STE is set by regulation and based on soil evaluation (Va. State Board of Health, 1982). The actual STE loading rate ( $q_d$ ) is taken as the daily household flow divided by the area of the drainfield. Local precipitation ( $q_r$ ) is taken as the average daily rainfall. Saturated hydraulic conductivity is either determined from observed water table heights using Eq. 5 with the correct multiplier or estimated from experience. The three remaining parameters,  $L_d$ ,  $L_f$  and  $L_w$ , are variables and effect the maximum height of the water table.

From Eq. 11, one can see that  $Z_m$  will be at a minimum when  $L_w$  is minimized. This is logical because the shortening of  $L_w$  will lengthen the trenches for a given volume of STE and take advantage of the drainage ditches running perpendicular to the absorption trenches and also the STE will be introduced closer to the parallel ditches. The distance from the ditch to edge of the drainfield ( $L_d$ ) is set to assure adequate purification of STE as it flows toward the ditch.

A decrease in  $Z_m$  when the length of the drainfield ( $L_f$ ) increases is true regardless of a constant or variable overall drainfield

geometry when the incoming volume of STE is constant for all areas. This is illustrated in Figure 17 where the  $Z_m$  values for  $f = 90$  m are lower than those for  $f = 30$  m irrespective of concentrating or widely distributing the effluent within the same drainage area given by  $L = 60$  m. The difference between  $Z_m$  when  $f = 90$  and  $f = 30$  when concentrating effluent near the drainage ditch is only 3.8 cm at  $L_d = 3$  m and 11.8 cm at  $L_d = 21$  m and varied nearly linearly between 3 and 21 m. The gain of unsaturated flow in this case would have to be balanced against the additional expense in installing such a long and narrow system and staying within the guidelines of regulations.

For fitting a system to an existing drainage system, the optimum placement of the drainfield to result in the lowest rise in the water table would involve a decision of concentrating near one ditch or placing the system in the center of the drainage system. Placement in the center would reduce the amount of effluent draining to a ditch by one-half. The extreme points of "B" curves in Figure 17 represent the placement of a drainfield of the same dimensions as an "A" drainfield midway of the drainage system. In all cases in Figure 17, the "A" curve is below the extreme end points of the "B" curves. These results indicate that it is more effective ( a lower water table mound) to place the drainfield near the ditch (0 to 21 m) rather than in the middle of the drainage system. Distributing the same volume of effluent over a large area centered between the ditches would of course lower  $Z_m$  as shown by the "B" curves in Figure 17 but only slightly (a maximum of 9 cm at  $L_d = 21$  m for the given situations) and probably would not merit

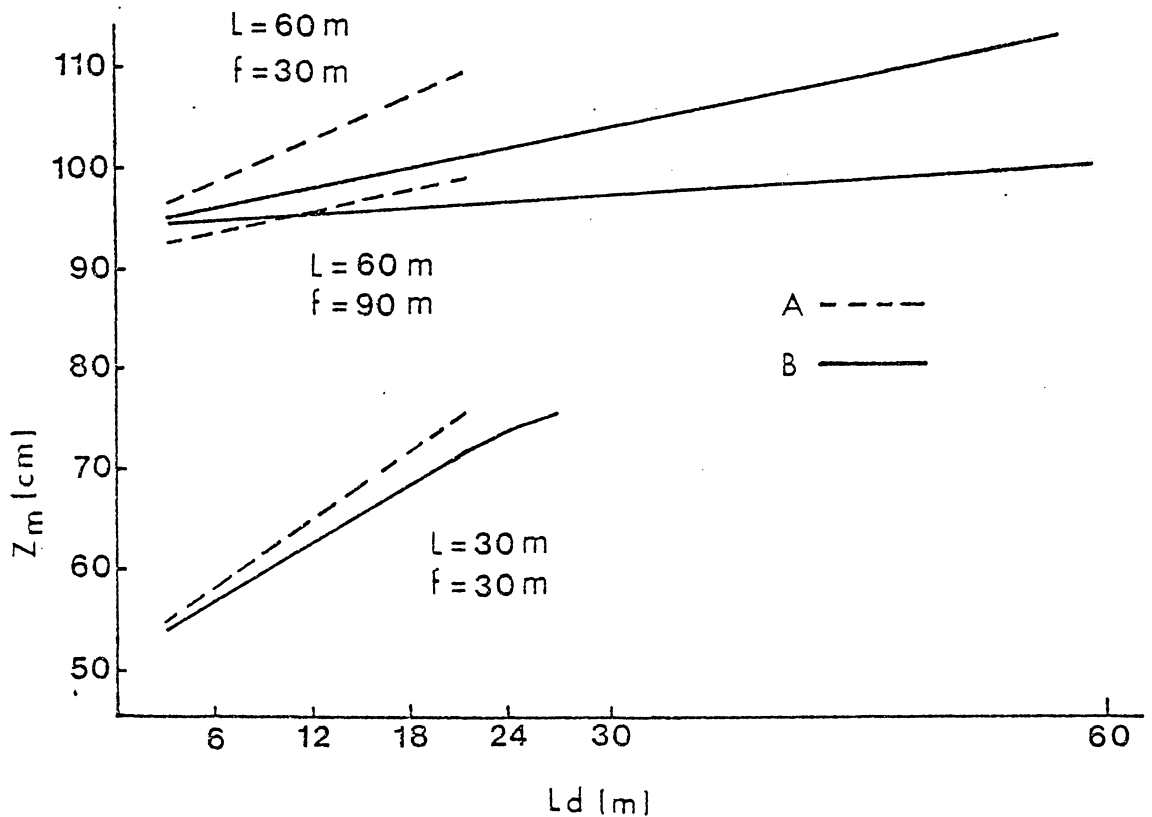
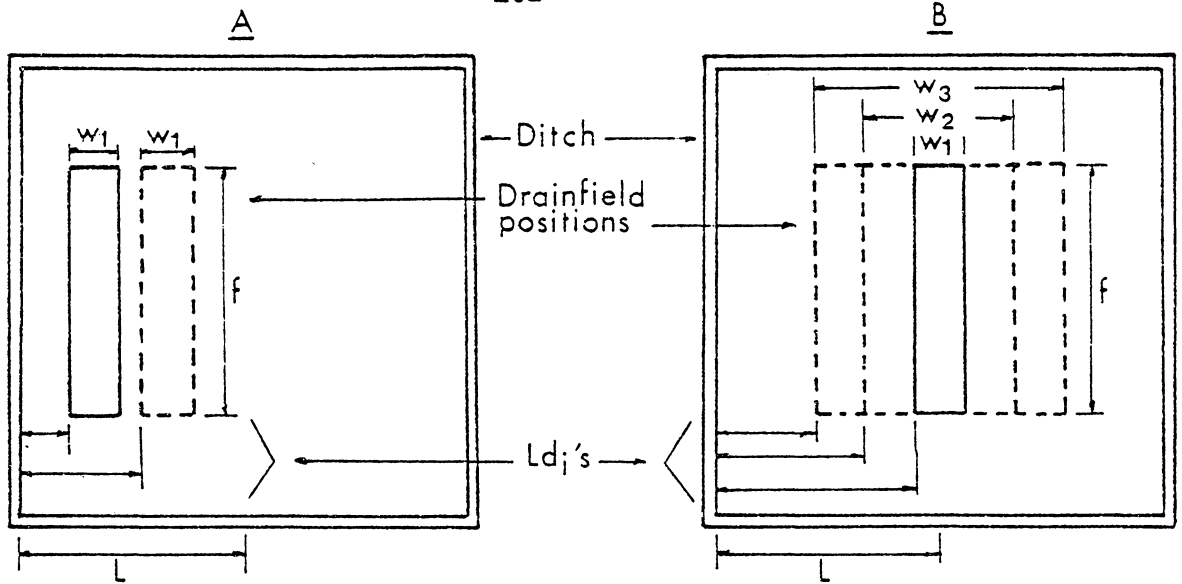


Figure 17. The effect of concentrating (A above) near the drainage ditch and spreading (B above) from the center of drainage system 1800 liter/day of STE on the maximum rise of  $Z_m$  with distance from ditch to drainfield ( $L_d$ ) for drainfield lengths ( $f$ ) of 30 and 90 m and ditch half spacing ( $L$ ) of 60 and 30 m.

the additional expense of a larger drainfield.

When sizing both the drainfield and the drainage system, the optimum conditions (lowest water table mound) result from a long narrow shaped drainfield consistent with soil properties, STE flow and site boundaries surrounded by a orthogonal sets drainage ditches. The separation distance between the drainfield and the drainage ditches would be constant and set to afford attenuation of potential pollutants thus minimizing public health risks.



## SUMMARY AND CONCLUSIONS

Several practical aspects of disposing home sewage near drainage ditches were realized from collection and analysis of data from this study. These are outlined below.

1. Single variable depth sampling wells are needed to monitor groundwater constituents. Variable depth wells instead of a nest of fixed-depth wells will afford more accurate sampling of the upper zone of the water table regardless of the depth of the water table surface.
2. The use of gate valves to control the flow of STE to each absorption trench was very satisfactory following piping alterations. Initially, clogging occurred in the valves. Equalizing the flow to each trench as determined by the standing head at the distal end of the distribution pipe was accomplished only after installing an additional gate valve in the main line and reducing the size of gate valves at each trench.
3. The lack of adequate filter material between the trench gravel and backfill resulted in the sealing of the gravel-distribution pipe interface in the trench nearest the drainage ditch. This sealing caused effluent to surface during pumping. The movement and deposition of fine soil particles at the interface was caused by saturated conditions resulting from surface runoff.
4. Infiltration of STE into the soil beneath the trenches was

less than uniform in three of the experimental trenches. However, STE infiltration was more uniform than could be expected from conventional gravity flow distribution. The use of dividers in the trench are suggested to prevent effluent transversing the trench length thereby providing more uniform infiltration.

5. The high SAR's (18-45) of the STE had no significant deleterious effects on STE infiltration as no effluent ponding in the trenches was observed. The lack of effect is attributed to the predominance of nonswelling kaolinitic type clay and presence of large amounts of sesquioxides.
6. The combination of excessive sloping of the ground surface and the gentle slope of the water table resulted in an increasing depth of unsaturated zones with distance away from the drainage ditch. Ground surface grades greater than 0.2 percent are not recommended when STE disposal fields are included.
7. Water table fluctuations were dampened in the vicinity of drainage ditch.
8. The predominant pathway of effluent movement in the saturated zone was laterally towards the drainage ditch and in the upper zone of the water table. The velocity of movement depended on the water table gradient. Vertical movement below the water was limited.

9. The presence and accumulation of fecal coliforms and chemical components of STE in the groundwater beneath the disposal trenches was inversely proportional to the mean distance between the trench bottoms and the water table. Significant differences, at the 5 percent level, for FC and chemical concentrations existed between systems in spite of small differences ( $\leq 26$  cm) in the mean unsaturated depths.
10. Fecal coliform movement to the underlying saturated zones was retarded during the first year of operation. The density of FC increased with continued introduction of STE which extended the septic zone and enhanced conditions for FC survival. Lateral movement of FC  $> 200$  MPN/100 ml at the 2.25-m distance was limited to the system closest to the water table and the shallowest mean water table depth. With time, FC densities can be expected to increase in the groundwaters beneath absorption trenches.
11. Ammonium movement was restricted because of soil absorption and nitrification. Mean  $\text{NH}_4\text{-N}$  levels  $> 1.0$  ppm in groundwater was detected only at the system with the least unsaturated flow.
12. Nitrification was observed in zone immediately beneath the absorption trenches during low water table periods. The nitrates were subsequently partially denitrified as the water table levels increased at Systems 2 and 3, the systems with the shallowest and intermediate mean water table depths. Limited denitrification occurred beneath System 4, the system with the

deepest mean water table. The lack of denitrification at System 4 suggests  $\text{NO}_3$  may accumulate when the mean depth of unsaturated flow is  $\geq 90$  cm.

13. The reduction of  $\text{NO}_3$  concentrations in ground water with distance away from the absorption trenches is attributed to dilution rather than denitrification.
14. Based on considerations of depth of unsaturated flow, water table stability, the current practiced setback distance of 21 m between drainage facilities and drainfields is considered counterproductive. A separation distance of 9-10 m would provide 1) better treatment of STE, 2) adequate buffer for significant rainfall events and 3) a high dilution potential. The above recommendation is conditional and valid only in situations where the artificial drainage system is adequate to maintain the water table levels below the trench bottoms, pressure distribution is used and the ground surface is not excessively sloped.

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## APPENDIX

- I. Tables of Data
- II. Soil Description

## Appendix I - Tables of Data

Appendix Table 1. Fecal coliform densities, pH, EC and concentrations of P, Cl, NH<sub>4</sub>, NO<sub>3</sub> in household waters, septic tank effluent and native groundwaters.

Yr	Mo	Day	Source	FC MPN/100ml	pH	EC umho	P	Cl	NH <sub>4</sub>	NO <sub>3</sub>
							-----ppm-----			
81	5	81	STE	2400000	7.5	675	17.9	110	28.20	0.05
81	6	16	STE	93000	8.1	583	9.95	81.9	31.10	0.16
81	9	13	STE	240000	8.1	564	7.04	97.5	26.10	0.16
81	10	11	STE	11000000	8.2	640	14.1	102	34.10	0.06
81	12	13	STE	640000	7.7	647	15.6	105	29.60	0.31
82	1	10	STE	93000	8.4	738	14.6	110	35.30	0.27
77	2	28	Ground	3	7.9	210	0.00	13.5	0.00	0.00
77	3	13	Ground	3	7.9	182	0.00	11.6	0.00	0.00
77	3	27	Ground	3	7.8	199	0.00	10.3	0.00	0.00
77	4	17	Ground	3	7.9	203	0.00	10.4	0.00	0.00
77	5	1	Ground	3	8.0	234	0.00	10.3	0.00	0.00
77	11	13	Ground	4	8.4	219	0.00	12.3	0.00	0.00
78	2	5	Ground	0	8.1	171	0.00	11.1	0.00	0.01
78	4	2	Ground	0	8.1	156	0.00	10.1	0.00	0.00
78	4	30	Ground	0	8.2	152	0.00	8.0	0.00	0.00
78	5	21	Ground	0	7.5	178	0.00	7.7	0.00	0.00
77	2	28	House	3	8.4	410	0.11	66.6	0.01	0.00
77	3	13	House	3	8.0	418	0.09	68.4	0.10	0.00
77	3	27	House	3	.	431	0.02	67.1	0.00	0.00
77	4	17	House	3	8.0	419	0.11	68.8	0.00	0.00
77	5	1	House	3	7.8	412	0.12	68.4	0.00	0.00
77	6	5	House	0	7.9	380	0.15	69.7	0.40	0.00
77	9	11	House	.	7.5	387	0.09	22.7	0.20	0.00
77	10	16	House	0	7.2	388	0.00	65.7	0.20	0.00
78	1	8	House	0	8.0	392	0.10	65.3	0.30	0.07
78	3	12	House	0	7.3	155	0.00	40.6	0.00	0.02
78	8	21	House	0	7.9	424	0.77	78.3	0.20	0.00



Appendix Table 2. Nitrite, Na, Ca, and Mg concentrations in household waters, septic tank effluent and native ground waters.

Yr	Mo	Day	Source	NO <sub>2</sub>	Na	Ca	Mg	SAR
				----- ppm -----				
81	5	81	STE	0.007	190	4.00	0.80	32.1
81	6	16	STE	0.014	205	9.40	1.54	11.5
81	9	13	STE	0.053	280	6.60	1.00	18.9
81	10	11	STE	0.040	385	5.70	1.19	43.7
81	12	13	STE	0.009	375	6.80	1.19	37.2
82	1	10	STE	0.200	460	6.20	1.41	46.9
77	2	28	Ground	0.000	9.50	71.00	4.00	0.3
77	3	13	Ground	0.000	15.00	57.80	2.50	0.5
77	3	27	Ground	0.002	11.80	5.50	2.40	1.1
77	4	17	Ground	0.001	12.50	66.00	3.30	0.4
77	5	1	Ground	0.000	11.70	82.10	4.30	0.3
77	11	13	Ground	0.000	16.00	41.50	2.10	0.7
78	2	5	Ground	0.000	12.70	55.00	2.35	0.5
78	4	2	Ground	0.000	10.70	52.50	1.70	0.4
78	4	30	Ground	0.000	9.80	48.80	2.35	0.4
78	5	21	Ground	0.002	9.70	64.90	2.48	0.3
77	2	28	House	0.001	179	20.00	3.00	9.8
77	3	13	House	0.000	240	0.60	0.00	85.2
77	3	27	House	0.004	72.00	2.70	1.80	8.3
77	4	17	House	0.001	208	1.40	0.10	45.7
77	5	1	House	0.124	222	0.90	0.04	62.1
77	6	5	House	0.000	200	1.20	0.10	47.0
77	9	11	House	0.000	196	0.60	0.04	66.0
77	10	16	House	0.007	178	0.09	0.07	48.5
78	1	8	House	0.000	160	3.60	0.53	20.0
78	3	12	House	0.011	.	.	.	.
78	8	21	House	0.000	57.00	11.80	1.20	4.2

Appendix Table 3. Fecal coliform densities, EC and P and Cl concentrations in groundwaters averaged for both rows with distance and depth for Systems 2, 3 and 4.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC
				m		ppm		umhos	MPN/100ml
80	6	1	2	0.75	1.5	0.00	18.0	64.4	0.0
80	12	14	2	0.75	1.5	0.00	123.5	266.0	4.5
81	1	11	2	0.75	1.5	0.00	92.4	293.5	75.0
81	1	25	2	0.75	1.5	0.00	113.4	335.0	4.5
81	2	8	2	0.75	1.5	0.01	68.6	210.5	37.5
81	2	22	2	0.75	1.5	0.01	28.8	136.0	7.5
81	3	15	2	0.75	1.5	0.02	25.3	115.2	115.0
81	4	12	2	0.75	1.5	0.02	24.2	113.8	217.0
81	4	26	2	0.75	1.5	.	24.3	126.5	48.5
81	5	17	2	0.75	1.5	0.08	43.2	235.5	0.0
81	5	31	2	0.75	1.5	.	59.9	337.3	1050.0
80	6	1	2	0.75	2.4	0.00	7.7	51.4	0.0
80	9	14	2	0.75	2.4	0.00	7.1	44.8	0.0
80	10	19	2	0.75	2.4	0.00	7.5	51.0	0.0
80	12	14	2	0.75	2.4	0.00	17.5	61.1	0.0
81	1	11	2	0.75	2.4	0.00	17.0	67.0	0.0
81	1	25	2	0.75	2.4	0.00	24.3	77.1	120.0
81	2	8	2	0.75	2.4	0.00	25.7	76.9	0.0
81	2	22	2	0.75	2.4	0.00	20.1	75.1	2.0
81	3	15	2	0.75	2.4	0.00	16.0	66.8	0.0
81	4	12	2	0.75	2.4	0.02	15.8	70.0	21.5
81	4	26	2	0.75	2.4	.	12.6	79.1	4.5
81	5	17	2	0.75	2.4	0.02	15.2	59.3	0.0
81	5	31	2	0.75	2.4	.	12.7	60.8	0.0
81	7	12	2	0.75	2.4	0.29	22.2	104.8	142.0
80	6	1	2	2.25	1.5	0.00	7.6	46.4	0.0
80	9	14	2	2.25	1.5	0.00	9.3	52.9	0.0
80	10	19	2	2.25	1.5	0.00	9.6	55.1	0.0
80	12	14	2	2.25	1.5	0.00	48.6	100.4	0.0
81	1	11	2	2.25	1.5	0.00	98.5	267.0	0.0
81	1	25	2	2.25	1.5	0.00	93.8	277.0	0.0
81	2	8	2	2.25	1.5	0.00	93.0	265.0	0.0
81	2	22	2	2.25	1.5	0.00	58.9	211.5	2.0
81	3	15	2	2.25	1.5	0.00	22.3	116.6	2.0
81	4	12	2	2.25	1.5	0.00	46.6	217.6	115.0
81	4	26	2	2.25	1.5	.	29.2	165.4	11.5
81	5	17	2	2.25	1.5	0.00	16.3	111.2	0.0
81	5	31	2	2.25	1.5	.	30.0	166.5	0.0

Appendix Table 3 continued.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC
				--- m ---		--- ppm ---		umhos	MPN/100mI
80	6	1	2	2.25	2.4	0.00	7.6	48.6	0.0
80	9	14	2	2.25	2.4	0.00	7.7	46.6	0.0
80	10	19	2	2.25	2.4	0.00	7.4	49.5	0.0
80	12	14	2	2.25	2.4	0.00	9.5	50.5	0.0
81	1	11	2	2.25	2.4	0.00	14.9	62.0	0.0
81	1	25	2	2.25	2.4	0.00	17.0	66.0	0.0
81	2	8	2	2.25	2.4	0.00	22.7	70.0	0.0
81	2	22	2	2.25	2.4	0.00	22.2	78.8	0.0
81	3	15	2	2.25	2.4	0.00	22.6	78.0	0.0
81	4	12	2	2.25	2.4	0.00	18.8	72.8	0.0
81	4	26	2	2.25	2.4	.	15.9	69.2	2.0
81	5	17	2	2.25	2.4	0.00	18.1	60.3	0.0
81	5	31	2	2.25	2.4	.	14.5	64.8	0.0
81	7	12	2	2.25	2.4	0.00	11.6	59.1	21.5
80	6	1	3	0.75	1.5	0.00	10.7	56.2	0.0
80	12	14	3	0.75	1.5	0.00	81.9	167.5	2.0
81	1	11	3	0.75	1.5	0.00	84.6	200.2	46.5
81	1	25	3	0.75	1.5	0.00	89.7	43.1	46.5
81	2	8	3	0.75	1.5	0.00	87.6	204.3	2.0
81	2	22	3	0.75	1.5	0.00	43.4	140.9	4.5
81	3	15	3	0.75	1.5	0.00	61.0	190.2	0.0
81	4	12	3	0.75	1.5	0.00	64.8	295.4	2300.0
81	4	26	3	0.75	1.5	.	64.3	310.8	46.5
81	5	17	3	0.75	1.5	0.00	56.6	287.1	2.0
81	5	31	3	0.75	1.5	.	62.0	268.1	75.0
80	6	1	3	0.75	2.4	0.00	7.0	55.2	0.0
80	9	14	3	0.75	2.4	0.00	8.4	51.7	0.0
80	10	19	3	0.75	2.4	0.00	7.4	55.0	0.0
80	12	14	3	0.75	2.4	0.00	19.0	70.2	0.0
81	1	11	3	0.75	2.4	0.01	18.8	76.1	0.0
81	1	25	3	0.75	2.4	0.00	18.4	80.0	0.0
81	2	8	3	0.75	2.4	0.00	14.1	67.0	0.0
81	2	22	3	0.75	2.4	0.00	9.5	61.6	0.0
81	3	15	3	0.75	2.4	0.00	9.1	53.0	0.0
81	4	12	3	0.75	2.4	0.00	8.9	55.8	0.0
81	4	26	3	0.75	2.4	.	8.9	55.9	1.5
81	5	17	3	0.75	2.4	0.00	10.0	51.8	0.0
81	5	31	3	0.75	2.4	.	8.5	54.6	0.0
81	7	12	3	0.75	2.4	0.00	9.7	52.5	0.0
80	6	1	3	2.25	1.5	0.00	7.4	45.7	0.0
80	12	14	3	2.25	1.5	0.00	8.7	42.0	0.0

Appendix Table 3 continued.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC	
				---	m	---	ppm	---	umhos	MPN/100ml
81	1	11	3	2.25	1.5	0.00	40.6	104.8	0.0	
81	1	25	3	2.25	1.5	0.00	49.3	128.9	0.0	
81	2	8	3	2.25	1.5	0.00	64.0	158.3	0.0	
81	2	22	3	2.25	1.5	0.00	67.6	171.2	0.0	
81	3	15	3	2.25	1.5	0.00	17.8	66.0	0.0	
81	4	12	3	2.25	1.5	0.12	58.4	142.3	0.0	
81	4	26	3	2.25	1.5	.	59.2	159.7	0.0	
81	5	17	3	2.25	1.5	0.00	50.8	137.0	0.0	
81	5	31	3	2.25	1.5	.	55.7	140.7	0.0	
80	6	1	3	2.25	2.4	0.00	8.2	50.0	0.0	
80	9	14	3	2.25	2.4	0.01	8.7	48.3	0.0	
80	10	19	3	2.25	2.4	0.00	7.0	54.1	1.5	
80	12	14	3	2.25	2.4	0.00	9.2	54.2	0.0	
81	1	11	3	2.25	2.4	0.00	8.3	56.0	0.0	
81	1	25	3	2.25	2.4	0.01	9.2	55.8	0.0	
81	2	8	3	2.25	2.4	0.00	9.0	57.3	0.0	
81	2	22	3	2.25	2.4	0.00	10.3	56.0	0.0	
81	3	15	3	2.25	2.4	0.00	9.4	54.5	0.0	
81	4	12	3	2.25	2.4	0.00	8.8	55.0	0.0	
81	4	26	3	2.25	2.4	.	8.7	55.0	0.0	
81	5	17	3	2.25	2.4	0.00	10.1	51.3	0.0	
81	5	31	3	2.25	2.4	.	8.2	56.3	0.0	
81	7	12	3	2.25	2.4	0.00	8.6	48.4	0.0	
80	6	1	3	5.25	1.5	0.02	8.3	54.3	0.0	
80	9	14	3	5.25	1.5	0.00	12.5	56.6	0.0	
80	10	19	3	5.25	1.5	0.00	12.2	61.8	0.0	
80	12	14	3	5.25	1.5	0.00	16.1	52.1	0.0	
81	1	11	3	5.25	1.5	0.00	9.8	48.4	0.0	
81	1	25	3	5.25	1.5	0.00	7.7	43.6	0.0	
81	2	8	3	5.25	1.5	0.01	6.5	39.1	0.0	
81	2	22	3	5.25	1.5	0.00	13.4	51.9	0.0	
81	3	15	3	5.25	1.5	0.00	26.4	91.6	0.0	
81	4	12	3	5.25	1.5	0.00	7.8	46.7	0.0	
81	4	26	3	5.25	1.5	.	22.2	64.0	0.0	
81	5	17	3	5.25	1.5	0.00	16.5	59.8	0.0	
81	5	31	3	5.25	1.5	.	17.3	62.2	0.0	
81	7	12	3	5.25	1.5	0.00	7.4	47.1	0.0	
80	6	1	3	5.25	2.4	0.00	10.3	52.1	0.0	
80	9	14	3	5.25	2.4	0.00	8.3	48.5	0.0	
80	10	19	3	5.25	2.4	0.00	8.1	53.2	2.0	
80	12	14	3	5.25	2.4	0.00	12.1	52.0	0.0	

Appendix Table 3 continued.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC
				--- m ---		--- ppm ---		umhos	MPN/100ml
81	1	11	3	5.25	2.4	0.00	9.2	55.8	0.0
81	1	25	3	5.25	2.4	0.00	10.6	58.5	0.0
81	2	8	3	5.25	2.4	0.00	10.2	56.9	0.0
81	2	22	3	5.25	2.4	0.00	9.9	57.5	0.0
81	3	15	3	5.25	2.4	0.00	9.9	55.4	0.0
81	4	12	3	5.25	2.4	0.00	10.8	54.9	0.0
81	4	26	3	5.25	2.4	.	9.6	58.8	0.0
81	5	17	3	5.25	2.4	0.00	11.4	57.6	0.0
81	5	31	3	5.25	2.4	.	8.3	58.3	0.0
81	7	12	3	5.25	2.4	0.00	9.6	55.9	4.5
81	1	11	4	0.75	1.5	0.00	106.5	273.5	0.0
81	1	25	4	0.75	1.5	0.00	101.5	289.0	2.0
81	2	8	4	0.75	1.5	0.00	109.8	295.0	0.0
81	2	22	4	0.75	1.5	0.00	109.5	300.0	2.0
81	3	15	4	0.75	1.5	0.00	100.3	291.0	4.5
81	4	12	4	0.75	1.5	0.00	98.1	285.0	2.0
81	4	26	4	0.75	1.5	.	96.0	302.5	0.0
80	6	1	4	0.75	2.4	0.07	12.4	66.9	0.0
80	9	14	4	0.75	2.4	0.00	11.1	53.0	0.0
80	10	19	4	0.75	2.4	0.00	10.3	63.9	0.0
80	12	14	4	0.75	2.4	0.00	15.8	62.1	0.0
81	1	11	4	0.75	2.4	0.00	12.0	64.8	0.0
81	1	25	4	0.75	2.4	0.00	15.3	69.4	0.0
81	2	8	4	0.75	2.4	0.00	14.5	69.2	0.0
81	2	22	4	0.75	2.4	0.00	13.3	67.9	0.0
81	3	15	4	0.75	2.4	0.00	13.4	69.0	0.0
81	4	12	4	0.75	2.4	0.00	13.9	64.9	0.0
81	4	26	4	0.75	2.4	.	12.0	65.2	0.0
81	5	17	4	0.75	2.4	0.00	16.3	58.8	0.0
81	5	31	4	0.75	2.4	.	12.5	66.5	0.0
81	7	12	4	0.75	2.4	0.00	13.1	67.4	0.0
80	6	1	4	2.25	1.5	.	.	133.0	0.0
80	12	14	4	2.25	1.5	0.00	23.0	104.1	0.0
81	1	11	4	2.25	1.5	0.00	21.1	128.0	0.0
81	1	25	4	2.25	1.5	0.00	13.1	83.4	0.0
81	2	8	4	2.25	1.5	0.00	36.0	159.5	0.0
81	2	22	4	2.25	1.5	0.01	73.3	232.5	0.0
81	3	15	4	2.25	1.5	0.00	83.2	266.0	0.0
81	4	12	4	2.25	1.5	0.00	86.3	282.5	0.0
81	4	26	4	2.25	1.5	.	95.2	296.5	0.0
80	6	1	4	2.25	2.4	0.00	13.5	61.4	0.0

Appendix Table 3 continued.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC
				--- m ---		--- ppm ---		umhos	MPN/100ml
80	9	14	4	2.25	2.4	0.00	10.6	54.9	0.0
80	10	19	4	2.25	2.4	0.00	10.4	63.6	0.0
80	12	14	4	2.25	2.4	0.00	15.2	62.5	0.0
81	1	11	4	2.25	2.4	0.00	11.5	62.0	0.0
81	1	25	4	2.25	2.4	0.00	23.9	117.5	0.0
81	2	8	4	2.25	2.4	0.00	11.8	62.1	0.0
81	2	22	4	2.25	2.4	0.00	12.2	60.8	0.0
81	3	15	4	2.25	2.4	0.00	14.3	65.4	0.0
81	4	12	4	2.25	2.4	0.00	13.0	64.3	0.0
81	4	26	4	2.25	2.4	.	11.9	63.7	0.0
81	5	17	4	2.25	2.4	0.00	14.5	59.7	0.0
81	5	31	4	2.25	2.4	.	12.1	64.9	0.0
81	7	12	4	2.25	2.4	0.00	12.7	63.8	0.0
80	5	18	4	5.25	1.5	0.00	19.7	135.0	0.0
80	6	1	4	5.25	1.5	0.00	22.1	137.0	0.0
80	12	14	4	5.25	1.5	0.00	15.3	125.3	0.0
81	1	25	4	5.25	1.5	0.00	19.6	115.5	0.0
81	2	8	4	5.25	1.5	0.00	24.2	134.5	0.0
81	2	22	4	5.25	1.5	0.00	22.6	121.8	0.0
81	3	15	4	5.25	1.5	0.00	27.7	133.4	0.0
81	4	12	4	5.25	1.5	0.00	31.6	150.0	0.0
81	4	26	4	5.25	1.5	.	28.1	144.0	0.0
80	5	18	4	5.25	2.4	0.00	10.9	68.8	0.0
80	6	1	4	5.25	2.4	0.00	13.3	88.7	0.0
80	9	14	4	5.25	2.4	0.00	11.0	65.4	0.0
80	10	19	4	5.25	2.4	0.04	6.6	46.3	115.0
80	12	14	4	5.25	2.4	0.00	17.8	63.5	0.0
81	1	11	4	5.25	2.4	0.01	12.3	67.5	0.0
81	1	25	4	5.25	2.4	0.00	12.5	67.8	0.0
81	2	8	4	5.25	2.4	0.00	11.2	66.3	0.0
81	2	22	4	5.25	2.4	0.00	11.8	61.6	1.5
81	3	15	4	5.25	2.4	0.00	22.0	89.2	0.0
81	4	12	4	5.25	2.4	0.00	22.2	90.7	0.0
81	4	26	4	5.25	2.4	.	20.7	95.1	0.0
81	5	17	4	5.25	2.4	0.00	18.4	74.1	0.0
81	5	31	4	5.25	2.4	.	19.0	75.4	0.0
81	7	12	4	5.25	2.4	0.00	24.1	87.0	11.5
80	5	18	4	20.25	1.5	0.00	15.9	79.0	4.5
80	6	1	4	20.25	1.5	0.00	12.8	75.6	0.0
80	9	14	4	20.25	1.5	0.00	13.8	76.2	0.0
80	10	19	4	20.25	1.5	0.00	17.3	130.0	0.0

Appendix Table 3 continued.

Yr	Mo	Day	Syst	Dist	Depth	P	Cl	EC	FC	
				---	m	---	ppm	---	umhos	MPN/100ml
80	12	14	4	20.25	1.5	0.00	14.1	92.0	0.0	
81	1	11	4	20.25	1.5	0.00	9.7	94.6	0.0	
81	1	25	4	20.25	1.5	0.00	11.4	95.4	0.0	
81	2	8	4	20.25	1.5	0.00	11.1	90.1	0.0	
81	2	22	4	20.25	1.5	0.00	12.7	79.2	0.0	
81	3	15	4	20.25	1.5	0.00	13.6	74.4	0.0	
81	4	12	4	20.25	1.5	0.00	16.7	86.8	0.0	
81	4	26	4	20.25	1.5	.	17.3	94.4	0.0	
81	5	17	4	20.25	1.5	0.00	20.3	92.7	0.0	
81	5	31	4	20.25	1.5	.	20.1	104.6	0.0	
80	5	18	4	20.25	2.4	0.00	13.6	83.0	0.0	
80	6	1	4	20.25	2.4	0.01	14.3	87.2	0.0	
80	9	14	4	20.25	2.4	0.00	18.6	84.0	0.0	
80	10	19	4	20.25	2.4	0.00	20.9	95.3	0.0	
80	12	14	4	20.25	2.4	0.00	35.3	85.8	0.0	
81	1	11	4	20.25	2.4	0.00	22.0	91.5	0.0	
81	1	25	4	20.25	2.4	0.00	23.2	89.6	0.0	
81	2	8	4	20.25	2.4	0.00	18.6	92.3	0.0	
81	2	22	4	20.25	2.4	0.00	18.7	92.5	0.0	
81	3	15	4	20.25	2.4	0.00	19.4	109.9	0.0	
81	4	12	4	20.25	2.4	0.00	18.2	85.8	0.0	
81	4	26	4	20.25	2.4	.	17.9	86.7	0.0	
81	5	17	4	20.25	2.4	0.00	18.5	81.8	0.0	
81	5	31	4	20.25	2.4	.	18.8	84.2	0.0	
81	7	12	4	20.25	2.4	0.00	17.8	79.2	0.0	

Appendix Table 4. Sodium absorption ratios and Na, Ca and Mg concentrations in groundwaters averaged for both rows with distance and depth for Systems 2, 3 and 4.

Yr	Mo	Day	Syst	Dist	Depth	Na	Ca	Mg	SAR
				m		ppm			
80	12	14	2	0.75	1.5	.	1.1	0.7	.
81	1	25	2	0.75	1.5	140.5	0.8	0.4	62.48
81	2	22	2	0.75	1.5	71.0	0.6	0.3	26.34
81	4	26	2	0.75	1.5	78.0	0.8	0.5	35.33
81	5	31	2	0.75	1.5	152.7	0.6	0.4	56.48
80	9	14	2	0.75	2.4	16.5	9.1	1.4	1.33
80	10	19	2	0.75	2.4	11.0	9.1	1.5	0.89
80	12	14	2	0.75	2.4	.	10.2	1.7	.
81	1	25	2	0.75	2.4	8.0	10.3	1.7	0.59
81	2	22	2	0.75	2.4	22.0	11.2	1.9	1.59
81	4	26	2	0.75	2.4	33.0	9.2	1.8	2.67
81	5	31	2	0.75	2.4	42.0	8.1	1.3	3.62
81	7	12	2	0.75	2.4	45.5	8.8	1.2	3.87
80	9	14	2	2.25	1.5	16.5	2.7	1.4	2.01
80	10	19	2	2.25	1.5	13.5	2.1	1.3	1.79
80	12	14	2	2.25	1.5	.	1.8	1.2	.
81	1	25	2	2.25	1.5	127.0	0.5	0.3	67.78
81	2	22	2	2.25	1.5	116.5	0.5	0.3	52.33
81	4	26	2	2.25	1.5	92.7	0.8	0.5	59.67
81	5	31	2	2.25	1.5	94.5	0.8	0.5	32.25
80	9	14	2	2.25	2.4	14.5	9.9	1.6	1.13
80	10	19	2	2.25	2.4	11.5	8.4	1.4	0.96
80	12	14	2	2.25	2.4	.	9.6	1.6	.
81	1	25	2	2.25	2.4	8.5	9.8	1.9	0.64
81	2	22	2	2.25	2.4	27.5	11.3	2.1	1.95
81	4	26	2	2.25	2.4	25.5	10.9	1.9	1.86
81	5	31	2	2.25	2.4	33.5	9.2	1.6	2.69
81	7	12	2	2.25	2.4	27.5	8.3	1.4	2.34
80	12	14	3	0.75	1.5	.	1.5	0.6	.
81	1	25	3	0.75	1.5	89.5	0.9	0.6	57.99
81	2	22	3	0.75	1.5	77.5	1.1	0.6	46.88
81	4	26	3	0.75	1.5	177.3	1.2	0.6	53.53
81	5	31	3	0.75	1.5	166.5	1.2	0.6	49.44
80	9	14	3	0.75	2.4	8.0	8.7	2.0	0.63
80	10	19	3	0.75	2.4	11.5	8.1	1.9	0.94
80	12	14	3	0.75	2.4	.	10.2	2.3	.
81	1	25	3	0.75	2.4	17.0	8.8	2.1	1.29
81	2	22	3	0.75	2.4	19.5	8.0	1.8	1.61
81	4	26	3	0.75	2.4	23.5	9.9	2.1	1.76



Appendix Table 4 continued.

Yr	Mo	Day	Syst	Dist	Depth	Na	Ca	Mg	SAR
				---	m	-----	ppm	-----	
81	4	26	3	2.25	1.5	74.0	1.2	0.8	33.37
81	5	31	3	0.75	2.4	31.0	8.8	1.8	2.48
81	7	12	3	0.75	2.4	26.5	8.0	1.8	2.19
80	12	14	3	2.25	1.5	.	2.4	1.2	.
81	1	25	3	2.25	1.5	44.0	1.7	1.2	6.29
81	2	22	3	2.25	1.5	72.0	1.9	1.1	12.31
81	5	31	3	2.25	1.5	69.0	1.0	0.5	21.02
80	9	14	3	2.25	2.4	14.0	10.1	1.7	1.09
80	10	19	3	2.25	2.4	12.6	9.2	1.6	1.00
80	12	14	3	2.25	2.4	.	10.5	1.9	.
81	1	25	3	2.25	2.4	7.5	9.2	1.7	0.60
81	2	22	3	2.25	2.4	14.5	9.6	1.8	1.12
81	4	26	3	2.25	2.4	24.0	10.1	1.8	1.81
81	5	31	3	2.25	2.4	30.0	9.7	1.7	2.34
81	7	12	3	2.25	2.4	26.5	10.0	1.8	2.02
80	9	14	3	5.25	1.5	11.0	5.7	2.5	0.96
80	10	19	3	5.25	1.5	14.0	4.8	2.4	1.30
80	12	14	3	5.25	1.5	.	5.3	2.5	.
81	1	25	3	5.25	1.5	7.5	3.4	1.9	0.81
81	2	22	3	5.25	1.5	20.5	3.4	1.8	2.22
81	4	26	3	5.25	1.5	32.5	3.6	1.9	3.55
81	5	31	3	5.25	1.5	40.0	3.1	1.4	4.92
81	7	12	3	5.25	1.5	27.5	2.5	1.5	3.55
80	9	14	3	5.25	2.4	18.8	11.3	1.8	1.37
80	10	19	3	5.25	2.4	19.3	10.1	1.7	1.46
80	12	14	3	5.25	2.4	.	11.6	1.8	.
81	1	25	3	5.25	2.4	8.5	10.5	1.7	0.63
81	2	22	3	5.25	2.4	17.5	10.8	1.8	1.30
81	4	26	3	5.25	2.4	25.5	12.8	2.2	1.73
81	5	31	3	5.25	2.4	32.5	12.6	1.9	2.25
81	7	12	3	5.25	2.4	27.5	11.0	1.8	2.01
80	12	14	4	0.75	1.5	.	6.2	2.5	.
81	1	25	4	0.75	1.5	98.0	5.5	2.5	11.05
81	2	22	4	0.75	1.5	119.0	4.2	1.8	13.54
81	4	26	4	0.75	1.5	131.5	2.9	1.2	19.09
80	9	14	4	0.75	2.4	22.1	8.4	2.2	1.83
80	10	19	4	0.75	2.4	19.8	8.1	2.3	1.61
80	12	14	4	0.75	2.4	.	9.5	2.7	.
81	1	25	4	0.75	2.4	10.0	8.5	2.5	0.77
81	2	22	4	0.75	2.4	15.0	6.3	2.4	1.34
81	4	26	4	0.75	2.4	24.5	9.3	2.5	1.83
81	5	31	4	0.75	2.4	33.0	8.6	2.3	2.57
81	4	26	3	2.25	1.5	74.0	1.2	0.8	33.37
81	7	12	4	0.75	2.4	28.0	8.5	2.3	2.18

Appendix Table 4 continued.

Yr	Mo	Day	Syst	Dist	Depth	Na	Ca	Mg	SAR
				--- m ---		----- ppm -----			
80	12	14	4	2.25	1.5	.	6.1	2.9	.
81	1	25	4	2.25	1.5	12.5	6.9	2.4	1.04
81	2	22	4	2.25	1.5	66.5	7.2	3.5	5.08
81	4	26	4	2.25	1.5	111.0	5.6	2.5	9.96
80	9	14	4	2.25	2.4	13.0	9.7	2.3	0.98
80	10	19	4	2.25	2.4	14.5	8.2	2.2	1.16
80	12	14	4	2.25	2.4	.	9.3	2.4	.
81	1	25	4	2.25	2.4	28.5	6.3	2.3	2.58
81	2	22	4	2.25	2.4	17.5	8.2	2.2	1.39
81	4	26	4	2.25	2.4	23.5	9.3	2.4	1.77
81	5	31	4	2.25	2.4	34.0	8.9	2.2	2.63
81	7	12	4	2.25	2.4	27.0	9.0	2.3	2.07
80	5	18	4	5.25	1.5	28.0	19.8	4.7	1.42
80	12	14	4	5.25	1.5	.	4.6	2.3	.
81	1	25	4	5.25	1.5	22.0	6.6	3.6	1.75
81	2	22	4	5.25	1.5	33.5	6.9	3.8	2.71
81	4	26	4	5.25	1.5	39.0	9.4	5.3	2.52
80	5	18	4	5.25	2.4	17.0	8.8	2.8	1.27
80	9	14	4	5.25	2.4	13.5	9.0	2.6	1.02
80	10	19	4	5.25	2.4	12.0	5.1	1.8	1.17
80	12	14	4	5.25	2.4	.	8.0	2.7	.
81	1	25	4	5.25	2.4	9.5	6.1	2.2	0.83
81	2	22	4	5.25	2.4	18.0	5.7	2.0	1.63
81	4	26	4	5.25	2.4	31.0	8.0	2.9	2.40
81	5	31	4	5.25	2.4	38.0	6.8	2.1	3.24
81	7	12	4	5.25	2.4	29.5	8.3	2.8	2.26
80	5	18	4	20.25	1.5	11.6	13.0	2.1	0.79
80	9	14	4	20.25	1.5	14.0	14.4	1.4	0.94
80	10	19	4	20.25	1.5	23.0	25.6	3.8	1.12
80	12	14	4	20.25	1.5	.	18.0	2.3	.
81	1	25	4	20.25	1.5	9.0	18.2	2.1	0.53
81	2	22	4	20.25	1.5	18.0	12.0	1.6	1.29
81	4	26	4	20.25	1.5	27.0	10.3	1.8	2.04
81	5	31	4	20.25	1.5	39.0	11.9	2.0	2.75
80	5	18	4	20.25	2.4	19.9	12.4	3.5	1.30
80	9	14	4	20.25	2.4	17.5	12.3	3.1	1.15
80	10	19	4	20.25	2.4	20.0	10.6	3.3	1.37
80	12	14	4	20.25	2.4	.	7.6	3.2	.
81	1	25	4	20.25	2.4	13.0	9.0	3.1	0.95
81	2	22	4	20.25	2.4	17.0	9.3	3.1	1.23
81	4	26	4	20.25	2.4	24.5	9.6	2.9	1.78
81	5	31	4	20.25	2.4	35.0	9.7	2.8	2.55
81	7	12	4	20.25	2.4	28.0	10.1	2.7	2.02

Appendix Table 5. Nitrate-N and NH<sub>4</sub>-N concentrations and NO<sub>3</sub>-N/Cl and NH<sub>4</sub>-N/Cl ratios in groundwaters averaged for both rows with distance and depth for Systems 2, 3 and 4.

Yr	Mo	Day	Syst	Dist	Depth	NO <sub>3</sub>	NO <sub>3</sub> /Cl	NH <sub>4</sub>	NH <sub>4</sub> /Cl
				---	m	---			
						ppm			ppm
80	6	1	2	0.75	1.5	0.16	0.011	0.09	0.006
80	12	14	2	0.75	1.5	0.55	0.008	0.19	0.001
81	1	11	2	0.75	1.5	4.37	0.054	0.61	0.005
81	1	25	2	0.75	1.5	2.77	0.044	1.10	0.007
81	2	8	2	0.75	1.5	2.63	0.050	1.10	0.012
81	2	22	2	0.75	1.5	1.94	0.067	0.72	0.023
81	3	15	2	0.75	1.5	1.22	0.077	0.27	0.007
81	4	12	2	0.75	1.5	0.19	0.017	0.74	0.031
81	4	26	2	0.75	1.5	0.34	0.030	0.80	0.023
81	5	17	2	0.75	1.5	0.24	0.016	1.42	0.023
81	5	31	2	0.75	1.5	0.11	0.004	5.06	0.049
80	6	1	2	0.75	2.4	0.03	0.004	0.15	0.020
80	9	14	2	0.75	2.4	0.00	0.000	0.17	0.024
80	10	19	2	0.75	2.4	0.03	0.004	0.09	0.012
80	12	14	2	0.75	2.4	0.00	0.000	0.07	0.004
81	1	11	2	0.75	2.4	0.00	0.000	0.09	0.005
81	1	25	2	0.75	2.4	0.08	0.004	0.10	0.004
81	2	8	2	0.75	2.4	0.06	0.002	0.10	0.004
81	2	22	2	0.75	2.4	0.07	0.003	0.07	0.003
81	3	15	2	0.75	2.4	0.10	0.006	0.04	0.003
81	4	12	2	0.75	2.4	0.05	0.003	0.43	0.026
81	4	26	2	0.75	2.4	0.10	0.008	2.55	0.190
81	5	17	2	0.75	2.4	0.10	0.007	0.19	0.013
81	5	31	2	0.75	2.4	0.07	0.005	0.11	0.009
81	7	12	2	0.75	2.4	0.03	0.002	0.66	0.022
80	6	1	2	2.25	1.5	0.24	0.030	0.07	0.010
80	9	14	2	2.25	1.5	0.06	0.009	0.10	0.010
80	10	19	2	2.25	1.5	0.06	0.008	0.05	0.005
80	12	14	2	2.25	1.5	1.10	0.023	0.05	0.001
81	1	11	2	2.25	1.5	1.81	0.022	0.06	0.001
81	1	25	2	2.25	1.5	1.59	0.020	0.04	0.000
81	2	8	2	2.25	1.5	1.90	0.026	0.05	0.000
81	2	22	2	2.25	1.5	1.43	0.035	0.22	0.003
81	3	15	2	2.25	1.5	2.40	0.105	0.02	0.001
81	4	12	2	2.25	1.5	0.65	0.079	0.08	0.003
81	4	26	2	2.25	1.5	0.79	0.084	0.21	0.005
81	5	17	2	2.25	1.5	0.45	0.042	0.24	0.011
81	5	31	2	2.25	1.5	0.47	0.067	0.68	0.015

Appendix Table 5 continued.

Yr	Mo	Day	Syst	Dist	Depth	NO3	NO3/Cl	NH4	NH4/Cl
					--- m ---	ppm			
							ppm		
81	2	22	2	2.25	2.4	0.09	0.003	0.07	0.003
80	6	1	2	2.25	2.4	0.08	0.011	0.14	0.019
80	9	14	2	2.25	2.4	0.01	0.002	0.16	0.022
80	10	19	2	2.25	2.4	0.01	0.001	0.10	0.014
80	12	14	2	2.25	2.4	0.03	0.003	0.07	0.007
81	1	11	2	2.25	2.4	0.00	0.000	0.09	0.007
81	1	25	2	2.25	2.4	0.00	0.000	0.07	0.004
81	2	8	2	2.25	2.4	0.14	0.006	0.07	0.004
81	3	15	2	2.25	2.4	0.16	0.007	0.05	0.002
81	4	12	2	2.25	2.4	0.03	0.001	0.09	0.005
81	4	26	2	2.25	2.4	0.10	0.006	0.08	0.005
81	5	17	2	2.25	2.4	0.12	0.006	0.07	0.004
81	5	31	2	2.25	2.4	0.07	0.005	0.03	0.002
81	7	12	2	2.25	2.4	0.03	0.003	0.06	0.005
80	6	1	3	0.75	1.5	0.51	0.079	0.04	0.005
80	12	14	3	0.75	1.5	2.64	0.088	0.04	0.004
81	1	11	3	0.75	1.5	2.71	0.219	0.04	0.004
81	1	25	3	0.75	1.5	3.08	0.170	0.06	0.002
81	2	8	3	0.75	1.5	2.50	0.153	0.08	0.002
81	2	22	3	0.75	1.5	1.66	0.147	0.23	0.005
81	3	15	3	0.75	1.5	1.51	0.143	0.75	0.008
81	4	12	3	0.75	1.5	1.26	0.208	2.01	0.018
81	4	26	3	0.75	1.5	1.12	0.183	2.30	0.020
81	5	17	3	0.75	1.5	0.95	0.107	2.45	0.026
81	5	31	3	0.75	1.5	0.70	0.096	2.01	0.018
80	6	1	3	0.75	2.4	0.10	0.015	0.11	0.015
80	9	14	3	0.75	2.4	0.03	0.004	0.11	0.013
80	10	19	3	0.75	2.4	0.01	0.001	0.08	0.011
80	12	14	3	0.75	2.4	0.18	0.012	0.14	0.010
81	1	11	3	0.75	2.4	0.13	0.016	0.08	0.005
81	1	25	3	0.75	2.4	0.10	0.010	0.07	0.004
81	2	8	3	2.25	1.5	2.15	0.080	0.03	0.002
81	2	22	3	2.25	1.5	3.66	0.094	0.04	0.002
81	3	15	3	2.25	1.5	2.14	0.128	0.02	0.002
81	4	12	3	2.25	1.5	0.72	0.075	0.03	0.001
81	4	26	3	2.25	1.5	0.63	0.072	0.03	0.002
81	5	17	3	2.25	1.5	0.27	0.021	0.03	0.001
81	5	31	3	2.25	1.5	0.17	0.007	0.02	0.001
80	6	1	3	2.25	2.4	0.20	0.025	0.10	0.012
80	9	14	3	2.25	2.4	0.03	0.004	0.06	0.007
80	10	19	3	2.25	2.4	0.02	0.003	0.08	0.012
80	12	14	3	2.25	2.4	0.00	0.000	0.08	0.008

Appendix Table 5 continued.

Yr	Mo	Day	Syst	Dist	Depth	NO3	NO3/Cl	NH4	NH4/Cl
					--- m ---	ppm		ppm	
81	2	22	2	2.25	2.4	0.09	0.003	0.07	0.003
81	1	11	3	2.25	2.4	0.00	0.000	0.08	0.009
81	1	25	3	2.25	2.4	0.05	0.005	0.04	0.005
81	2	8	3	2.25	2.4	0.03	0.003	0.08	0.009
81	2	22	3	2.25	2.4	0.05	0.004	0.05	0.005
81	3	15	3	2.25	2.4	0.02	0.002	0.01	0.001
81	4	12	3	2.25	2.4	0.09	0.011	0.08	0.009
81	4	26	3	2.25	2.4	0.08	0.009	0.07	0.008
81	5	17	3	2.25	2.4	0.12	0.012	0.09	0.009
81	5	31	3	2.25	2.4	0.09	0.010	0.03	0.004
81	7	12	3	2.25	2.4	0.03	0.003	0.12	0.014
80	6	1	3	5.25	1.5	0.08	0.010	0.05	0.006
80	9	14	3	5.25	1.5	0.01	0.001	0.09	0.007
80	10	19	3	5.25	1.5	0.01	0.001	0.06	0.005
80	12	14	3	5.25	1.5	0.07	0.004	0.03	0.002
81	1	11	3	5.25	1.5	0.14	0.014	0.05	0.005
81	1	25	3	5.25	1.5	0.06	0.007	0.04	0.005
81	2	8	3	5.25	1.5	0.05	0.007	0.04	0.006
81	2	22	3	5.25	1.5	0.37	0.023	0.04	0.004
81	3	15	3	5.25	1.5	1.25	0.039	0.01	0.000
81	4	12	3	5.25	1.5	0.61	0.067	0.04	0.006
81	4	26	3	5.25	1.5	0.57	0.021	0.04	0.002
81	5	17	3	5.25	1.5	0.50	0.025	0.04	0.003
81	5	31	3	5.25	1.5	0.55	0.026	0.02	0.002
81	7	12	3	5.25	1.5	0.17	0.020	0.05	0.007
80	6	1	3	5.25	2.4	0.07	0.007	0.16	0.016
80	9	14	3	5.25	2.4	0.01	0.001	0.04	0.005
80	10	19	3	5.25	2.4	0.04	0.005	0.08	0.009
80	12	14	3	5.25	2.4	0.04	0.003	0.09	0.007
81	1	11	3	5.25	2.4	0.03	0.004	0.09	0.009
81	1	25	3	5.25	2.4	0.00	0.000	0.09	0.008
81	2	8	3	5.25	2.4	0.07	0.007	0.09	0.009
81	2	22	3	5.25	2.4	0.03	0.003	0.09	0.010
81	3	15	3	5.25	2.4	0.07	0.006	0.02	0.002
81	4	12	3	5.25	2.4	0.05	0.004	0.10	0.010
81	4	26	3	5.25	2.4	0.13	0.013	0.10	0.011
81	5	17	3	5.25	2.4	0.15	0.014	0.08	0.007
81	5	31	3	5.25	2.4	0.07	0.008	0.06	0.008
81	7	12	3	5.25	2.4	0.04	0.004	0.21	0.022
80	12	14	4	0.75	1.5	0.44	0.005	0.03	0.000
81	1	11	4	0.75	1.5	0.59	0.005	0.05	0.000
81	1	25	4	0.75	1.5	0.56	0.005	0.03	0.000

Appendix Table 5 continued.

Yr	Mo	Day	Syst	Dist	Depth	NO3	NO3/Cl	NH4	NH4/Cl
					--- m ---	ppm			
							ppm		
81	2	22	2	2.25	2.4	0.09	0.003	0.07	0.003
81	2	8	4	0.75	1.5	1.82	0.014	0.03	0.000
81	2	22	4	0.75	1.5	3.73	0.027	0.03	0.000
81	3	15	4	0.75	1.5	4.07	0.032	0.04	0.000
81	4	12	4	0.75	1.5	2.93	0.025	0.02	0.000
81	4	26	4	0.75	1.5	3.61	0.032	0.07	0.000
80	6	1	4	0.75	2.4	0.08	0.006	0.06	0.005
80	9	14	4	0.75	2.4	0.02	0.002	0.07	0.007
80	10	19	4	0.75	2.4	0.01	0.001	0.07	0.007
80	12	14	4	0.75	2.4	0.16	0.010	0.05	0.003
81	1	11	4	0.75	2.4	0.09	0.007	0.07	0.006
81	1	25	4	0.75	2.4	0.18	0.011	0.07	0.004
81	2	8	4	0.75	2.4	0.05	0.003	0.05	0.004
81	2	22	4	0.75	2.4	0.11	0.008	0.04	0.003
81	3	15	4	0.75	2.4	0.14	0.010	0.03	0.002
81	4	12	4	0.75	2.4	0.02	0.001	0.07	0.005
81	4	26	4	0.75	2.4	0.09	0.007	0.05	0.004
81	5	17	4	0.75	2.4	0.10	0.006	0.05	0.003
81	5	31	4	0.75	2.4	0.07	0.006	0.04	0.003
81	7	12	4	0.75	2.4	0.02	0.001	0.07	0.006
80	6	1	4	2.25	1.5	0.26	.	0.05	.
80	12	14	4	2.25	1.5	0.24	0.010	0.03	0.001
81	1	11	4	2.25	1.5	0.27	0.013	0.04	0.002
81	1	25	4	2.25	1.5	0.17	0.013	0.03	0.002
81	2	8	4	2.25	1.5	0.33	0.009	0.04	0.001
81	2	22	4	2.25	1.5	0.53	0.007	0.03	0.000
81	3	15	4	2.25	1.5	1.45	0.016	0.02	0.000
81	4	12	4	2.25	1.5	1.85	0.017	0.03	0.000
81	4	26	4	2.25	1.5	2.66	0.022	0.03	0.000
80	6	1	4	2.25	2.4	0.07	0.005	0.06	0.004
80	9	14	4	2.25	2.4	0.03	0.003	0.06	0.005
80	10	19	4	2.25	2.4	0.01	0.001	0.07	0.007
80	12	14	4	2.25	2.4	0.17	0.012	0.09	0.006
81	1	11	4	2.25	2.4	0.07	0.006	0.07	0.006
81	1	25	4	2.25	2.4	0.21	0.009	0.04	0.003
81	2	8	4	2.25	2.4	0.09	0.007	0.05	0.004
81	2	22	4	2.25	2.4	0.03	0.002	0.05	0.004
81	3	15	4	2.25	2.4	0.16	0.011	0.06	0.004
81	4	12	4	2.25	2.4	0.02	0.002	0.06	0.005
81	4	26	4	2.25	2.4	0.07	0.006	0.05	0.004
81	5	17	4	2.25	2.4	0.11	0.007	0.04	0.003
81	5	31	4	2.25	2.4	0.07	0.005	0.04	0.003

Appendix Table 5 continued.

Yr	Mo	Day	Syst	Dist	Depth	NO3	NO3/Cl	NH4	NH4/Cl
				---	m	----	ppm	ppm	
81	2	22	2	2.25	2.4	0.09	0.003	0.07	0.003
81	7	12	4	2.25	2.4	0.03	0.002	0.08	0.006
80	5	18	4	5.25	1.5	0.10	0.004	1.64	0.083
80	6	1	4	5.25	1.5	0.24	0.011	0.21	0.010
80	12	14	4	5.25	1.5	0.26	0.018	0.06	0.004
81	1	25	4	5.25	1.5	0.37	0.019	0.04	0.002
81	2	8	4	5.25	1.5	0.21	0.011	0.03	0.001
81	2	22	4	5.25	1.5	0.52	0.025	0.04	0.002
81	3	15	4	5.25	1.5	0.35	0.013	0.00	0.000
81	4	12	4	5.25	1.5	0.03	0.001	0.03	0.001
81	4	26	4	5.25	1.5	0.10	0.004	0.05	0.002
80	5	18	4	5.25	2.4	0.05	0.004	1.28	0.118
80	6	1	4	5.25	2.4	0.40	0.030	0.15	0.011
80	9	14	4	5.25	2.4	0.00	0.000	0.15	0.013
80	10	19	4	5.25	2.4	0.08	0.033	0.07	0.011
80	12	14	4	5.25	2.4	0.16	0.009	0.08	0.004
81	1	11	4	5.25	2.4	0.14	0.011	0.05	0.004
81	1	25	4	5.25	2.4	0.15	0.012	0.04	0.003
81	2	8	4	5.25	2.4	0.02	0.002	0.03	0.002
81	2	22	4	5.25	2.4	0.23	0.019	0.05	0.004
81	3	15	4	5.25	2.4	0.13	0.007	0.01	0.001
81	4	12	4	5.25	2.4	0.01	0.000	0.05	0.003
81	4	26	4	5.25	2.4	0.08	0.004	0.18	0.012
81	5	17	4	5.25	2.4	0.09	0.005	0.07	0.005
81	5	31	4	5.25	2.4	0.07	0.004	0.06	0.003
81	7	12	4	5.25	2.4	0.03	0.002	0.15	0.010
80	5	18	4	20.25	1.5	0.08	0.007	1.25	0.120
80	6	1	4	20.25	1.5	0.11	0.011	0.06	0.006
80	9	14	4	20.25	1.5	0.13	0.009	0.06	0.004
80	10	19	4	20.25	1.5	0.58	0.034	0.08	0.005
80	12	14	4	20.25	1.5	0.24	0.017	0.02	0.001
81	1	11	4	20.25	1.5	0.21	0.022	0.05	0.005
81	1	25	4	20.25	1.5	0.19	0.017	0.03	0.003
81	2	8	4	20.25	1.5	0.19	0.017	0.04	0.004
81	2	22	4	20.25	1.5	0.58	0.046	0.04	0.003
81	3	15	4	20.25	1.5	0.26	0.019	0.01	0.001
81	4	12	4	20.25	1.5	0.09	0.005	0.02	0.001
81	4	26	4	20.25	1.5	0.14	0.008	0.02	0.001
81	5	17	4	20.25	1.5	0.15	0.007	0.02	0.001
81	5	31	4	20.25	1.5	0.23	0.011	0.02	0.001
80	5	18	4	20.25	2.4	0.05	0.003	1.45	0.107
80	6	1	4	20.25	2.4	0.06	0.004	0.14	0.010

Appendix Table 5 continued.

Yr	Mo	Day	Syst	Dist	Depth	NO3	NO3/Cl	NH4	NH4/Cl
				---	m	----	ppm	ppm	
81	2	22	2	2.25	2.4	0.09	0.003	0.07	0.003
80	9	14	4	20.25	2.4	0.00	0.000	0.15	0.008
80	10	19	4	20.25	2.4	0.01	0.000	0.12	0.006
80	12	14	4	20.25	2.4	0.07	0.002	0.09	0.002
81	1	11	4	20.25	2.4	0.28	0.012	0.09	0.004
81	1	25	4	20.25	2.4	0.12	0.005	0.08	0.003
81	2	8	4	20.25	2.4	0.19	0.010	0.07	0.004
81	2	22	4	20.25	2.4	0.18	0.010	0.09	0.005
81	3	15	4	20.25	2.4	0.18	0.009	0.03	0.002
81	4	12	4	20.25	2.4	0.01	0.001	0.08	0.004
81	4	26	4	20.25	2.4	0.07	0.004	0.08	0.004
81	5	17	4	20.25	2.4	0.09	0.005	0.07	0.003
81	5	31	4	20.25	2.4	0.06	0.003	0.04	0.002
81	7	12	4	20.25	2.4	0.02	0.001	0.10	0.005



## Appendix II - Soil Description

Typifying Pedon: Study site soil, the Tomotley series, a fine-loamy, mixed, thermic Typic Ochraquult.<sup>†</sup> (Colors are for moist soil).

0	3.8 - 0	Dark, partially decomposed and decomposed roots and leaves.
A1	0 - 18	Very dark gray (10 YR 3/1) sandy loam; weak medium granular structure; friable; slightly sticky; slightly plastic; common very fine, and few fine tubular pores; common very fine, fine, medium and coarse roots (some up to 20 cm); abrupt smooth boundary.
Blg	18-38	Light brownish gray (10 YR 6/2) sandy loam, common medium and coarse prominent brownish yellow (10 YR 6/8) mottles; weak medium and coarse subangular blocky structure; friable; slightly sticky; slightly plastic; common very fine and fine vesicular and tubular pores; few very fine and fine and common medium roots; clear smooth boundary.
B2ltg	38-73	Light gray (10 YR 6/1) sandy clay loam, many medium and coarse prominent yellowish brown (10 YR 5/8) mottles; moderate coarse prismatic

<sup>†</sup>Soil classified and described by Mr. D. R. Hatch, Chief Surveyor, Va. Tech Soil Survey, Virginia Beach, VA.

structure; friable; slightly sticky; slightly plastic; many very fine and fine vesicular and tubular and many medium tubular pores; common very fine, few fine, and few medium roots; few discontinuous thin clay films; clear smooth boundary.

- B22tg      73-103      Light gray (10 YR 6/1) sandy clay loam, many coarse prominent yellowish brown (10 YR 5/8) and common medium and coarse prominent strong brown (7.5 YR 5/6) mottles; moderate coarse prismatic structure; friable; slightly sticky; slightly plastic; many very fine, common fine and common medium vesicular and tubular pores; common very fine, few fine and medium roots along ped faces; gradual smooth boundary.
- C1      103-128      Very pale brown (10 YR 7/3) and strong brown (7.5 YR 5/8) sand, common fine and medium prominent yellowish red (5 YR 5/6) mottles; single grain structure; loose; nonsticky; nonplastic; few fine vesicular pores; abrupt smooth boundary.
- C2      128-183      Pale brown (10 YR 6/3) and brownish yellow (10 YR 6/8) with thin horizontal bands of very pale brown (10 YR 7/3) sand, common fine and medium strong brown (7.5 YR 5/8) mottles;

single grain structure; loose; nonsticky;  
nonplastic; abrupt smooth boundary.

C3g 183-208+

Gray (N/5) sand, single grain structure;  
loose; nonsticky; nonplastic.

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