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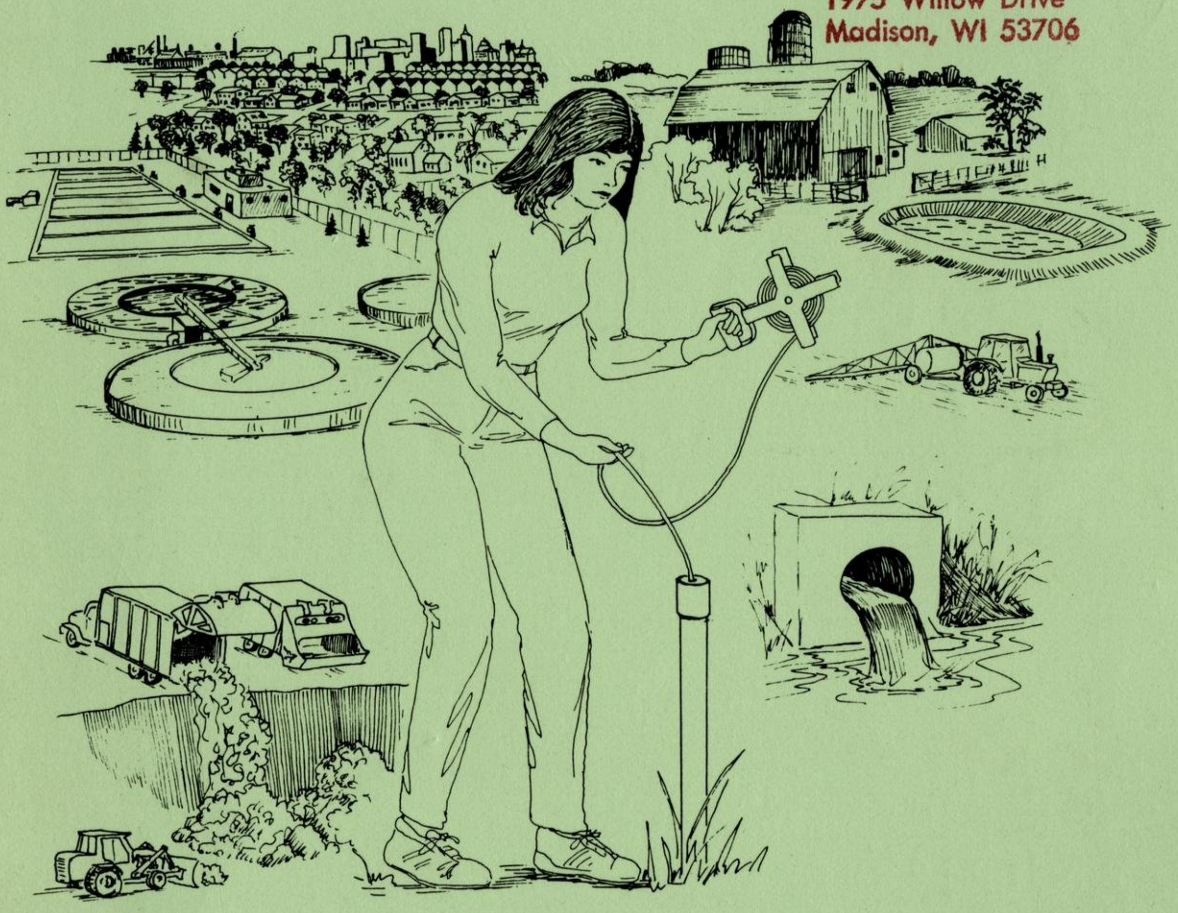

GROUNDWATER
Wisconsin's
buried treasure

Wisconsin Department of Natural Resources

FIELD INVESTIGATION OF GROUNDWATER IMPACTS OF ABSORPTION POND SYSTEMS USED FOR WASTEWATER DISPOSAL

BY

LAURIE PARSONS

An Advanced Independent Study
Submitted in Partial Fulfillment
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SECTION 1 - INTRODUCTION

Land application of wastewater is a common method of treatment and disposal. Absorption ponds, also known a seepage cells, are one type among several land application techniques which have evolved as a cost effective means for land disposal of wastewater. Absorption pond systems are used for treatment and disposal of a variety of waste types including wastewaters generated by the dairy and cheese industry, food processing industry and municipal sewage treatment plants.

The definition of an absorption pond system is broad, ranging from natural depression areas in the landscape to systems that are carefully designed and constructed (WDNR 1984). The types of wastewater discharged to absorption ponds also varies. In Wisconsin the major types include secondary effluent from municipal wastewater treatment plants, dairy and cheese industry wastes, and food processing industry wastes. For smaller industries (rural cheese factories the absorption ponds are often the only means of treatment and disposal. Other larger systems may use biomechanical pretreatment. Due to this diversity associated with absorption ponds, it is difficult to generalize about all systems.

The systems are typically comprised of basins excavated into moderately to highly permeable soils. The basins are designed to distribute wastewater over the pond bottom and to control the rate of wastewater infiltration to the soil.

Design of these systems is dependent on the capability for lateral and vertical flow away from the application site. A cyclic application is the typical mode of operation with a flooding period (loading) followed by drying (resting). For operational flexibility the systems usually consist of two or more basins.

In Wisconsin there are approximately 250 absorption pond systems currently in use. Site selection and design has been based primarily on hydraulic capabilities of the soil and organic loading rates. Design standards to minimize groundwater quality degradation have required site isolation (sufficient distance from water supply wells) and maintenance of an adequate depth to groundwater. In light of the current groundwater standards (Wisconsin Administrative Code NR 140), these system design standards need to be re-evaluated.

This paper presents the results of a field study which was conducted to characterize the soil treatment capabilities and effects on the groundwater at two representative absorption pond sites Included is a review of background literature related to specific goals of the project (Section 2), a description of the monitoring network (Section 3), results and a discussion of the data collected (Sections 4 and 5), and a presentation of conclusions and implications for future designs of these systems (Sections 6 and 7).

This 15 month study was initiated in October 1984 by the Wisconsin Department of Natural Resources (WDNR). The two sites selected for study were the wastewater disposal pond site at Brunkow Cheese Cooperative located 5 miles north of Darlington, Wisconsin, in LaFayette County and the City of Evansville wastewater treatment plant in Rock County, Wisconsin.

A primary goal in this research was to characterize nitrogen removal efficiencies by soil absorption and transformation processes within the unsaturated zone below each system. Absorption ponds have generally proven to be effective in removing wastewater constituents such as BOD_5, COD, phosphorus and coliform bacteria under the current design criteria. In theory these systems can be designed and operated to reduce nitrogen concentrations in the infiltrate to meet the desired groundwater quality standard of 10 mg/l nitrate-nitrogen. In practice, however, as nitrogen removal depends upon temperature, an available carbon source, time and other factors, it appears that significant nitrogen removal does not occur at all sites and at all times of the year.

Other goals of the project which aided in evaluating nitrogen removal were to: 1) characterize the wastewater applied to the two absorption pond systems; 2) characterize the soil pore water and groundwater in the vicinity of each system; and 3) evaluate the overall effectiveness of wastewater treatment (in the pretreatment system and unsaturated zone) including seasonal variations at each site under the operation maintenance procedures used.

SECTION 2 - LITERATURE REVIEW

2.1 Introduction

The land application process utilizing absorption ponds has been established as an economically attractive, low cost alternative which can provide substantial wastewater treatment if properly designed and operated. In an ideal situation, as the unsaturated soil beneath the absorption ponds receives wastewater, treatment occurs through physical, chemical and biological processes prior to reaching the groundwater. The mechanisms involved in these processes are infiltration at the soil surface and flow and pollutant transformation in the unsaturated soil profile. The impacts on groundwater are determined by the quality of the wastewater percolate and local soil and groundwater flow conditions.

Past research has shown the effectiveness of both slow rate and rapid rate land application for treatment. Efforts have focused on laboratory evaluation of wastewater infiltration and removal mechanisms in soil columns (Lance 1972 and 1976, Enfield 1977, Keeney 1979, Gilbert 1979, Reddy 1981, Piwoni 1986) and on small scale studies through the use of test plots (large lysimeters) (Iskandar 1978, Leach 1983, Brown 1984).

While these studies provide valuable information regarding removal rates and interpretation, they do not reflect the effects of large scale heterogeneities in the soil or variable field conditions. A limited number of investigations have been done with full scale operating systems (Bennett 1983, Rice 1984); however, these studies do not evaluate either long term treatment or groundwater impacts at distances downgradient of the site.

Most previous work has focused on determining two design parameters: 1) optimal hydraulic loading rates; and 2) necessary treatment levels (chiefly nitrogen removal). A majority of the investigative work evaluates wastewater loading schedules as a means to satisfy both of these design objectives.

Full scale operations need to be evaluated more completely to quantify those field conditions, such as cold temperatures, reduced infiltration capacity due to long term use, variability of wastewater, hydrogeologic and site characteristics that influence the magnitude of groundwater contamination. A description of these field conditions follows.

2.2 System Design: Hydraulic Capacity and Infiltration

The design of an absorption pond system is dependent on the type of wastewater, required loading rates and site conditions. Rapid infiltration (RI) basin is the term which has been assigned to large systems. Land treatment by rapid infiltration has been defined as the controlled application of wastewater to earthen basins in permeable soils at rates ranging from .3 ft/week to 8 ft/week (Reed 1984).

The initial step in system design is site selection. General procedures for site selection of land treatment systems are well documented (EPA Process Design Manual 1981, Reed 1984, Overcash 1979). Potential sites must be selected based on land area requirements (subject to hydraulic loading rates), soil type, topography, hydrogeologic conditions and proximity to residences and to water supply wells. Table 2.1 summarizes design requirements for both municipal and industrial absorption pond systems in Wisconsin.

Table 2.1 - System Design Requirements in Wisconsin

Design Parameter	Municipal*	Industrial**
Hydraulic Loading	90,000 gal/acre/day	-
BOD_5	37.5 lb/acre/day	25 lb/acre/day
Nitrogen Loading	15 lb/acre/day	-
Distance to Residence	500 ft.	500 ft.
Distance to Water Supply Well (Public)	1000 ft.	1000 ft.
Depth to Water Table	5 ft.	4 ft.
Depth to Bedrock	10 ft.	10 ft.

* Based on Wisconsin Administrative Code NR 206 and 110
** Based on Wisconsin Administrative Code NR 214

One of the more difficult design tasks is selecting soil hydraulic loading rates which are compatible with treatment objectives. Hydraulic failures (due to soil clogging from organic mats and/or insufficient infiltration capacity) are the most common types of system failures (Reed 1985, WDNR 1983). Based on a review of RI systems, Reed et al. have concluded that the primary basis for system design should be field test results. To avoid hydraulic failure, they made the following conclusions:

a. Field testing at the actual site and planned depth is imperative (soil borings, pilot infiltration tests, in-situ permeability tests, etc.)

b. Construction on backfill should be avoided.

c. Clayey sands with more than 10% clay content are unsuitable for use as fill material in the infiltration area.

d. Construction activities for infiltration areas should only be permitted when the soil is on the dry side of optimum moisture content for all fill (to reduce compaction).

e. For soils with more than 10% clay or silt content, mixing additional sand, gravel, lime or sawdust to increase infiltrative capacity was not successful. Sealing or clogging of the basin surface resulted from the resorting and redistribution of fines during flooding. However, stabilization with grass cover has proven effective.

f. Where significant amounts of algae or industrial wastes with high solids content are expected, the design should be based on infiltration tests with the actual wastewater.

Wastewater is applied to the bottom surface of the basins via single outlets or distribution pipes. Infiltration is then limited by the soil capacity and/or an organic mat which develops over time (depending on the type of wastewater). Two important physical parameters in the soil treatment process are: 1) soil infiltration capacity; and 2) residence time in the unsaturated zone.

Infiltration from a flooded surface as a function of time generally decreases with increasing time to an ultimate steady value, called the infiltration capacity (Childs, 1969). Infiltration into previously drained soils (with some air filled pores) gives rise to unsaturated flow conditions (Bouma 1975).

The theory of water movement in partially saturated soils has been described by several authors (Bear 1970, Childs 1969, Hillel 1970 and 1980, Freeze and Cherry 1979). Aspects of the theory pertinent to land application concepts are presented in the following discussion.

Similar to saturated flow, movement through an unsaturated soil is governed by Darcy's Law with the provision that the hydraulic conductivity is a function of the soil moisture content (or tension), thus

$$q = -K(\theta)\ \text{grad}\ \phi \tag{1}$$

where
- q = specific discharge [L/T]
- $K(\theta)$ = hydraulic conductivity of the porous medium [L/T]
- grad ϕ = hydraulic gradient [dimensionless]
- θ = moisture content

The soil moisture content is a function of the soil water tension (negative pressure) in the porous medium (Freeze and Cherry 1979). For unsaturated flow the hydraulic head includes both suction and gravitational components. With z positive downward,

$$\phi = -(h+z) \tag{2}$$

where
- ϕ = hydraulic head [L]
- h = soil moisture tension head = p/γ[L]
- z = elevation head [L]
- p = fluid pressure (tension) [FL^{-2}]
- γ = specific (unit) weight [FL^{-3}]

The hydraulic gradient is then (for one dimensional flow):

$$\partial\phi/\partial z = -\ (h+z)/\partial z = -(\partial z/\partial z + \partial h/\partial z) = -(1+\partial h/\partial z) \tag{3a}$$

$$\text{and } q = K((\partial h/\partial z) + 1)$$

In the case of a uniform wetting front, the specific discharge can be expressed by the relation:

$$q = K((dh/dz)+1) \tag{3b}$$

In an initially dry soil the soil moisture tension gradients (dh/dz) caused by a wetting front constitute a significant moving force (many times greater than the gravitational force (Hillel 1980). However, if the soil moisture tension below a system is uniform with depth (i.e. $dh/dz = 0$), then the hydraulic gradient reduces to unity and $q=K$.

Combining the Darcy equation, (3a), with the continuity equation for the volumetric soil water content, θ, leads to the Richards equation for 0 (Dagan 1983):

$$\theta/\partial t = -\text{grad}\bullet q = -\partial/\partial z(K_{(\theta)}(\partial h/\partial\theta)(\partial\theta/\partial z)) + \partial K_{(\theta)}/\partial z \tag{4}$$

Generally the $K(\theta)$ and $h(\theta)$ relationships needed to solve this equation are determined empirically. Soil moisture tension vs water content curves and hydraulic conductivity vs soil moisture tension curves have been developed for major soil types by Bouma (1975). These are presented in Figure 2.1. Although these curves provide valuable information on the general characteristics, every soil type exhibits unique characteristic curves. If in the design of a land treatment system it is important to know these relationships they should be determined for the particular soil types.

Methods for both field and lab testing of hydraulic conductivity in partially saturated soils are described in detail by Olson and Daniel (1985). Other methods for infiltration testing are described in the EPA Process Design Manual and Supplement for Land Treatment of Municipal Wastewater (1984).

The second key physical parameter in the soil treatment process is the residence time of the wastewater in the unsaturated soil. A simple method for determining the time of travel for a conservative solute through the unsaturated zone is presented by Hillel (1980).

$$tr = L\theta_r/K(\theta_r)$$

where tr = average residence time [T]
L = vertical distance from pond bottom to the groundwater table [L]
θ_r = volumetric water content at specific retention (drained soil water content)
$K(\theta_r)$ = unsaturated hydraulic conductivity at θ_r [L/T]

This equation is for flow impelled by gravity alone (uniform moisture content) at a velocity equal to the average linear velocity. The drawback in this approach is that it assumes that the transport of solutes occurs by uniform convection alone (ignores spatial variations due to heterogeneities and to water content). Solute transport is also controlled by diffusion, hydrodynamic dispersion, and chemical reactions. To quote Hillel, "solutes resemble a group of rather rowdy passengers (on a train) who constantly move from car to car and occasionally jump off the train entirely while others join in their stead". While it is important to understand these mechanisms, they are difficult to model for all wastewater constituents. Historically, the approach taken for designing land application systems is to identify the limiting design parameters (i.e. BOD_5 and nitrogen) and then design and operate systems to create the desired conditions for removal via biological reactions.

2.3 Treatment in the Unsaturated Zone

Beyond hydraulic considerations, soil treatment capabilities are more difficult to estimate. The optimal operating range for absorption pond systems is one that will produce the best combination of removal for the four constituents: BOD_5, phosphorus, ammonia and nitrate. Actual removal of each constituent will vary from site to site.

The movement of nonvolatile, degradable organic compounds through the unsaturated zone is affected by both the physical adsorption (filtering) capacity of soils and the biological oxidation by soil microorganisms. Laboratory and field studies showed that about 80% of the total organic carbon was removed from sewage water in rapid infiltration systems (Lance 1984). At slower rates, degradation (and removal) of the carbon would be even greater, depending on the concentrations in the wastewater.

The removal of nitrogen in land treatment systems is complex due to the many potential forms of nitrogen (organic N, NH_3, NH_4, NO_2, NO_3, and N_2) and interactions in the soil which cause changes from one oxidation state to another (Reed 1984). The chemical and biological reactions which result in these changes, however, can be used to remove nitrogen from wastewater by proper design and operation of land treatment systems. These reactions result in temporary storage of nitrogen in the soil and removal of nitrogen from the soil (Lance 1984).

The forms of nitrogen typically present in wastewaters are ammonium (NH_4), organic nitrogen, and nitrate-nitrogen (NO_3). The organic nitrogen fraction (usually associated with particulate matter) can be filtered out during the infiltration process. The NH_4 fraction can be lost by volatilization or taken up by plants. NH_4 also can be removed by adsorption onto cation exchange sites, fixation by clay minerals or organic matter, or incorporation into microbial tissue. These processes are generally temporary because the retained NH_4 is easily oxidized to nitrate (nitrification) under anaerobic conditions (Reed 1984).

Temporary storage of NH_4 via cation exchange is common in land treatment systems and dependent on the amount of clay and organic matter in the soil. The CEC may range from 1 meq to 2 meq/100g of soil for very sandy soil to more than 100 meq/100g for soils high in clay or organic matter or both. The importance of the CEC of soils in the overall design of absorption pond systems is discussed later in this section.

Nitrate is very mobile in the unsaturated zone because of its anionic form and because it is not limited by solubility constraints in the concentration ranges typical of wastewaters (Hensel 1984).

Nitrate is converted to nitrogen gas by denitrification. Biological denitrification is a reaction capable of removing large quantities of nitrogen in soils receiving wastewater under the right conditions. This is the desired reaction for nitrogen removal. Four conditions required to denitrify percolating wastewater are:

1) oxidation of NH_4 to NO_3 (nitrification)
2) passage through a reducing zone after oxidation;
3) provision of adequate energy source (carbon) in the reducing zone; and
4) favorable temperatures, pH and sufficient CEC.

The interrelationships of these conditions have been the subject of much research, particularly with RI systems receiving domestic wastewater. Ideally, systems should be operated so that both nitrification and denitrification are optimized.

Several authors propose that appropriate flooding and drying schedules will provide nitrogen removals from 30% to 80% (Lance 1972, Leach 1983, Bouwer 1974, Enfield 1977, Bennett 1985). The suggested lengths of flooding and drying periods depend on the form of nitrogen in the wastewater and on the amount of organic carbon available in the water or soil.

When the nitrogen is mostly in the ammonium form and the organic carbon levels are low, the effluent should be applied for a sufficiently long period to cause oxygen depletion in the soil. The ammonium can no longer be converted to nitrate once oxygen depletion occurs and then can be held by the cation exchange sites (Bouwer 1974). Application of the wastewater should be stopped before the soil is saturated with ammonium. A simple calculation to predict the amount of NH_4 which can be held up in the soil can be made from the CEC of the soil and the concentrations of the principal competing divalent cations (Lance 1972).

Once the soils are allowed to drain dry, oxygen entering the pores will cause the adsorbed ammonium to be nitrified. Bouwer (1974) suggests that after nitrification occurs, denitrification will occur in the "micro" anaerobic zones. When the wastewater is applied again nitrate laden capillary water mixes with incoming "carbon rich" water and denitrification occurs once anaerobic conditions are reached. Lance and Whisler reported optimal removal with loading cycles of 2 to 3 weeks (secondary effluent).

If the organic carbon level in the wastewater is high (such as in dairy wastes or animal wastes) the application length can be shorter sufficient levels of carbon are left after the wastewater passes through the aerobic zone for denitrification to occur at depth (Bouwer 1974).

Carbon to nitrogen ratios for optimal nitrogen removal ranging from 1:1 to 2:1 (C:N) have been cited (Enfield 1977, Lance 1976, St. Amant 1969, Rice 1984, and Reed 1984). The lower ratios were cited in studies where carbon sources were methanol or glucose. Reed and Crites (1984) presented the following equation for estimating nitrogen removal:

$$N = (TOC - 5)/2$$

where N = change in total nitrogen [mg/l]
TOC = total organic carbon in the applied wastewater [mg/l]

The 5 mg/l of residual TOC is typical for municipal wastewater after passage through about 5 ft. of soil. The coefficient 2 is based on empirical data where 2 grams of carbon were required to denitrify 1 gram of wastewater nitrogen. However, the use of this relationship assumes optimal conditions for denitrification. The effects of temperature and pH on the nitrification and denitrification processes are well documented. The optimal temperature range for nitrification is between 59•F and 95•F. At 54• nitrification activity drops by 50% (Nazih 1986). The optimal pH range for nitrification is 5 to 9.6. Denitrification activities at temperatures below 59•F decrease to 25% of the activity at 77•F (Keeney 1979). These restrictions preclude the possibility of significant nitrogen removal if soil temperatures drop below 60•F.

Phosphorus removal in soils can occur by adsorption and/or precipitation. Bouwer (1974) reported removal of about 50% after 30 ft. of movement and 90% removal after 100 ft. of movement. An empirical equation to predict phosphorus removal at RI sites is presented by Reed and Crites (1984). Reed and Crites (1984) also report phosphorus removals to near background levels for natural groundwater for 11 high rate and slow rate application sites with sand soils or finer. They reported that 3 RI systems which were discharging to gravelly sands did not provide adequate phosphorus removal.

2.4 Effects on the Groundwater Flow System

Wastewater percolate from absorption pond systems can affect both the groundwater flow patterns and groundwater quality.

The presence of a groundwater table or impeding layer at depth below a system will cause the formation of a groundwater mound or perched mound. Flow in the mound region is governed by a combination of vertical and horizontal hydraulic conductivities. The capability for lateral flow away from the application site along with the loading rate, size of loading area and saturated thickness controls the extent of mounding that will occur beneath absorption ponds.

Estimation of mound height is critical in the design of absorption pond systems, since the presence of a mound reduces the thickness of the unsaturated one. Hantush (1967) and Glover (1961) present the theoretical basis of mound height predictions for shallow aquifers where the Dupuit Forcheimer assumptions apply. The EPA Process Design Manual and Supplement presents a simplified method for mound height analysis.

2.5 Solute Transport in the Groundwater

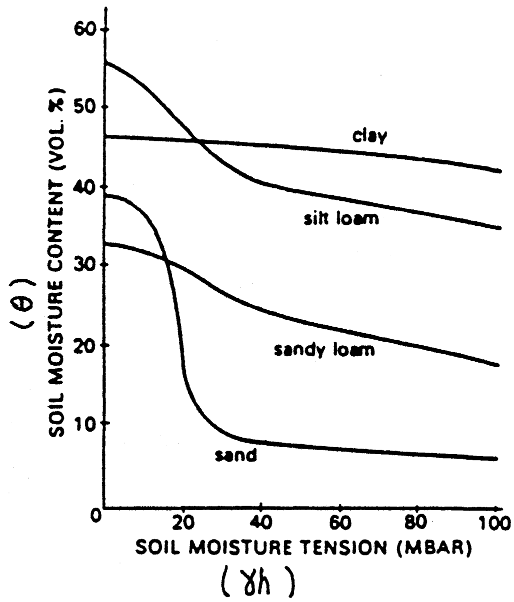
Wastewater constituents that are not removed in the unsaturated soil below absorption ponds will reach the groundwater. In the saturated zone solutes are transported by the physical processes of advection and dispersion. Advection of solutes occurs with the flowing groundwater. Dispersion occurs as a result of mechanical mixing and molecular diffusion (Freeze and Cherry, 1979). Thus, percolating effluent which reaches the groundwater will move in solution with the groundwater and will disperse and diffuse in the direction of decreasing concentration gradients.

The distribution of wastewater (effluent) in the groundwater is characterized as a contaminant plume. The concentration of contaminants will be highest in the center of the plume and decrease toward the edges. The shape of the plume depends on the loading rates of the groundwater flow velocities and the geologic heterogeneity of the aquifer.

Field studies have shown that the influence of dispersion on contaminant concentrations increases with distance travelled (due to increasing scales of heterogeneity) [Molz 1983]. Therefore, when considering short distances, dispersion often may be negotiated with the major emphasis placed on determining true flow paths (Hensel 1984).

Studies of groundwater impacts from absorption pond systems report varying levels of contaminant influence. In all cases the quality of percolate water was not as good as that of native groundwater. Once the percolate reaches the groundwater, dilution by the process of dispersion and mixing is the primary mechanism which will reduce the levels of conservative constituents to acceptable concentrations at a distance from the source.

Since groundwater is typically devoid of organic carbon, once nitrogen in the form of nitrate reaches the groundwater it is usually not attenuated other than by dilution (Bouwer 1976).



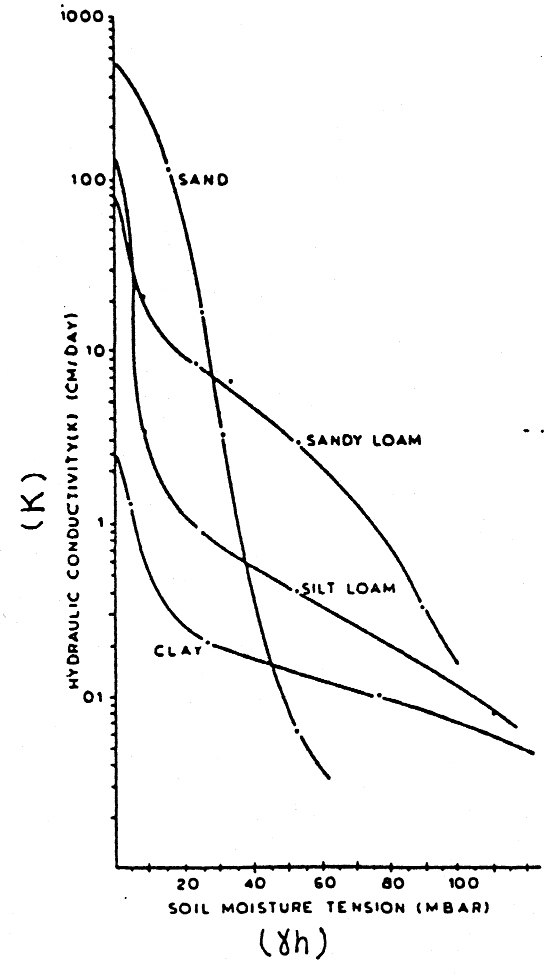


FIGURE 2.1
SOIL MOISTURE CHARACTERISTIC CURVES
(FROM BOUMA 1975)

SECTION 3 - MATERIALS AND METHODS

3.1 Study Design

The field investigation portion of this study focused on the performance of two absorption pond systems in Wisconsin. One received effluent from the wastewater treatment plant which serves the City of Evansville, and the other received raw washwater wastes from the Brunkow cheese factory near Darlington, Wisconsin. Site location maps are presented in Figures 4.1 and 5.1 respectively. The systems were selected by Wastewater Management staff members of WDNR as representative of typical existing systems in Wisconsin. The following variables were considered in the selection of each system:

- Soil type and geology;
- Depth to groundwater;
- Age of system;
- Operational flexibility (number of cells for resting/loading);
- Size of system and;
- Wastewater characteristics.

It is desired to select systems that were characteristic of the existing design standards. This allowed an assessment of performance in relation to the system design requirements (listed in Section 2, Table 2.1). The specific variables of interest at each site were:

1. The hydraulic, organic treatment and operational capabilities of each system;

2. The removal of nitrogen and other wastewater constituents in the unsaturated zone;

3. The extent of wastewater impact on the groundwater below the system and at the site boundaries;

4. The monitoring network employed at each site.

The two sites selected were very different both in system design and in the characteristics of subsurface conditions. Several of the major differences are summarized below:

<u>Evansville</u>	<u>Brunkow</u>
Treated domestic wastewater	Dairy wastewater
Low strength/high rate application	High strength/low rate application
14 to 16 ft. to water table	5 to 8 ft. to water table
Ave. discharge 300,000 gpd	Ave. discharge 3,000 gpd

It was not a major goal of this study to compare performances of the two systems due to their significant differences. It was, however, important to quantitatively and qualitatively understand and explain how each system performed. The geologic and wastewater conditions of each site governed how each monitoring network was set up and how the field work was conducted.

General instrumentation and field methods are presented in the following discussion. Details of individual system design and site characteristics are described in Sections 4 and 5 for Evansville and Brunkow, respectively.

3.2 Field Methods and Materials

The monitoring networks were set up to: 1) characterize the groundwater quality upgradient, below and downgradient of each system; 2) determine vertical and horizontal groundwater flow components; 3) determine vertical flow characteristics in the unsaturated zones; and 4) determine nitrogen transformations and BOD removal in the unsaturated zone pore water.

The networks included water table wells and piezometers in the saturated zone and lysimeters and tensiometers in the unsaturated soils. A schematic of the subsurface monitoring network is shown in Figures 4.2 and 5.2 for Evansville and Brunkow, respectively.

During October through December 1984, 2 inch ID schedule 40 PVC monitoring wells were installed at both sites. These wells were used for groundwater sampling and measuring groundwater elevations. The wells were located based on preliminary estimates of travel time, distance requirements for application of groundwater quality standards list in Wis. Adm. Code, NR 140, and physical site constraints. Additional wells were installed in May 1985 at Brunkow and in May and October 1985 at Evansville to define the plume characteristics and extent further.

Boreholes for the wells were drilled with 6 inch diameter continuous flight, solid stem augers. Water table wells were installed with 5 ft. screen lengths and piezometers were installed with 2.5 ft. screen lengths (.006 inch slot width). A typical well installation schematic is presented in Figure 3.1. For the most part, these specifications were followed for the wells at Brunkow. Silica sand was not used at Evansville since the soils were sandy and caved in around the screen when the casing was installed. Well installation details for Evansville are listed in Appendix C.2 and in Appendix D.1 for Brunkow. Elevations were taken from the top of the well casing, tied in with a standard USGS Datum and used during the study to convert measured water levels to elevations.

Some problems were encountered during well installation at Evansville. The presence of large cobbles in the soil prevented advancement of the auger to the planned depth at certain well locations. A second problem was that boreholes would collapse before the PVC well could be installed. In the sandy soils at the Evansville site, use of hollow stem augers would have been a better drilling method.

Problems encountered during drilling at Brunkow were associated with the shallow depth to bedrock. Two bedrock wells were installed (Well 3 and Well 6). Well 3 was drilled with solid stem augers but it was a slow process. Well 6 was drilled using hollow stem augers with a special bit for drilling in bedrock. Clear water and a 6" diameter casing seated at the bedrock location was used during drilling. Water for drilling was pumped from the creek near the site.

Vacuum pressure lysimeters were installed in the spring of 1985 at the Evansville site. Teflon cup lysimeters from Timco Mfg. Inc. were used. Figure 3.2 shows a typical installation. The lysimeters were used to draw samples of soil pore water at depths of 2.5 ft., 5 ft. and 8 ft. below the seepage cells in regions where the wastewater percolated. A plan view of lysimeter locations is presented in Figure 4.2

To install the lysimeters, 6 inch or 3 inch diameter hand augers were used to drill a bore hole to the desired depths. The lysimeters were carefully placed in the boreholes, and a silica slurry was used to maintain hydraulic connection between the porous teflon and the surrounding soils. The silica pack was allowed to harden with minimal disturbance of the lysimeter casing. The borehole was backfilled with bentonite followed by natural soil and a final layer of bentonite to grade to prevent channeling of wastewater down to the lysimeter.

Lysimeters were not installed at the Brunkow site. This decision was made based on results from the preceding ridge and furrow study (Doran 1985) and limitations regarding use of lysimeters in general. Due to the low permeability soils, sample collection times of up to 1 month were expected to obtain a sufficient sample. This length of sample collection time would have been unacceptable for the parameters of interest (due to chemical transformations within the lysimeter sample reservoir.

Tensiometers were installed in June 1985 at both sites at depths of 1, 2, 3.5 and 5 ft. below the pond bottoms. The tensiometers were "jet fill" type developed by Soil Moisture, Inc. They were used to measure soil moisture tension which is reflective of the wetness or soil moisture content. A schematic of the relative placement of nested tensiometers and lysimeters in the unsaturated zone beneath the Evansville absorption ponds is presented in Figure 3.3.

3.3 Sampling and Analytical Methods

The sampling program, which was initiated in October 1984 at Brunkow and December 1984 at Evansville, was developed to provide an adequate amount of information regarding system performance for the 15 month study.

After wells, lysimeters and tensiometers were installed, monthly field visits were made to each site. During each visit wastewater, lysimeter (Evansville only), and groundwater samples were collected, groundwater elevations were recorded, soil tension measurements were recorded and site observations were made. During some site visits, when conditions were favorable, pond infiltration estimates were made via staff gauges at Evansville. Staff gauges at Brunkow were read at each site visit. Infiltration rings were placed on the inside slope of the berms for Ponds 2 and 3 at Brunkow.

As data was reviewed during the study, changes were made in the sampling schedule. The number of parameters tested for was reduced at certain wells where results were predictable.

At Evansville during a 3 week period from June 17 through June 28, 1985, an intensive sampling program was conducted to determine more accurately the variability of soil pore water quality and shallow groundwater quality as function of the wastewater loading schedule. During this period wastewater and wells located below and immediately downgradient were sampled 4 times per week. Lysimeters were sampled 2 times per week. During this period and throughout the summer, continuous water levels were monitored electronically in shallow well 5S between the two seepage cells.

Details of the field and analytical sampling methods are presented in the following discussion.

Soil samples were taken from the auger during well installation at both sites. The samples were taken at depths where visible changes in soil types were observed and at 5 or 10 ft. increments at Evansville where no observable changes in soil type were noted (with the exception of well 11). The soil samples were sealed in plastic bags and analyzed by the Soil Plant Analysis Laboratory, University of Wisconsin Extension. All samples were analyzed for total nitrogen, CEC, pH and % organic matter. Samples from Brunkow were analyzed for % sand, silt and clay. Samples from Evansville were analyzed for grain size distribution and % P200 (finer than 200 mesh). Results of the soils analyses are listed in Appendix A.3 for Evansville and in Appendix B.2 for Brunkow.

In November 1985 an additional 21 soil samples were taken from the southwest seepage cell at Evansville and analyzed for % sand/silt/clay, CEC, % organic matter, moisture content and dry bulk densities. Bulk densities were determined on samples taken with a drive cylinder following procedures described in the 1985 ASTM Standards manual, Section 4 on Construction, Volume 0.408 for Soil and Rock; Building Stones, method D2937. Moisture contents were determined using the ASTM D2216 procedure. The samples for moisture content were taken within 2 ft. of the tensiometers and at corresponding depths. This placement was done to correlate moisture contents with in-situ readings of soil moisture tension. Results of the dry bulk density and moisture content determinations are listed in Appendix A.3.

Slug tests were done at both sites at selected wells. These provided in-situ estimates of hydraulic conductivity. The method used to interpret the slug test data was developed by Hvorslev. Details of the method used and results are presented in Appendix C.1 for Evansville and in Appendix D.1 for Brunkow.

Staff gauges were used for infiltration tests in the absorption ponds at both sites. The staff gauges were read over time when discharge to the cell had ceased. Additional infiltration measurements were made at Brunkow using infiltration rings constructed with stove pipe material.

Wastewater flow estimation and sampling procedures were tailored to each site and are discussed in Section 4 for Evansville and in Section 5 for Brunkow. The wastewater samples were tested for pH and temperature in the field but were not field filtered. Nutrient (nitrogen and phosphorus) and COD samples were acidified in the field with sulfuric acid to $pH<2.0$. Metals samples were acidified in the field with nitric acid to $pH<2.0$.

Prior to the initial sampling of lysimeters at Evansville, a volume of water equivalent to 30% of the original volume of deionized water used to mix the silica slurry was removed and discarded. This volume, recommended by the manufacturer, was intended to replace the pore water in the silica pack with ambient pore water (Morrison 1983). Volumes of water removed each time were recorded and electrical conductivity measurements were made on the pore water. This measurement served as an added check on the 30% removal recommendation. When the measured conductivities reached levels near those in the wastewater and remained nearly constant, sampling of the lysimeters was initiated. This process took approximately 5 weeks.

Several problems were encountered in the initial sampling efforts from the lysimeters. Four out of nine failed to work properly at first. Three had to be reinstalled. The major problem was thought to be an inadequate seal between the silica pack and the teflon cup.

The lysimeters were sampled by applying the recommended vacuum of 20 inches Hg and typically returning one to two days later to retrieve the accumulated sample. Samples were withdrawn by forcing the collected water up to the ground surface using a positive pressure hand pump.

Groundwater samples were taken approximately monthly. Standing water was purged from the wells and discarded prior to sampling. At Evansville, due to the rapid recovery, 3 volumes of the well were purged to assure complete removal of standing water. All sampling and bailing was done with a PVC 1.25 inch bailer. Early in the study an ISCO sampler (bladder type) with a compressor was used for sampling. This method was deemed cumbersome and slower than bailing. Sampling was begun upgradient at less contaminated wells and proceeded to downgradient wells. All sampling equipment was rinsed with deionized water between wells. Sampling blanks were taken several times by pouring deionized water through all equipment and following standard sampling procedures, including filtering. No contamination was detected in blank samples submitted to the Lab.

Samples were usually field filtered through 0.45 micron filter paper using a Masterflex peristaltic pump and a geofilter pressure filter stand. Temperature and pH measurements were also made in the field. All samples were acidified with the appropriate acid to $pH>2.0$ and transported on ice to the State Laboratory of Hygiene.

Twice during the study samples were taken from the stream at Brunkow both upgradient and downgradient of the absorption ponds.

Chemical analysis of wastewater, lysimeter, groundwater and stream samples were performed by the Wisconsin State Laboratory of Hygiene. A list of parameters and frequency of sampling is presented in Table 3.1.

Table 3.1
Chemical Analysis Sampling Frequency*

Parameter	Wastewater	Lysimeter	Groundwater
BOD_5, tot	1	2	-
BOD_5, diss	3	2	3
COD, tot	3	2	-
COD, diss	-	-	1
TSS	1(B)	-	-
TDS	1(E)	-	1
TKN, tot	1,2	2	-
TKN, diss	3	-	1,2
NH_3-N, NO_3-N	1,2	2	1,2
Cl-	1,2	2	1,2
pH	1,2	2	1,2
Alk, tot	1	-	-
Alk, diss	-	-	1
Hardness as $CaCO_3$	1	-	1
Phosphorus, tot	3	-	3
SO_4, tot	3	-	-
SO_4, diss	-	-	3
NA^+, tot	3	-	-
NA^+, diss	-	-	3
CA^+, tot	3	-	-
CA^+, diss	-	-	3
Mg^+, tot	3	-	-
Mg^+, diss	-	-	3
K^+, tot	3(B)	-	-
K^+, diss	-	-	3(B)
Coliform bacteria	3(E)	-	3(E)

*Frequencies: 1 = monthly
2 = weekly during selected summer months
3 = periodically
E = Evansville only, B = Brunkow only

Note: Stream samples taken at Brunkow were analyzed for all of the above.

Complete laboratory procedures and analytical detection limits are described in the Manual of Analytical Methods - Inorganic Chemistry Unit, written by the State Laboratory of Hygiene in 1980.

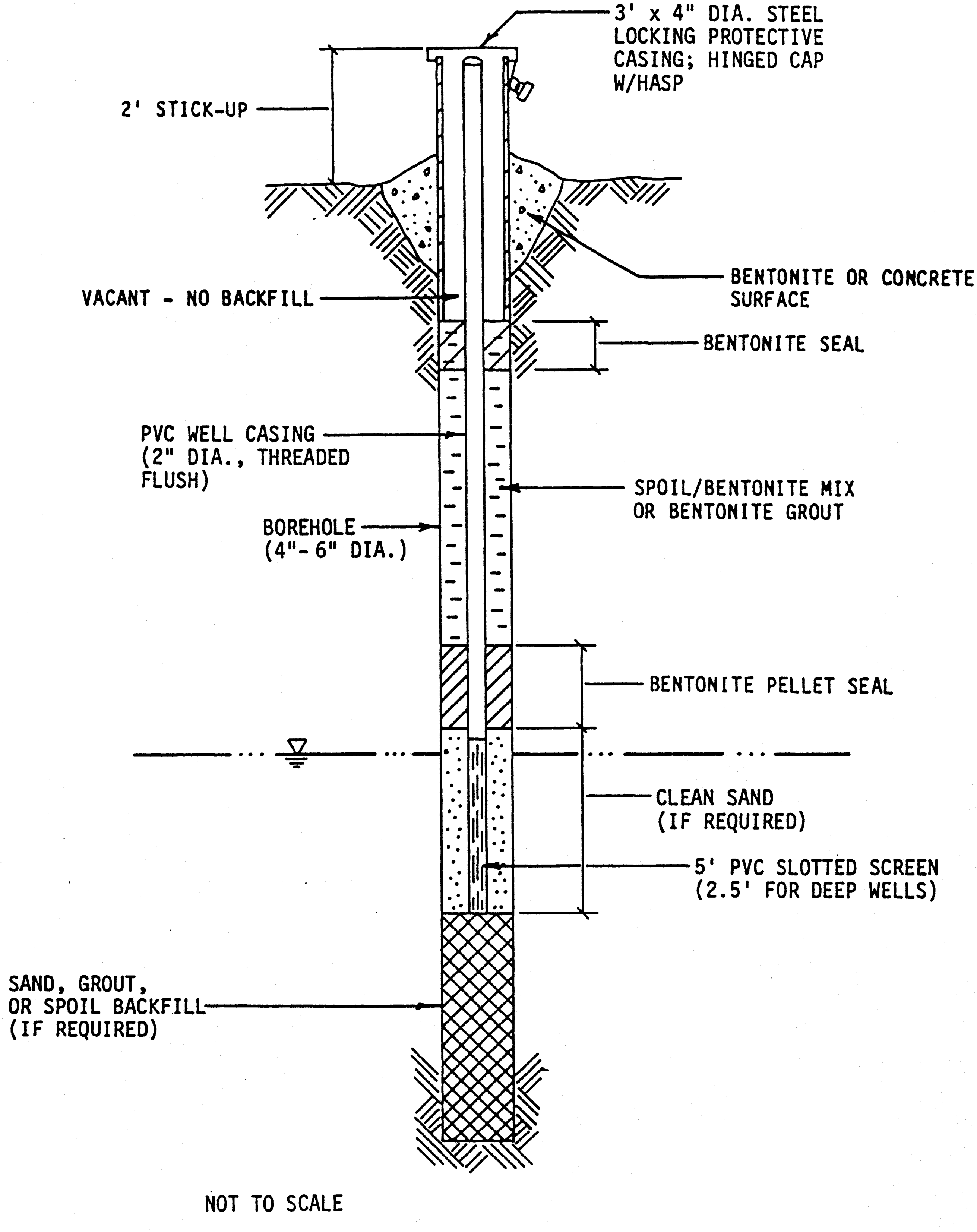


FIGURE 3.1
TYPICAL MONITORING WELL SCHEMATIC

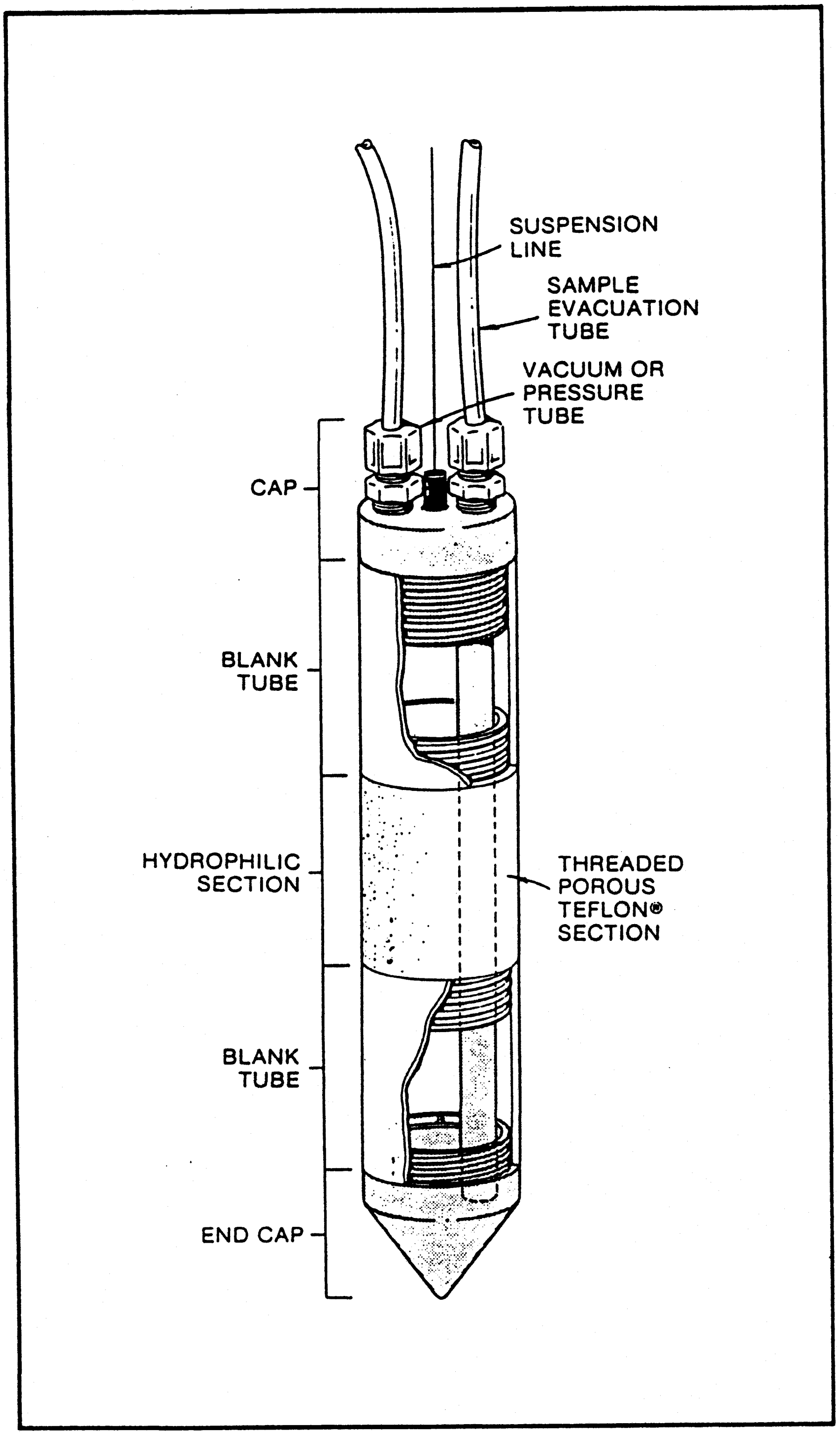


FIGURE 3.2
TYPICAL SUCTION LYSIMETER DETAIL (TIMCO MFG., INC.)

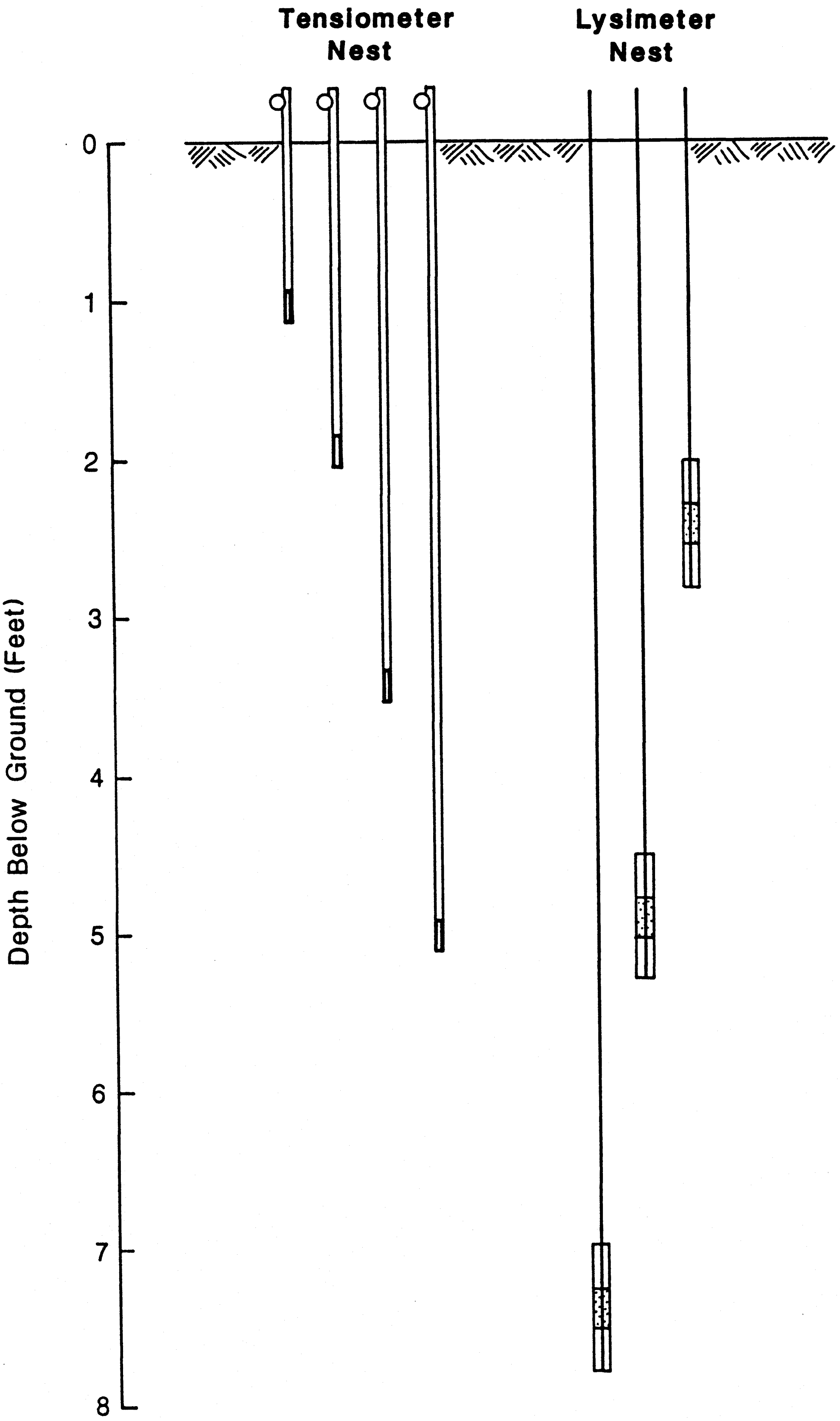


FIGURE 3.3
TYPICAL TENSIOMETER AND LYSIMETER NEST
EVANSVILLE RAPID INFILTRATION SYSTEM

SECTION 4 - EVANSVILLE-PRESENTATION AND DISCUSSION OF RESULTS

4.1 System and Site Description

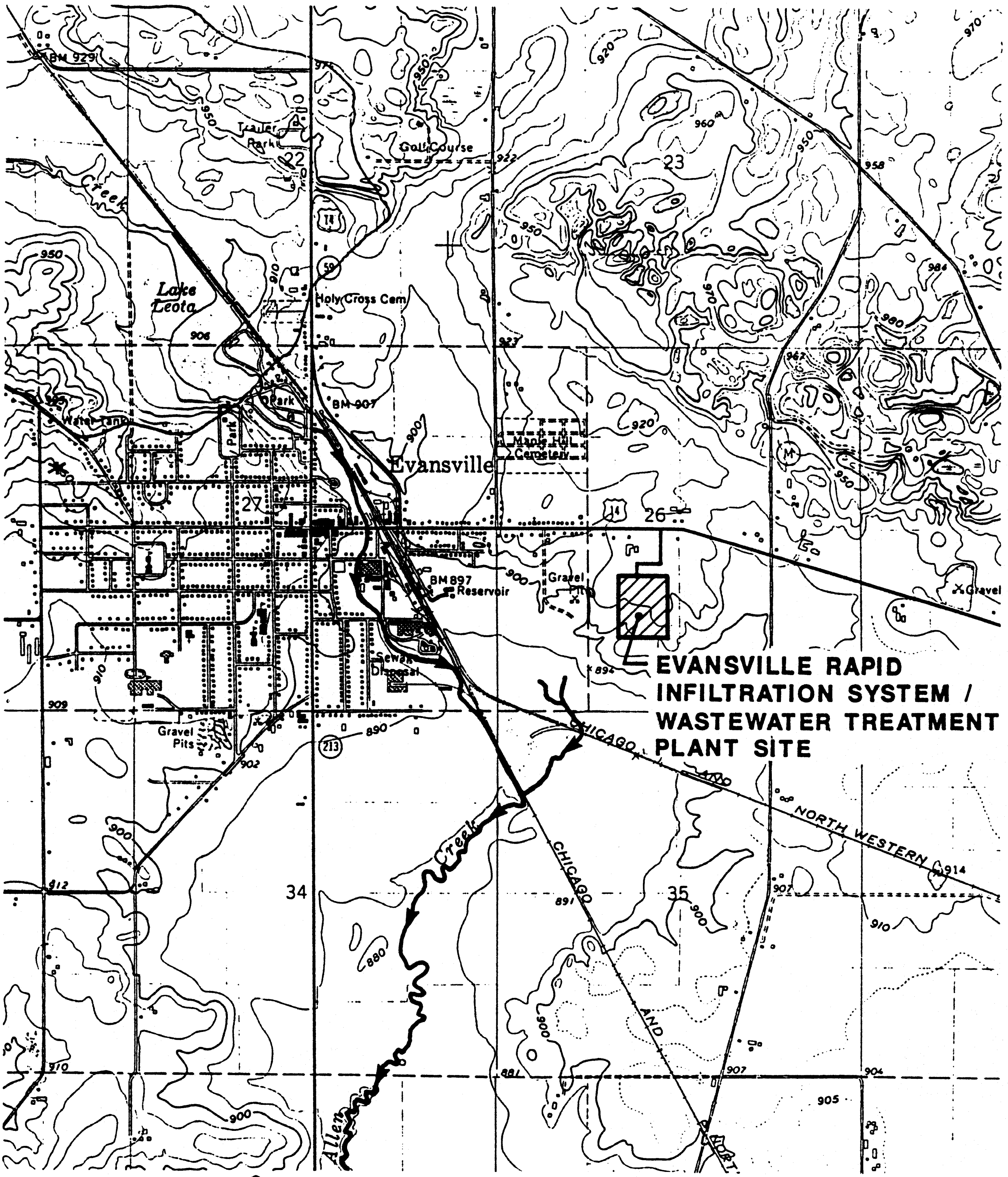
The City of Evansville's existing wastewater treatment plant was constructed in 1983 and is located east of the city on Hwy 14. A location map for the site is presented in Figure 4.1. The treatment plant consists of two aerated lagoons operated in series followed by a holding pond. The system was designed to discharge to four seepage cells (absorption ponds), each approximately 1.6 acres in area at a rate of 600,000 gpd (design capacity). During this study the plant was operating at approximately half its design capacity. Discharge to only two of the four cells allowed operation of the land disposal portion of the system at levels near the design loading rates. Figure 4.2 shows a plan view of the system.

During the period of study from October 1984 through March 1986 only the northwest (NW) and southwest (SW) cells were used. Average flows to the system were 310,000 gpd. For the total area of 3.2 acres that was used, the loading rate on a per acre basis was 96,875 gal/acre/day. Effluent was discharged from the holding pond to the seepage cells by gravity flow. The effluent was distributed over the length of each cell by a single 2 ft. diameter cast iron distribution pipe.

The seepage cells were constructed on a sand and gravel outwash plain that was determined to be at least 70 ft. thick in the area immediately below the seepage cells.
Groundwater was approximately 15 ft. below the bottom of the cells. A schematic cross section of the site is shown in Figure 4.3. Soil borings taken prior to construction of the site indicated that the outwash material is a fairly homogenous well sorted sand and gravel deposit with a few minor bands of higher silt content soils in the area where the southeast cell is located. These silt layers were removed during the construction phase. In November 1985, 21 soil samples were taken from the SW cell. These showed the following characteristics: 87-97% sand; 0-8% silt; 2-5% clay; 5-10.2% moisture (dry weight basis); 1-12 centibars tension; 0-6 meq/100g CEC; and .1-.2% organic matter.

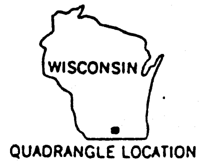
It was observed that the CEC for soils within 1-2 ft. of the surface of the bed was 3 meq/100g or less, and from depths greater than 3.5 ft. the CEC was in the 5-6 meq/100g range. Grain size analyses were performed on 13 soil samples taken from well borings at depths ranging from 20 to 75 ft. below ground level. These analyses showed that all samples contained less than 5% passing a no. 200 sieve with the exception of one sample taken at 75 ft. which contained 10% very fine sand, silt and clay. Detailed soils data are presented in Appendix 8.3.

Ground surface elevations decline at a slope of .05 ft/ft downgradient of the site to a low area 450 ft. south where the depth to groundwater was 3 to 4 ft. below ground level. Soil samples taken during the installation of monitoring well 11 in this area contained 29% sand, 60% silt and 11% clay. The impact on the groundwater flow system as a result of this silty subsurface material is discussed in a later section.



north

Scale: 1" = 2000'



Note: Site location map was developed from the U.S.G.S. 7½ minute quadrangle map, Evansville, Wisconsin, dated 1961, photo-revised 1971.

FIGURE 4.1
SITE LOCATION MAP
EVANSVILLE RAPID
INFILTRATION SYSTEM
CITY OF EVANSVILLE
ROCK COUNTY, WISCONSIN

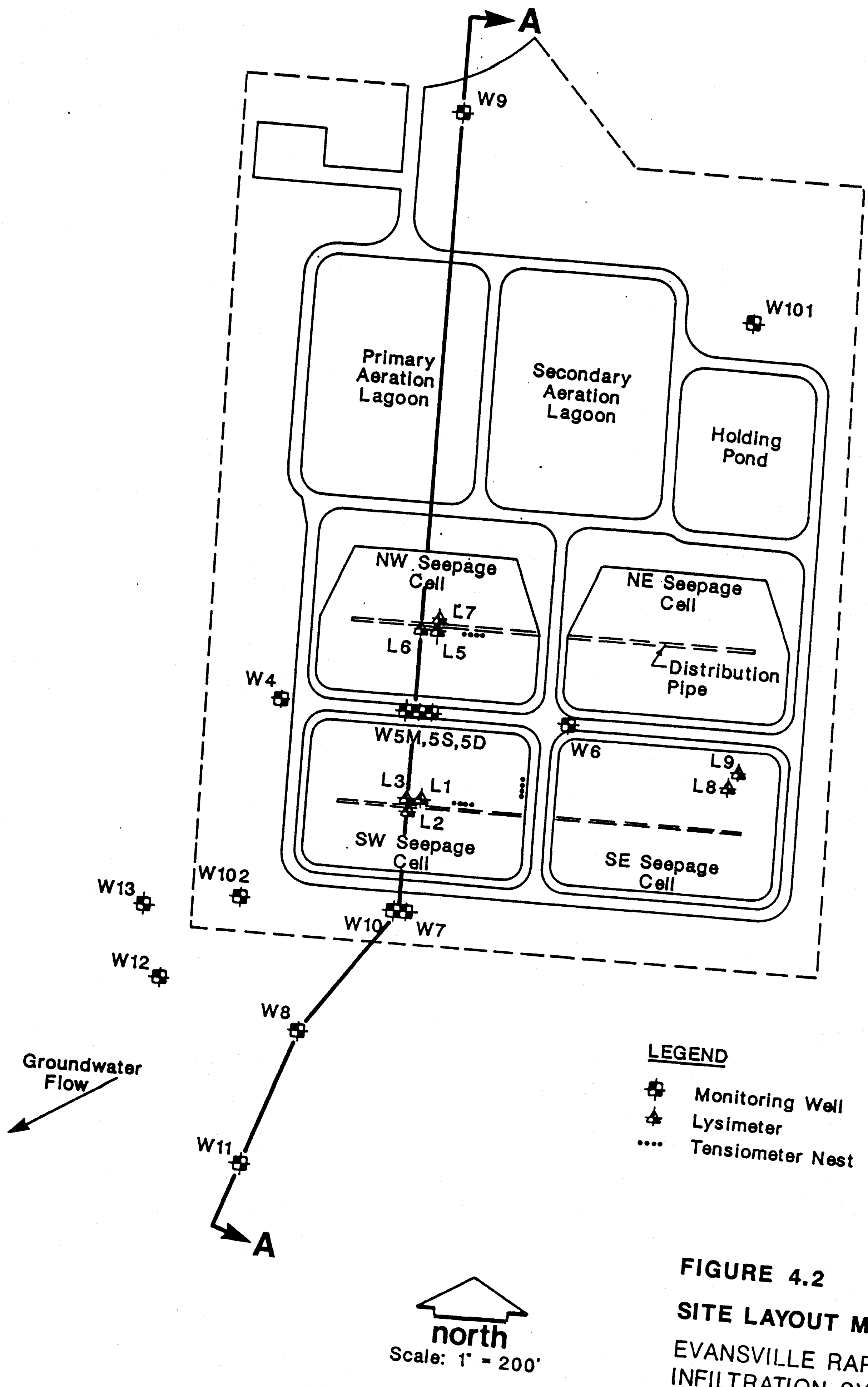


FIGURE 4.2

SITE LAYOUT MAP

EVANSVILLE RAPID INFILTRATION SYSTEM

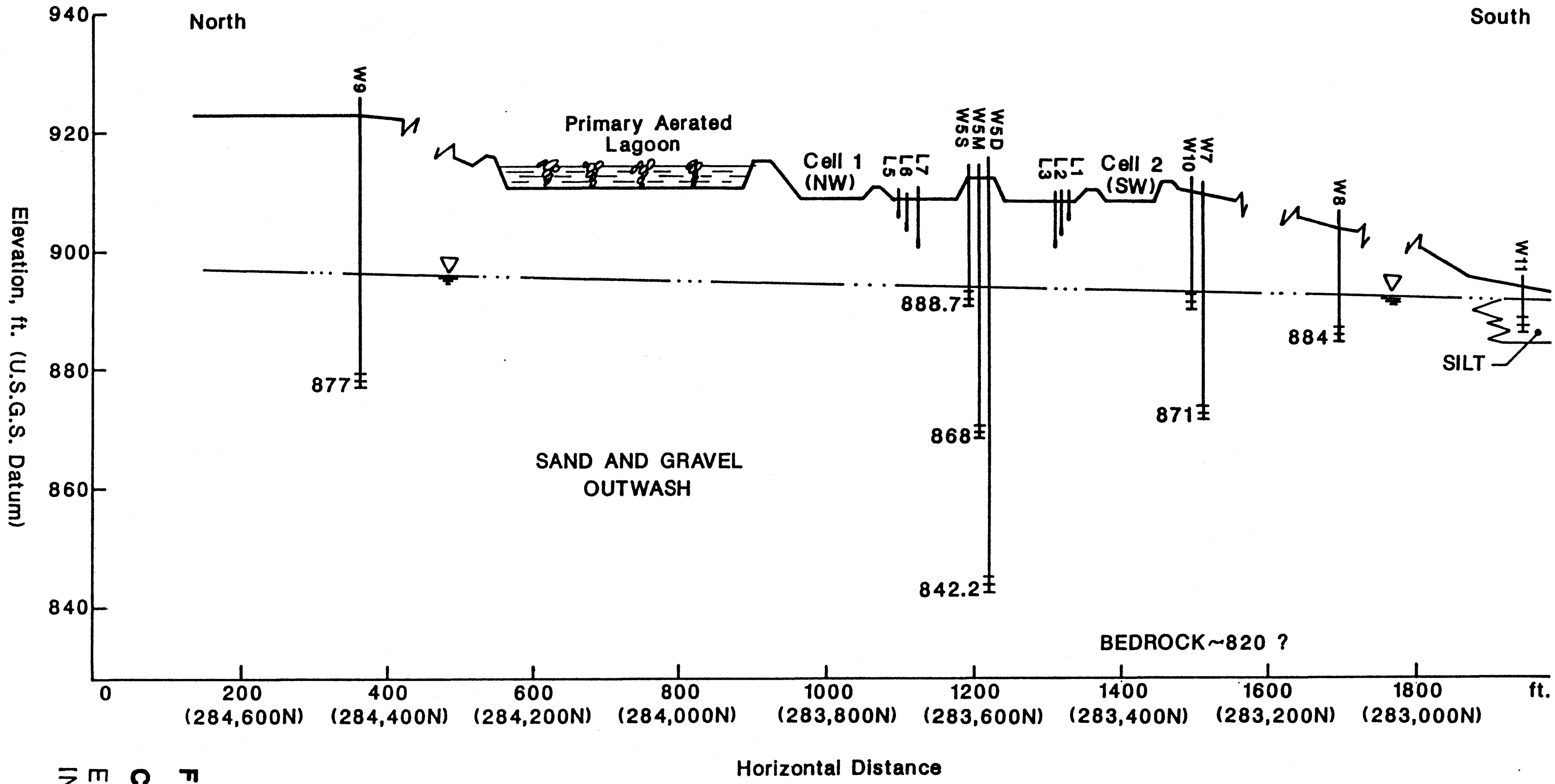


FIGURE 4.3

CROSS SECTION A-A

EVANSVILLE RAPID
INFILTRATION SYSTEM

NOTES

1. Vertical scale is expanded.
2. Water table elevations are based on measurements from December, 1984.
3. Water table wells have 5 ft. deep screens, piezometers have 2.5 ft. screens.

4.2 Wastewater Characterization

The wastewater effluent BOD_5 concentrations ranged from 11 mg/l to 37 mg/l with a mean of 26 mg/l during the period of study. Mean, range and standard deviations of selected parameters are listed in Table 4.1. Raw data for the wastewater are listed in Appendix A.1.

Table 4.1
Evansville Effluent Parameters
(December 1984-March 1985)

Parameter	Concentrations (mg/l) Mean	Standard Dev.	Range
BOD	26	8	11-37
TSS	23	12	8-47
TDS	833	59	734-930
COD	64	12	41-81
Chloride	245	20	210-290
Tot N	20.6	7.6	7.6-3.1
NH_3	12.7	10.8	<.1-28
NO_3	4.5	3.3	.8-10.5
TKN	16.2	10.2	3.4-30

*(BOD and TSS based on 16 samples, others based on 14 samples; Total N = TNO_3+TKN)

In addition to the overall quality of the wastewater the following three observations were made. First, the total nitrogen concentrations decreased by more than 50% during the summer months (a low of 7.6 mg/l as N) from concentrations in the winter months of NOvember 1984 through April 1985 (a high of 30 mg/l as N). This change was attributed to a loss of nitrogen within the treatment system prior to discharge to the cells. This loss most likely results from denitrification in the holding pond.

Secondly, nitrate was the primary form of nitrogen in the wastewater from May through November. This was caused by nitrification of NH_3 within the aeration lagoons with warmer temperatures. From December through April, NH_3 was the primary form of nitrogen in the effluent. Organic nitrogen content ranged from 1 to 6 mg/l as N. The higher levels near 6 mg/l were measured during the summer months when algae production was high.

Finally, total suspended solids (TSS) concentrations were highest in the summer (around 40 mg/l TSS) and dropped lowest levels in the winter (about 10 mg/l TSS). The increase in TSS coincided with increased algae growth in the lagoons and holding pond. It was noted that the increase in suspended solids did not decrease the infiltrative capacity of the seepage cells.

4.3 Loading Rates to the Seepage Cell System

Prior to October 1984, the schedule for loading the seepage cells was to discharge effluent on Mondays, Wednesday and Fridays, alternating between all four cells after each discharge. The operator at the plant indicated that during the winter of 1983-1984 the northeast cell was loaded most frequently. In October 1984, the loading schedule was changed such that only the northwest and southwest cells were used. The cells were still loaded on Monday, Wednesday, Friday basis; however, the new schedule was to load one cell four times consecutively and then alternate to the other cell for the period. This schedule simulated a 10 day rest/load cycle.

The volumes of effluent that were discharged typically ranged from 500,000 gallons to 1,000,000 gallons within 3 to 4 hours (2800 gpm to 5500 gpm). Discharges were measured via a staff gauge at the control manhole located between the holding pond and seepage cells. Levels in the manhole were recorded before and after discharge. The total volume discharged was calculated using a stage-volume curve, developed for the geometry of the holding pond.

The recent history of mass loading rates of BOD_5 to the seepage cells for 1984 through 1985 is illustrated in Figure 4.4. The mass loadings were calculated using monthly averaged discharge volumes and monthly average BOD_5 concentrations. The monthly average BOD_5 are based on weekly grab samples taken by the plant operator and analyzed at the laboratory facility on site. The data indicates that loadings did not increase or decrease substantially over this period and that some seasonal fluctuations existed.

To summarize the loading to the system for 1985, the average concentrations of selected parameters from Table 4.1 were converted to mass loadings using the average daily volumetric flow rate. These values are given in Table 4.2 and are compared with the system design loading (Wastewater Facility Plan 1979).

Table 4.2
Evansville Actual and Design Loading Rates

Parameter	Loading * (1985 ave.)	Design ** Loading
Flow (mgd)	.310	.600
(gal/acre/day)	96,875	93,750
BOD^5 (lb/acre/day)	21	39
TSS (lb/acre/day)	18.6	39
N (lb/acre/day)	14.5	23.5

*Actual loadings for 1985 based on area of 3.2 acres (2 cells)
**Design loading based on a total cell area of 6.4 acres

It should be noted that the design suspended solids and nitrogen loadings listed above were estimated based on expected concentrations for BOD^5 loading and design flow (Facility Plan, 1979). The design of this system was based primarily on the hydraulic capacity of the soils. Therefore, nitrogen and total suspended solids were not considered.

To illustrate the fluctuation in total nitrogen loading to the cells over the period of sampling, average monthly loading rates based on the results of a grab sample taken each month were calculated. These are shown with the BOD^5 loading rates in Figure 4.5.

It was observed early in the study that surface infiltration rates in the seepage cells were rapid. For a typical loading of 750,000 gallons of effluent, the time from when discharge began until no standing water remained in the cells was about 6 hours. Infiltration tests performed during the design of the system resulted in an estimated infiltration rate of 5.2 ft/day (Facility Plan 1979).

Further infiltration tests were performed during the study to verify earlier results and to measure any change in the infiltration rate due to changes in loading patterns. Staff gauges were placed in the cells and used to measure the falling level of the ponded surface just after the cells were loaded. This was the rate of drainage of the ponded effluent within the cells. The infiltration measurements ranged from 4 to 9 ft/day (2 to 4.5 in/hr). Due to these rapid infiltration rates, it was difficult to maintain ponded conditions for the appropriate time required for the denitrification process.

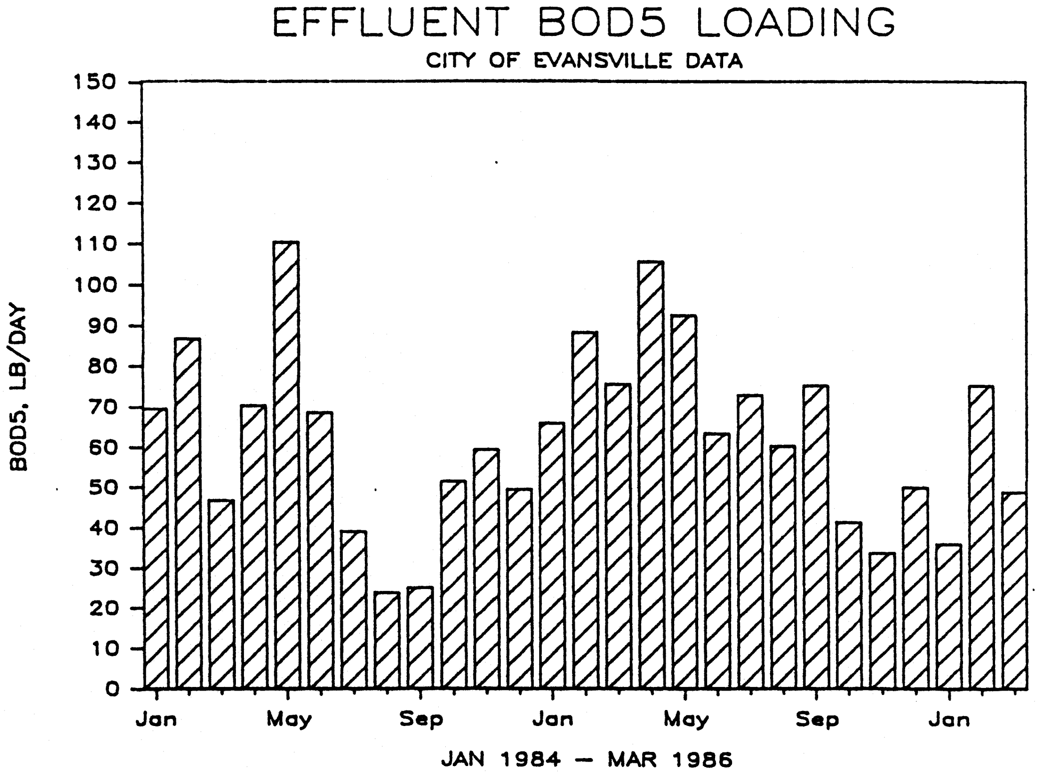


FIGURE 4.4

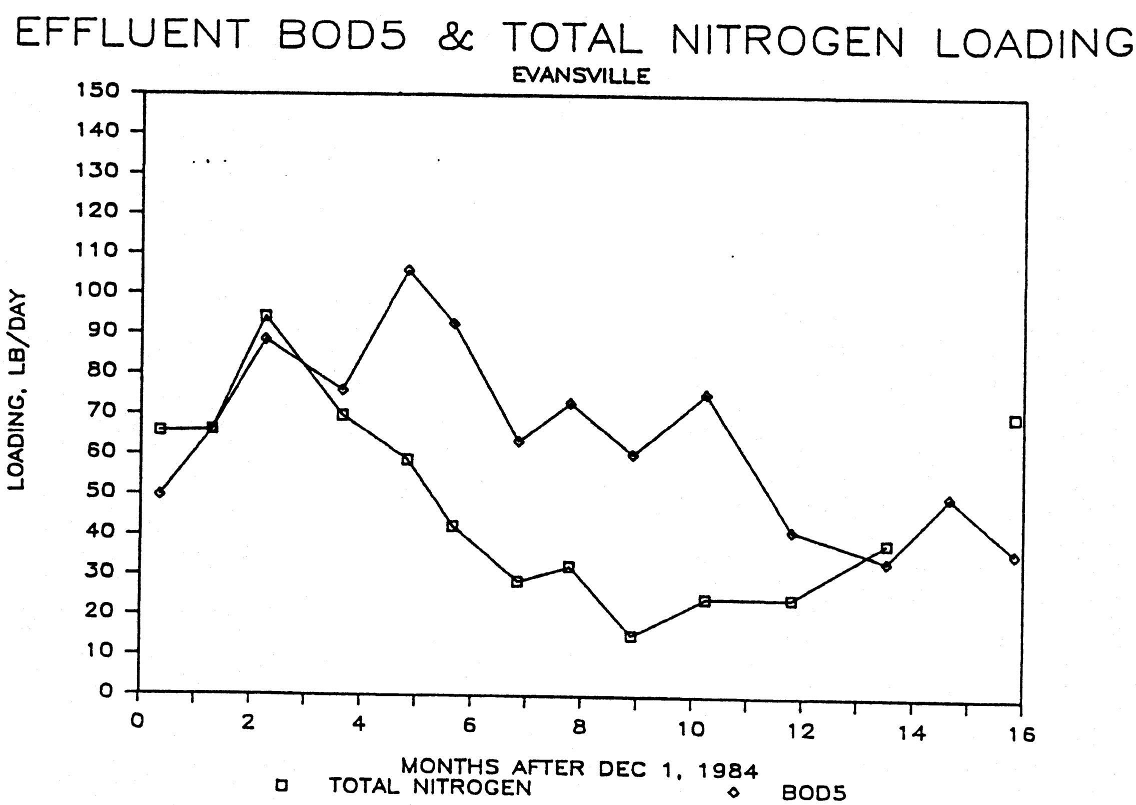


FIGURE 4.5

A decrease in the infiltration rates during the winter was observed. While this lower rate was not measured, it was evident because a solid ice cover formed on the cells by December and remained "floating" through February while effluent ponded below it and rose to depths of 4 ft. During both winters of 1984-85 and 1985-86 the NW cell was used most often as a result of frozen valves in the SW cell. Discharges to the SW cell occurred through an overflow culvert in the dike between the two cells.

It would be difficult to operate the seepage cells on a regular rest/load cycle throughout the winter. It is important to maintain a floating ice/snow cover that does not freeze solid to the soil surface or the system will fail hydraulically.

4.4 Treatment Within the Unsaturated Zone

The first area of interest in determining treatment levels occurring within the unsaturated soil profile is the flow time (or residence time) of the applied water as it moves through the unsaturated zone. The concept of unsaturated flow and governing equations are discussed in Section 2.3.

A first approximation of travel time in the unsaturated zone was made by assuming that the rate of infiltration at the surface governs the flow rate through the unsaturated soil profile. Applying Darcy's Law, with measured infiltration rates ranging from 4 to 9 ft/day, assuming a unit gradient for vertical flow (uniform moisture profile across the entire depth), and a volumetric water content of 15% (average of measured values that was believed to represent "drained" conditions) yields flow times ranging from 6 to 13.5 hours. This was calculated for an unsaturated depth of 15 ft. The major limiting assumption in this calculation is that the infiltration rate at the surface governs the flow rate over the entire depth. The assumption of a uniform gradient is realistic for an initially wet sand. However, due to variable loading conditions at the site, the soil moisture content distribution within the 15 ft. of unsaturated soil was not always uniform. The unit gradient assumption was used only for a gross estimation of flow time.

To better characterize flow conditions in the unsaturated zone, an alternate method was used. This involved measuring tensions as a function of time. These tensions were converted to a change in moisture volume over time. The unsaturated hydraulic conductivity, $K(\theta)$, was calculated from a relation where the inflow to a volume of soil less the outflow was equal to the change in volume over time. This relation is presented in Appendix C.3 with calculations for $K(\theta)$. $K(\theta)$ is included in the outflow term which is governed by Darcy's Law. The data required to make this calculation are summarized below.

Soil moisture tensions measured within the seepage cells at depths of 1, 2, 3.5 and 5 ft. ranged from 0 to 12 centibars (0 to 4.08 ft. of H_2O). Soil moisture content determined gravimetrically for 21 samples ranged from 5.4% to 10.1% on a dry weight basis. For 8 of these samples, soil moisture tensions were measured in the field for correlation to the moisture contents determined in the lab.

These moisture contents were converted to volumetric water content (volume of water/total volume of soil) by multiplying by the bulk density of the soil over the bulk density of water (1 g/cm^3). The average bulk density of 8 undisturbed soil samples taken from the SW cell was 1.71 g/cm^3. This resulted in volumetric water contents ranging from 9.2% to 17.3%. The 8 undisturbed samples were also used to determine an average porosity of 0.35 assuming a specific gravity of 2.65%. Methods for determination of bulk density and porosity were based on procedures listed in the 1985 ASTM Standards Manual for Soils, referred to in Section 3.3 of this report (Methods and Materials). Results of the soil testing are listed in Appendix A.3.

On July 19, 1985 tension measurements were taken manually at 4 depths as a function of time as the NW cell was loaded. These measurements represented the movement of the wetting front as the effluent was applied. Converting these readings to water content (using the correlation noted above), the change in volume of water over time was known for a unit area, finite depth volume of soil and could be set equal to the infiltration rate into this volume minus the outflow. The outflow is governed by: 1) the tension gradient across the outflow depth determined by the nested tensiometer readings; and 2) the hydraulic conductivity K(θ) for a particular volumetric water content (θ). (Darcy's Law).

The infiltration rate was measured in the field by a staff gauge. The tension (or pressure) gradients were measured directly from the tensiometers at nested depths for any given time and corresponding θ.

K(θ) was then calculated directly at each time and moisture content (θ), at a selected depth of 2 ft. Details and assumptions used in these calculations are listed in Appendix C.3. Results of the calculations for K(θ) are summarized in Table 4.3.

Table 4.3
Estimated Unsaturated Hydraulic Conductivities

Tension (ft H2O)	Volumetric Water Content %	K(θ) ft/day
1.70	15.5	13
1.36	16.5	22
1.02	18.0	25
0.	35.0	100*

*K (saturated) is included for comparison

As expected the K(θ) curve is steep at low tensions (high moisture contents). These results cover a range of 0 to 5 centibars of tension which makes interpretation of the data difficult as the tensiometer gauge has a precision of (+/-)1 centibar. In addition, it was not possible to obtain a full range of K(θ) and tension measurements due to the uncontrolled conditions in the field. The values in Table 4.3 apply only to the particular conditions in the field which were flooding of initially wet sand. Extrapolation of these results to conditions where tension were above 5 centibars would not yield accurate results. Given this limitation, a review of measurements through the summer showed tensions at the 5 ft. depth were in the 8 to 10 centibar range. It was concluded the K(θ)s less than or around 13 ft/day (from Table 4.3) were typical below the 3.5 to 5 ft. under unit gradient at 13 ft/day would be 4.3 hours assuming a volumetric moisture content of 15%.

Since the data are limited and do not represent the full range of K(θ) values that might occur, resulting travel times based on these data should be viewed as gross estimates only. The K(θ) relationship also changes depending on whether the soils are being wetted or dried. This hysteresis effect was not considered in the above analysis. Based on the discussion above, it was concluded that less than a one day residence time in the soil profile was typical. A portion of the effluent dose (trailing portion) would be retained as residual moisture content in the unsaturated zone. However, given the limitations of the measurement techniques and assumptions made, further interpretation of lysimeter and well data was made using flow times between 4 and 24 hours.

Analytical results of soil water samples taken from the suction lysimeters are listed in Appendix A.2. The lysimeter data showed high spatial and temporal variations. The soils drained rapidly, therefore, it was difficult to obtain a sufficient sample volume over a 2 day vacuum time (one cell loading). The total vacuum time was increased later in the study (3 days in August, 5 days in October and November) as it became increasingly difficult to draw enough sample to perform the necessary analyses. The data from the background lysimeter (which was located in the unused southeast cell) showed nitrate concentrations of 9 to 20 mg/l as N and COD concentrations of 12 to 37 mg/l. This lysimeter was initially intended to represent soil pore water quality unaffected by the wastewater. Results from a previous study in outwash sands at a site located 25 miles away gave background concentrations near detection limits for all parameters (Doran 1985). Based on this result and type of soils at Evansville it was concluded that past use of the cell should not have affected the results in the "background" lysimeter. Consequently, to estimate background conditions, results of the previous study by Doran were used.

Average chloride concentrations in the soil pore water taken from the lysimeters in both cells were similar to the effluent concentrations with the exception of early samples taken from the 7.5 ft. deep lysimeter in the SW cell. This deeper lysimeter was reinstalled after the first try and may not have been developed well enough prior to sampling. Values from this lysimeter, L3, were not used in the following interpretation until chloride concentrations reached levels typical of the wastewater.

Soil pore water chloride concentrations averaged 230 +/- 30 mg/l as compared to effluent concentrations that averaged 240 +/- 20 mg/l. This average suggests that the soil pore water was representatives of percolating effluent. COD concentrations measured in the effluent, in the soil pore water in both the SW and NW cells, and in the water table well 5S are shown in Figure 4.6. The soil pore water concentrations shown are averages for all times samples at each depth. There was no correlation between concentrations and depth for the number of samples taken. A comparison between COD concentrations in the effluent and water table well 5S located between the two cells suggests that reductions in COD concentrations are around 75%. Based on the lysimeter data, the reduction primarily occurs within the first 2 to 3 ft. of soil.

The soil pore water nitrogen data is less easily interpreted. In general the total nitrogen concentrations in the pore water at the various depths beneath both cells were variable, ranging from 3 to 30 mg/l total nitrogen. This variation could not be correlated to depth. for this reason, concentrations in the shallow wells immediately below the system were used to determine the overall removal rates in the unsaturated zone. The removal rates are discussed i Section 4.6. the lysimeter data was used to interpret and support conclusions based on the solute concentrations within the shallow groundwater system. Figure 4.7 illustrates total nitrogen concentrations in the effluent, lysimeters and well 5S. Although there is significant variability in successive samples from lysimeters there appear to be a general decrease over time as was measured in the wastewater. Samples taken every other day for two weeks in June 1985 for the wastewater and well 5S showed less than a 2 mg/l variation of total nitrogen between sampling (See Table 4.5). This suggests that some of the variability in the soil pore water data was due to the lysimeters and method of sampling (longer sample collection time and/or effects from the silical pack. There is also significant variability for discrete soil pore water samples between cells. Concentrations at all points in the SW cell were generally higher than soil pore water samples taken from the NW cell. The cause of this variability is not known due to the limited number of samples.

A value of the lysimeter data was that it allowed a description of the forms and distribution of the nitrogen series over the sampled depth. Figures 4.8 through 4.10 illustrate the nitrogen series distribution vs depth for selected sampling dates. Effluent and well 5S concentrations for the dates indicated were included for comparisons. Previous calculations suggest that travel time in the saturated zone is less than one day. Thus a comparison of the values at each depth can be made with no adjustment due to travel times between lysimeters.

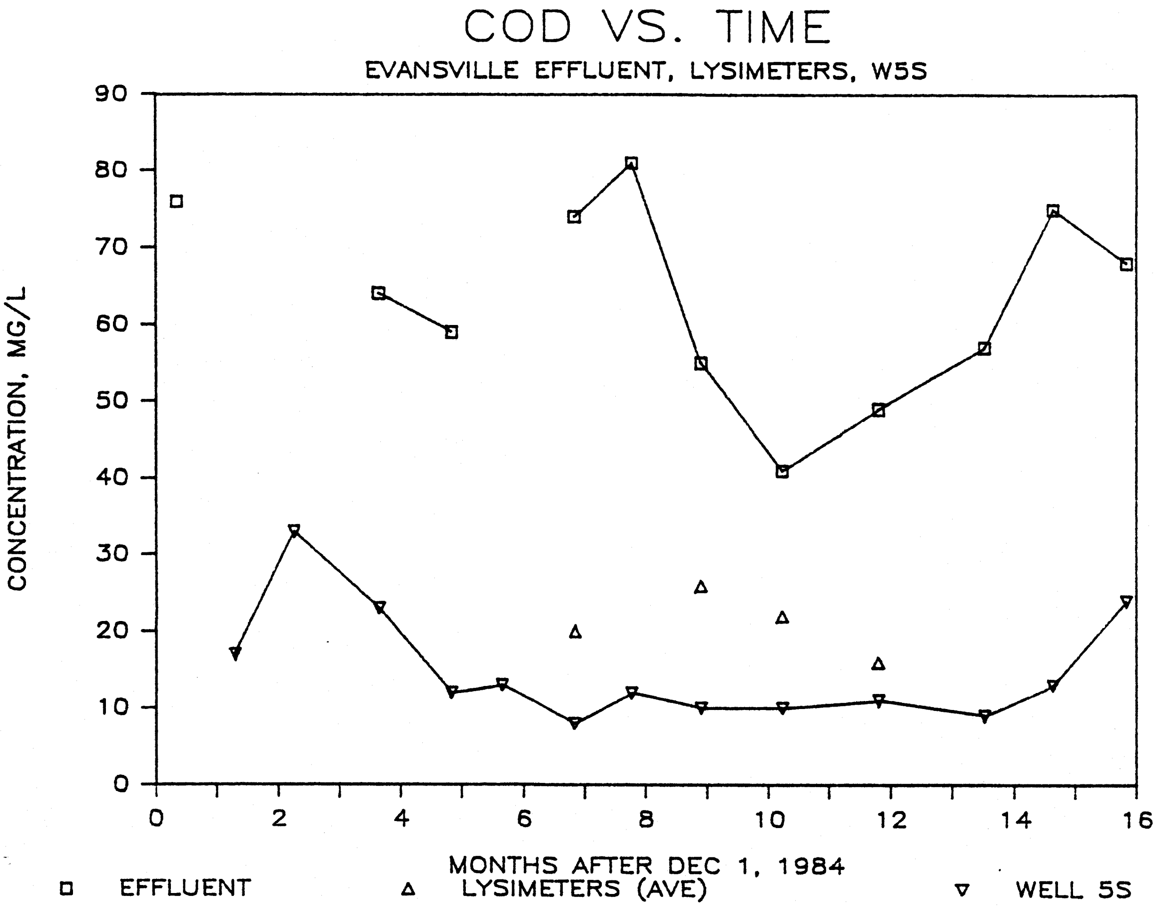
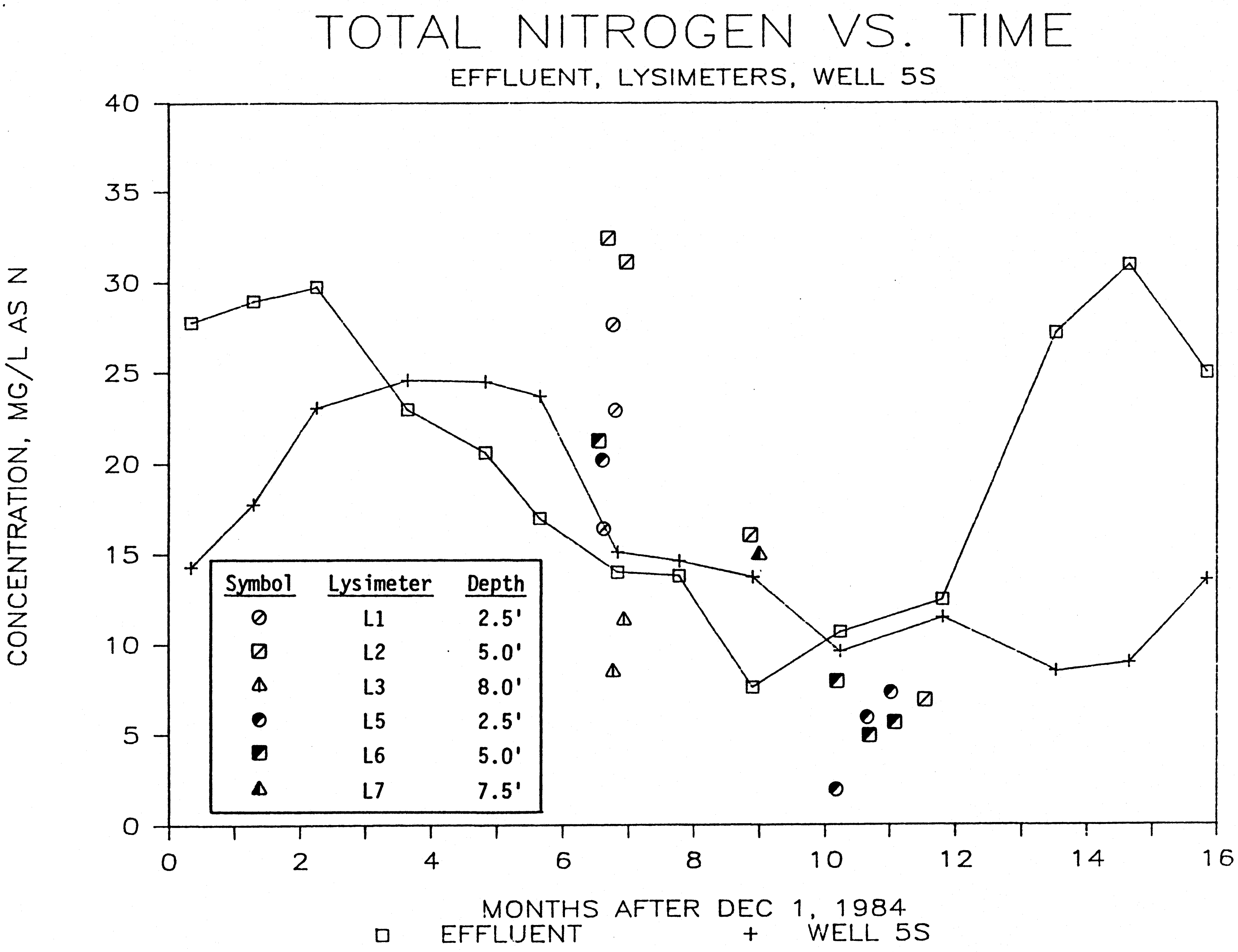
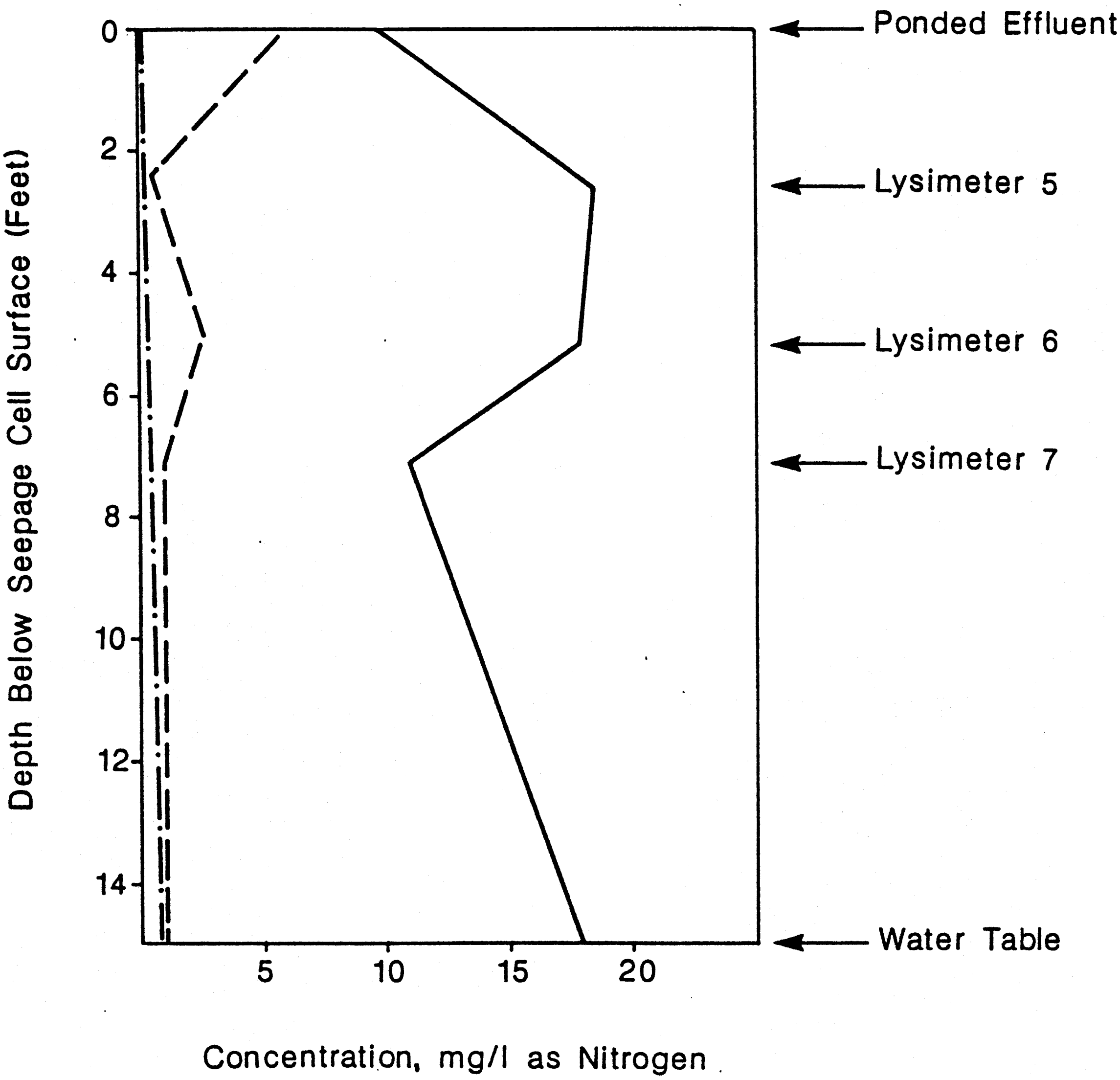

COD VS. TIME
EVANSVILLE EFFLUENT, LYSIMETERS, W5S
CONCENTRATION, MG/L
90
80
70
60
50
40
30
20
10
0
0
2
4
6
8
10
12
14
16
MONTHS AFTER DEC 1, 1984
EFFLUENT
LYSIMETERS (AVE)
WELL 5S

FIGURE 4.6



TOTAL NITROGEN VS. TIME
EFFLUENT, LYSIMETERS, WELL 5S
CONCENTRATION, MG/L AS N
0
5
10
15
20
25
30
35
40
0
2
4
6
8
10
12
14
16
MONTHS AFTER DEC 1, 1984
Symbol
Lysimeter
Depth
L1
2.5'
L2
5.0'
L3
8.0'
L5
2.5'
L6
5.0'
L7
7.5'
EFFLUENT
+ WELL 5S

FIGURE 4.7



NW Seepage Cell: Lysimeter vacuum set 6/19/85,
Sample collected 6/20/85.

Effluent sample collected 6/19/85.

Well 5S sample collected 6/20/85.

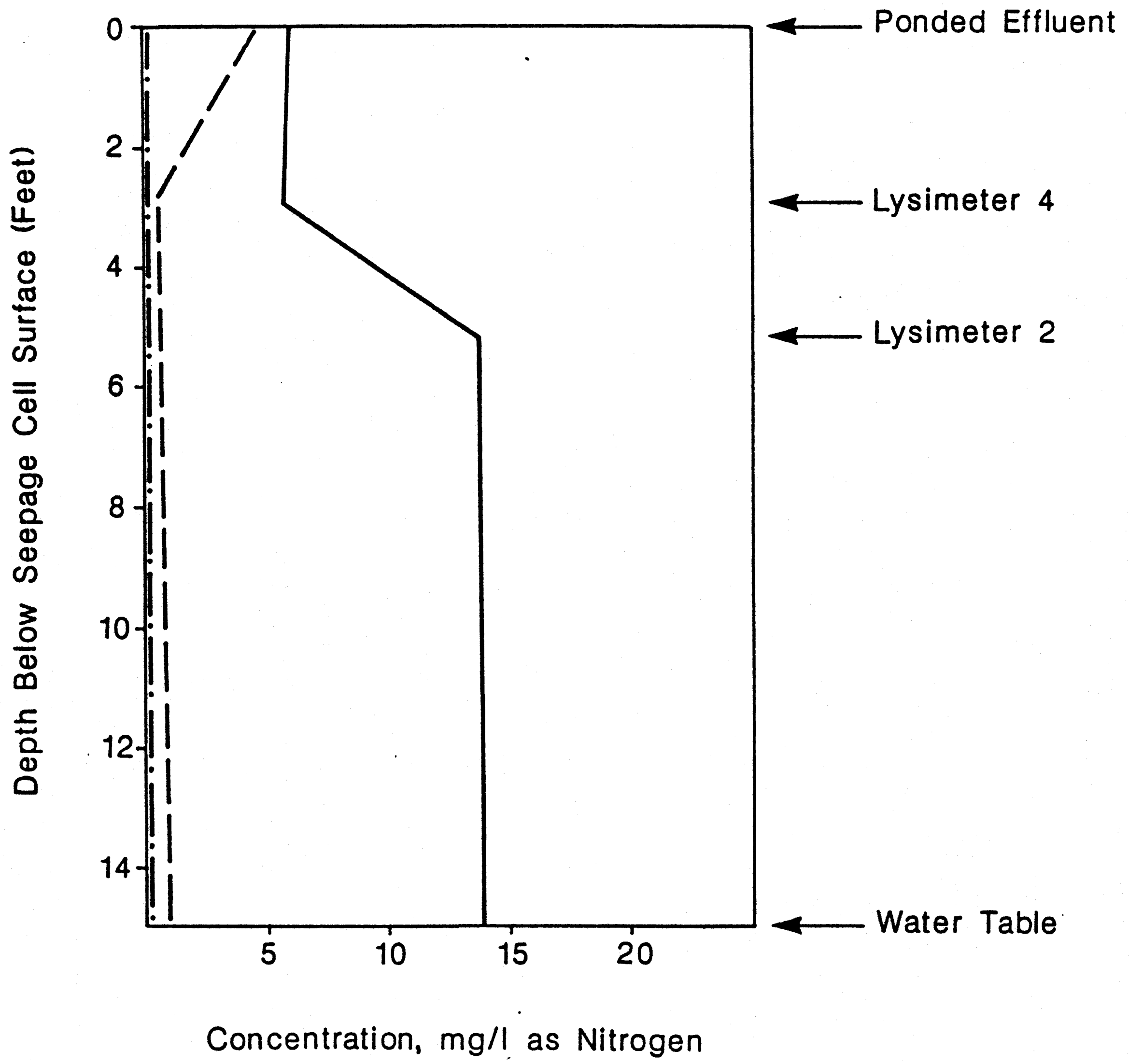
LEGEND

NO_3

TKN

NH_4

FIGURE 4.8
UNSATURATED ZONE
NITROGEN PROFILE
JUNE 19,1985



LEGEND

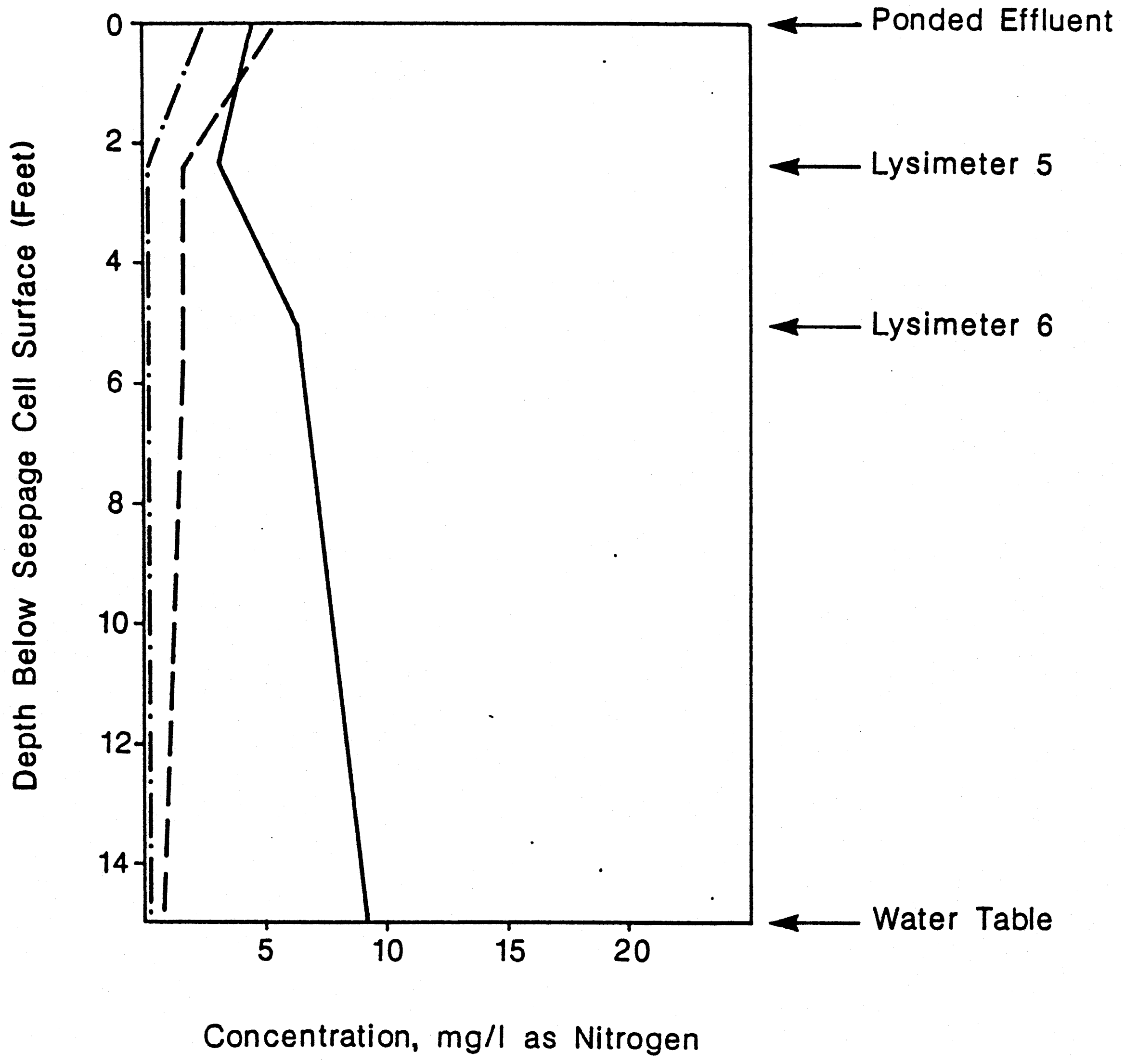
NO3 (solid line)
TKN (dashed line)
NH4 (dash-dot line)

SW Seepage Cell: Lysimeter vacuum set 8/27/85, Sample collected 8/29/85.

Effluent sample collected 8/28/85.

Well 5S sample collected 8/28/85.

FIGURE 4.9
UNSATURATED ZONE
NITROGEN PROFILE
AUGUST 27,1985



NW Seepage Cell: Lysimeter vacuum set 9/30/85, Sample collected 10/2/85.

Effluent sample collected 9/30/85.

Well 5S sample collected 10/7/85.

LEGEND

Line	Parameter
——————	NO_3
— — —	TKN
—·—·—	NH_4

FIGURE 4.10
UNSATURATED ZONE NITROGEN PROFILE SEPTEMBER 30,1985

During the period of time over which the lysimeters were sampled the effluent contained 3 to 7 mg/l organic nitrogen and nitrate concentrations ranged from 4 to 10 mg/l. Ammonia concentrations were below detection limit with the exception of the 9/30/85 sample which showed a concentration of 3.5 mg/l NH_2. The effluent nitrogen distribution vs time is shown in Figure 4.16 (presented in the next section). A general observation of the nitrogen data over the soil profiles was that the organic nitrogen was either converted to NO_3 or filtered out within the first 2 ft. of soil. This observation is evident for the sampling dates presented in Figures 4.8, 4.9, and 4.10.

Based on data from water table well 5S and the lysimeter results, there appeared to be very little nitrogen removal occurring within the unsaturated soil profile. The primary controlling factors appear to be that the gravelly, sandy soils offer little residence time and essentially no carbon source for the denitrification process to occur. Ponded conditions could not be achieved over long enough time periods for the system to become anaerobic which would promote denitrification. It was possible that a small amount of denitrification occurred in the micro-anaerobic zones (within individual pore spaces; however, the carbon to nitrogen ratio (C:N) may also be limiting at depths below 2 to 3 feet. Earlier results (Figure 4.6) showed that approximately 75% of the effluent constituents containing carbon sources were removed within 2.5 ft. while the more mobile NO_3 moved with the percolate towards the water table. To support these data, a comparison of input mass loadings of nitrogen was made to mass loadings reaching the water table. Results of these calculations are discussed at the end of this section.

4.5 Groundwater Flow System

It was necessary to describe the local groundwater flow system in order to evaluate the effects of the percolating effluent. Groundwater elevation contours based on borings taken prior to when the seepage cell system was installed are illustrated in Figure 4.11.

The general flow pattern was from the northwest to the southwest towards Allen Creek, which is 1000 feet downgradient of the site. The average horizontal gradient was .003 ft/ft. Seasonal water table fluctuations over the period of study were 2.18 ft, and 2.02 ft. as measured in the upgradient wells 101 (water table) and 9 (deep) respectively. Saturated hydraulic conductivities were determined from slug tests performed at 9 of the 13 monitoring wells. Due to the rapid recovery of the water levels (3 seconds or less) a pressure transducer and continuous chart recorder were used to obtain an instantaneous response curve. The conductivities were calculated using a method developed by Hvorslev for unconfined aquifers (Freeze and Cherry 1979). This calculation is described in Appendix C.1. The conductivities ranged from 46 ft/day to 194 ft/day (1.6×10^{-2} to 3.5×10^{-2} cm/sec). Hydraulic conductivities are listed with well specifications in Appendix C.1.

For an average K of 100 ft/day, a gradient of .003 ft/ft and a porosity of .35 the average linear groundwater velocity in a horizontal direction is .86 ft/day. Thus, it would take two months for the groundwater to flow 50 feet. assuming uniform horizontal flow.

The local groundwater flow was affected by the formation of a transient water table mound below this site. The basis for this conclusion is discussed below.

Continuous water levels that were recorded at well 5S showed a maximum rise in the water table of 0,.5 to 1.25 ft., typically within 4 hours after flooding of either the NW or SW cells began. Decay of the water table mound occurred during the day following loading and continued to decay over weekends when the cells were not loaded for two consecutive days. The levels after two days would often drop a few inches below the level prior to loading.

Groundwater contours based on measurements taken March 26, 1986 are shown in figure 4.12. Water tables slopes downgradient of the seepage cells were steeper than those shown under pre-site conditions (see Figure 4.11). It should be noted that elevations taken in March represent a "picture" of the groundwater flow system as affected by the past winter operation of the system. Since seepage rates were slower during the winter and ponded conditions resulted from December 1985 through March 1986, the height of the mounded surface may have changed less (in time) as compared to the summer when seepage rates were higher. Continuous water levels were monitored from June through september 1985 only, therefore, the effects of winter operation were not as clear. However, it was observed that horizontal gradients were steepest between wells 5S and 102 (.0065 ft/ft) and wells 10 and 8 (.005 ft/ft during the winter months.

Finally, well nests located between the cells allowed measurement of the vertical gradients below the site. Vertical downward gradients ranging from 0.0 to .006 ft/ft were measured at the well 5 nest and well 10 nest. A significant downward gradient existed when the cells were being loaded; there was a negligible gradient when the cell was rested.

The maximum height of the water table mound was estimated to be 2 ft. for normal loading conditions, using an analysis developed by Hantush (1967). The predicted rise and decay of a transient mound, using the same analysis, was in reasonable agreement with the observed response in well 5S. This analysis is described in Appendix C.2. A mound height of 2 ft. would result in a remaining unsaturated depth of 13 ft. This mound did not affect the hydraulic performance or treatment capability of the system.

There were upward vertical gradients measured between nested wells 5M and 5D which ranged from .002 to .003 ft/ft. This was above measurement error and is similar to the measured horizontal gradients. This upward gradient may result from regional hydrogeologic conditions which exist at greater distances than can be characterized by the data from this study.

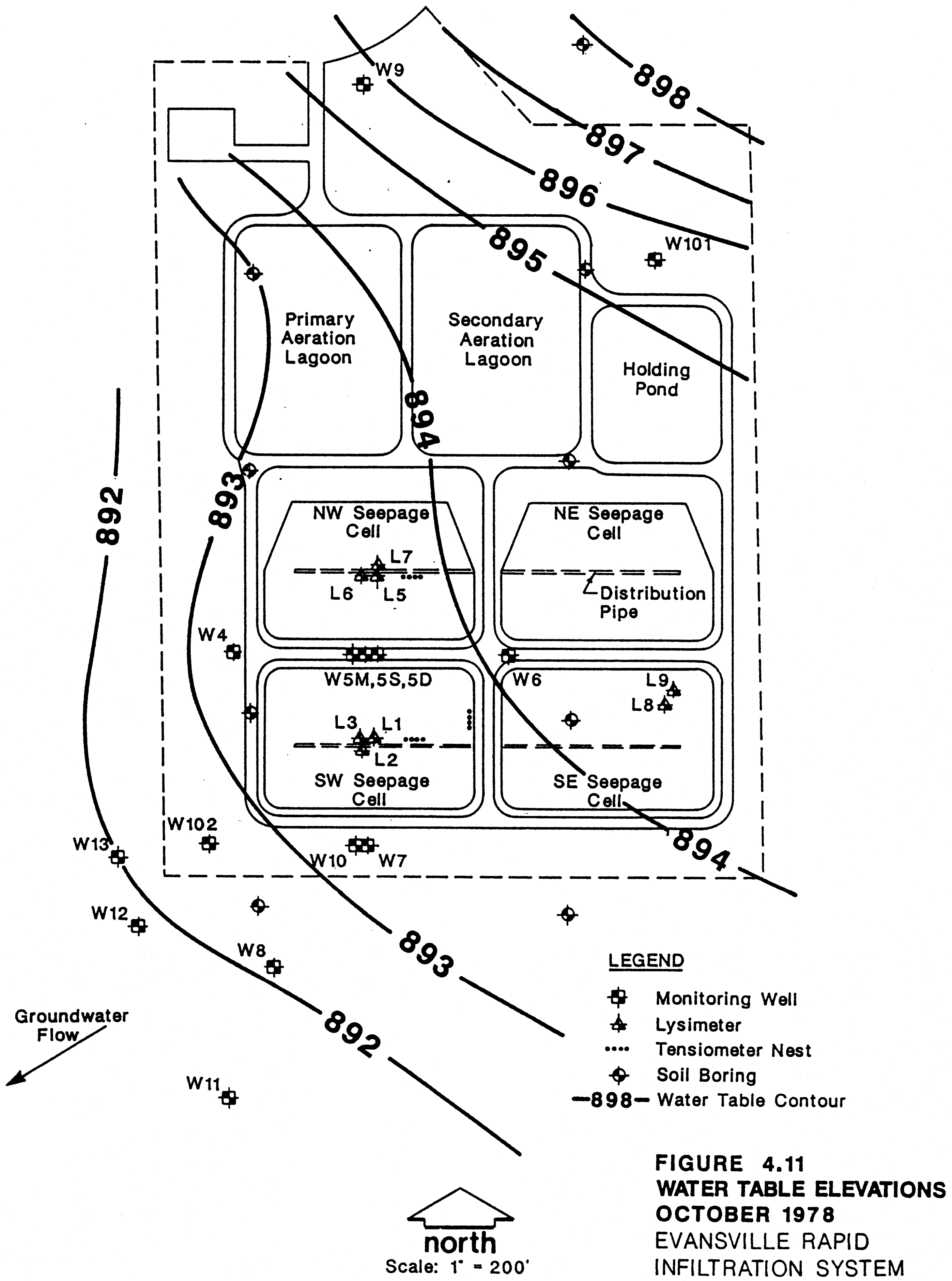


FIGURE 4.11
WATER TABLE ELEVATIONS
OCTOBER 1978
EVANSVILLE RAPID
INFILTRATION SYSTEM

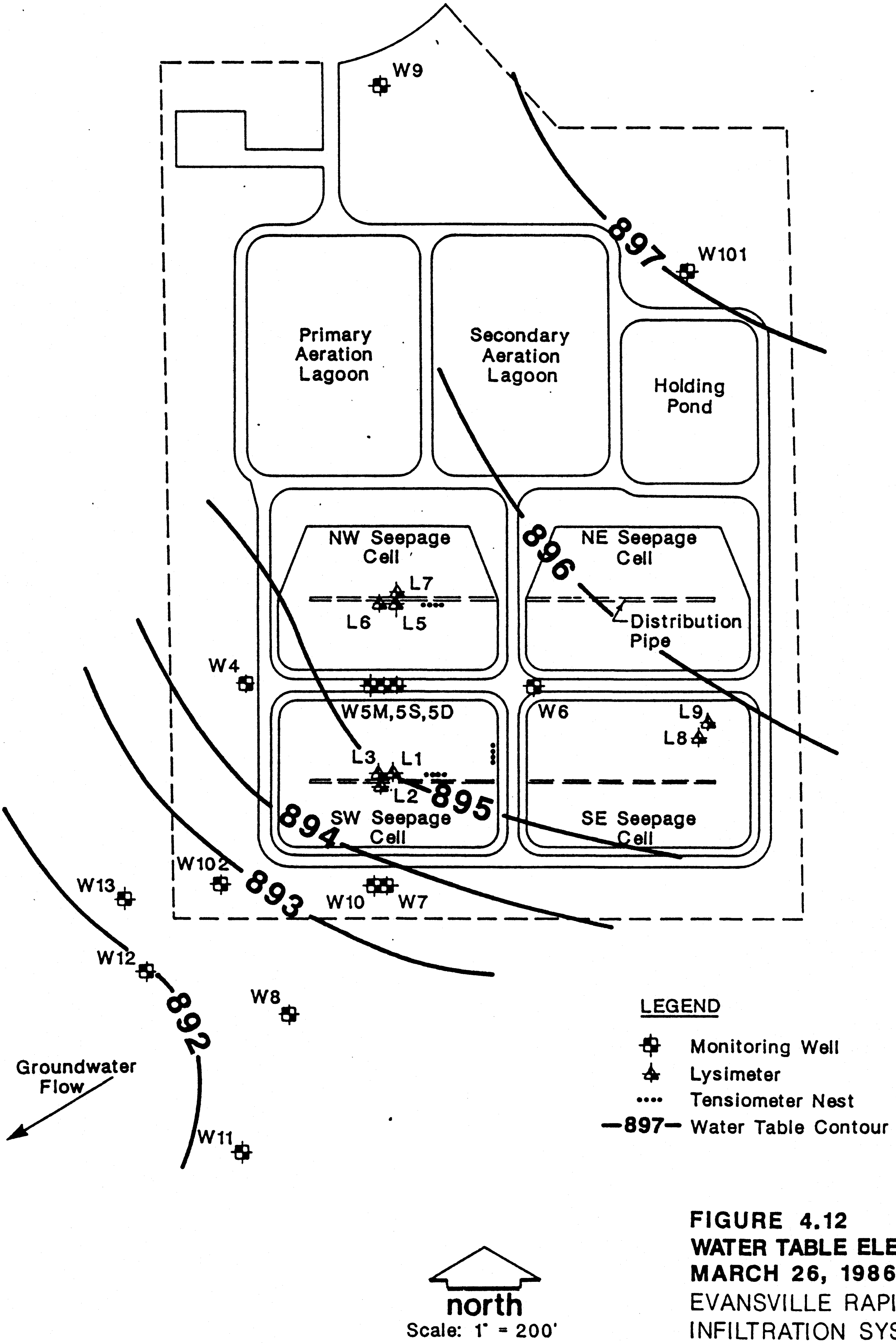


FIGURE 4.12
WATER TABLE ELEVATIONS
MARCH 26, 1986
EVANSVILLE RAPID
INFILTRATION SYSTEM

Table 4.4

GROUNDWATER CHEMICAL PARAMETERS - EVANSVILLE

Mean and Standard Deviations - Monthly Samples
December 1984 - March 1986

Well		Elev-ft	COD	Tot-N	TKN	NH_3-N	NO_3-N	Cl
9	mean:	895.53	<5	7.5	<0.2	<0.1	7.3	13.9
	std dev:	(0.58)	-	(0.3)	-	-	(0.3)	(1.1)
101	mean:	895.82	<5	11.9	<0.2	<0.1	11.7	40
	std dev:	(0.63)	-	(0.9)	-	-	(0.8)	(3)
6	mean:	894.16	-	8.7	0.6	0.3	8.1	135
	std dev:	(0.66)	-	(2.9)	(0.4)	(0.4)	(2.9)	(94)
5S	mean:	893.78	15	16.0	5.2	4.7	10.7	230
	std dev:	(0.80)	(7)	(5.6)	(7.1)	(6.8)	(5.3)	(20)
5M	mean:	893.75	15	15.2	3.9	3.5	11.2	240
	std dev:	(0.7)	(9)	(9.3)	(6.4)	(6.4)	(6.7)	(20)
5D	mean:	893.77	<5	4.9	0.3	0.2	4.9	13
	std dev:	(0.7)	-	(0.5)	(0.2)	(0.2)	(0.5)	(8.2)
4	mean:	893.18	-	12.5	1.7	1.4	10.8	170
	std dev:	(0.66)	-	(3.5)	(2.4)	(2.3)	4.1	(50)
102	mean:	892.17	11	14.8	1.6	1.3	13.2	240
	std dev:	(0.53)	(2.1)	(6.7)	(1.6)	(1.6)	(5.7)	(30)
10	mean:	892.75	9	16.5	1.8	2.5	14.9	220
	std dev:	(0.57)	(1)	(8.2)	(2.3)	(3.7)	(6.7)	40
7	mean:	892.91	12	14.4	5.5	4.9	8.9	160
	std dev:	(0.62)	(5)	(7.7)	(4.7)	(4.7)	(3.7)	50
8	mean:	891.67	10	13.7	0.8	0.5	12.9	210
	std dev:	(0.50)	(2)	(4.4)	(0.5)	(0.5)	(4.4)	20
11	mean:	891.67	-	7.9	<0.2	<0.1	7.9	11.6
	std dev:	(0.50)	-	(0.4)	-	-	(0.4)	(4.0)
12	mean:	891.47	12	8.5	0.7	0.4	7.8	215
	std dev:	(0.42)	(2)	(2.7)	(0.5)	(0.6)	(2.3)	10
13	mean:	891.89	8	10.4	1.2	1.0	9.2	220
	std dev:	(0.40)	(1)	(2.4)	(0.6)	(.6)	(1.9)	10

4.6 Groundwater Solute Distribution

Values of the chemical parameters that were analyzed at each well for all sampling dates are listed in Appendix A.1. Mean values for selected parameters are summarized in Table 4.4.

To illustrate the spatial (horizontal) distribution of contaminants within the flow systems, iso-concentration contours for chlorides are shown in Figure 4.13. These contours were drawn based on results from samples taken early in the study on January 9, 1985. In the interpretation of the groundwater chemical data, chlorides were used as a conservative tracer to detect the general level of impact at the sampling points. Chloride concentrations less than 40 mg/l represent background conditions (vs 240 mg/l in the wastewater). Wells 101, 9, 5D and 11 consistently showed concentrations around background levels. From later samples taken after June 1985 a shift in the center of mass of the plume was observed. This is shown in Figure 4.14 for data taken on January 16, 1986. This shift was detected by a sharp drop in chloride concentrations in well 6.

The decrease in chlorides was attributed to the change in operating conditions at the site. Prior to this study all four seepage cells were used. After October 1984, only the NW and SW cells were used. Thus, background conditions influenced the results in well 6 in January 1986 as it was upgradient of the area being recharged by percolating effluent. The decrease in chloride concentrations to background levels in well 6 was detected approximately 6 months after the NE and SE cells were no longer used. This shift in the plume center of mass correlates approximately with the estimated groundwater velocity of 1 to 2 ft/day.

At a distance of 250 ft. downgradient of the site, using chlorides as a tracer parameter, the amount of dilution over wastewater concentrations was about 10%. This estimate was based on concentrations in wells 8, 12 and 13 which are all located approximately 250 ft. downgradient. This result was checked by applying a simple mass balance completely mixed model. A volume defined by the width of the seepage cell system perpendicular to the groundwater flow and a length of the system parallel to the groundwater flow plus 250 ft. downgradient was used. The height of the mixing cell was set at 35 ft. to correspond to the vertical extent of the plume as it was determined to lie between 20 and 50 ft. below the water table (see chloride results for wells 5M and 5D in Table 4.4). Chloride concentrations in this "mixed cell model" were predicted using mass inputs from wastewater inflow, groundwater inflow, and an assumed amount of recharge. This relation is presented in Appendix C.4.

Results were calculated on a yearly basis. Since start up of the system in 1983 the concentrations in the mixed zone leveled off in 1985 to approximately 210 mg/l, using an input concentration of 245 mg/l chloride in the effluent, 40 mg/l chloride in the background water and 1 mg/l in the recharge water (precipitation). The concentration in 1986 was predicted to be 213 mg/l. These results are in good agreement with the observed concentrations. Details of these calculations are listed in appendix C.4.

Chloride concentrations in water table well 11 located 450 ft. downgradient of the seepage cells averaged 13 mg/l for 6 samplings. Based on this low concentration, it was concluded that the shallow groundwater at this location was not impacted by the effluent plume. Three possible reasons are offered to explain the absence of the plume at this location.

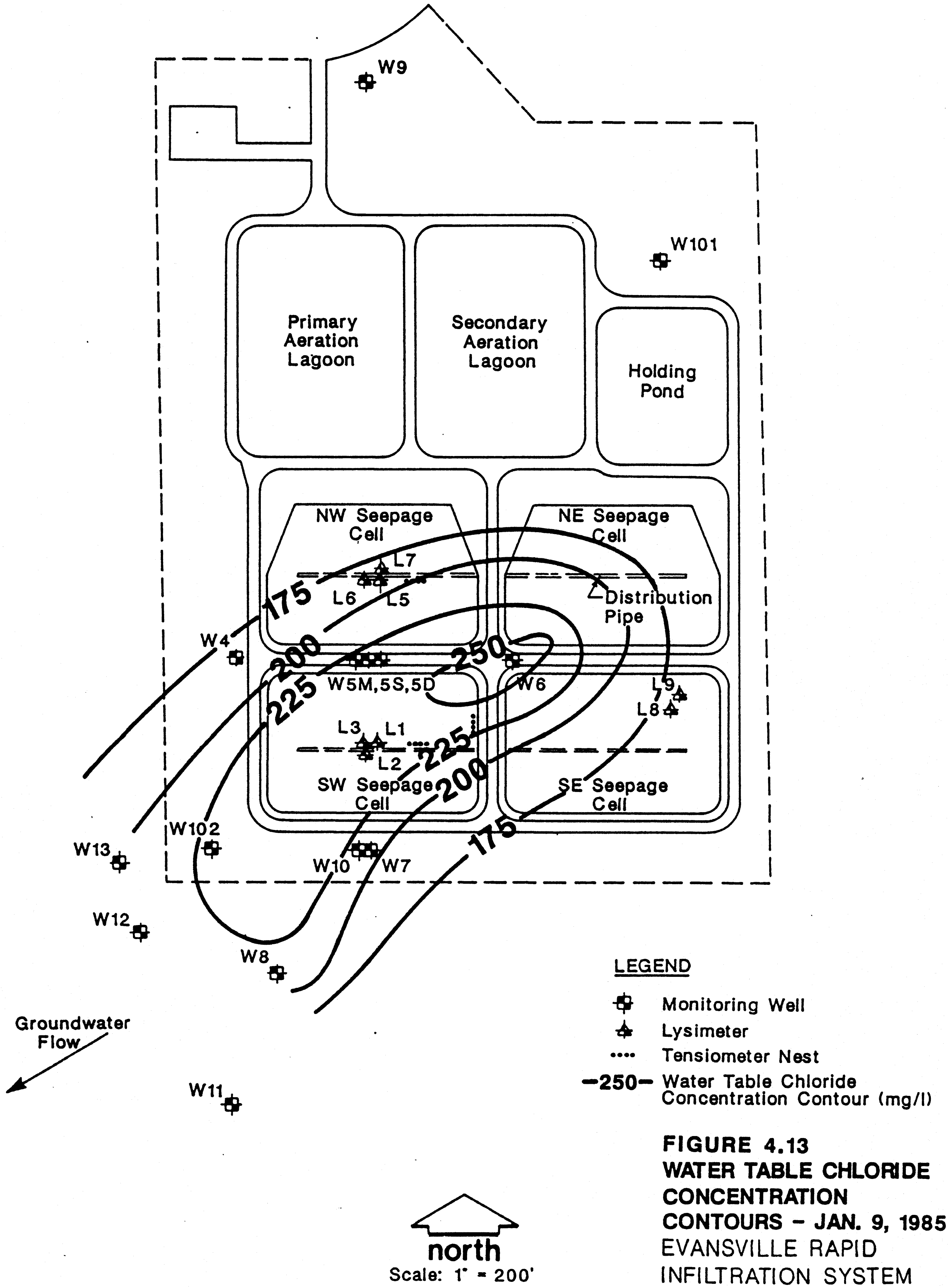
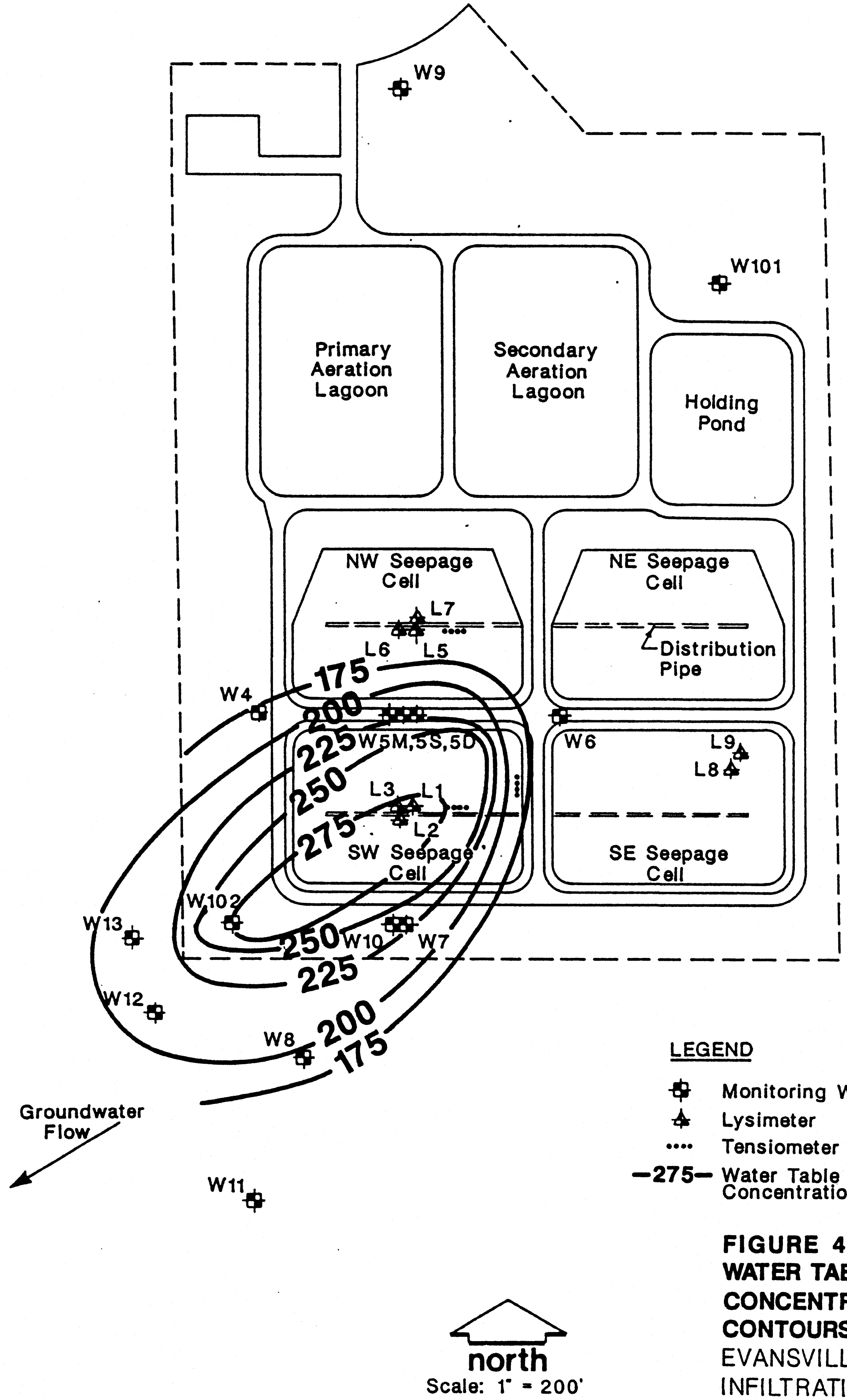


FIGURE 4.13
WATER TABLE CHLORIDE CONCENTRATION CONTOURS – JAN. 9, 1985
EVANSVILLE RAPID INFILTRATION SYSTEM



LEGEND

- Monitoring Well
- Lysimeter
- •••• Tensiometer Nest
- —275— Water Table Chloride Concentration Contour (mg/l)

FIGURE 4.14
WATER TABLE CHLORIDE CONCENTRATION CONTOURS - JAN. 16, 1986
EVANSVILLE RAPID INFILTRATION SYSTEM

First, assuming that advection was the dominant mechanism in the transport of the tracer parameters in the groundwater at this site, the effluent plume may be taking a path of less resistance (e.g. flowing within the outwash deposit which south of cells appears to extend further to the west). Second, the effluent plume may not have reached well 11, since the system was in operation for only 3.25 years. A travel time of 4.3 years was estimated at this location using a K of 50 ft/day (determined at well 8) a gradient of .002 ft/ft (maximum measured) and a porosity of .35. A third possibility is that the outwash material extends below the silty material and contaminants are moving within that deposit. The vertical extent of the silt layer is unknown below 10 ft. from the ground surface. Additional well drilling and field work would be necessary to describe the flow system accurately and resulting implications on contaminant transport in this area.

The vertical extent of impacted groundwater immediately below the site can be inferred from results of samples taken from the well nest located between the cells, including W5S, W5M, and W5D.

Well 5D screened 48-51 ft. below the water table showed no impact from the effluent based on a mean chloride concentration of 12.4 (+/-8 mg/l). Well 5M screened 23-26 ft. below the water table showed a mean chloride concentration of 240 mg/l (+/-19 mg.l). It was concluded that the vertical extent of the plume in the area immediately below the NW and SW seepage cells was between 30 to 45 ft. below the water table.

COD concentrations were highest in wells 5S and 5M below cells (6-33 mg/l compared to wastewater concentrations ranging from 41 to 81 mg/l COD). The highest concentrations occurred during the month of March in both wells for both 1985 and 1986. This coincides with high effluent concentrations and cold soil temperatures (inferred from groundwater temperature data from well 5S). Monthly analytical results for the wastewater and well 5S are shown in Figure 4.6.

An average nitrate concentration of 11.7 (+/-0.8 mg/l) as nitrogen was measured in water table well 101, located 250 ft. upgradient from the seepage cells. Effluent constituents were measured at background levels at this location. In addition, samples taken prior to construction of the seepage cells in 1983 showed similar nitrate concentrations in well 101. Nitrate concentrations averaged (7.3 +/-0.3 mg/l) in the deeper upgradient well 9. This suggests that past and/or upgradient agricultural practices (application of fertilizer or manure) have contributed to the increased nitrogen levels in the upper portion of the groundwater system. Such water bearing unconsolidated aquifers, comprised of outwash materials, have been identified as susceptible to contamination from ground surface activities (Zaporozec 1985). Background concentrations of organic or NH_3-N were less than detection limits.

Below and downgradient of the seepage cells the total nitrogen concentrations reflected those of the effluent. Temporal fluctuations of total nitrogen in well 102 (125 ft. downgradient), well 5S and the effluent are compared to background levels in Figure 4.15. From these data it was evident that the total nitrogen input undergoes very little removal in the unsaturated zone, and that over the 125 ft. distance to well 102 effects from reduction or from dispersion (dilution) were not detected. The lag of total nitrogen concentrations in well 5S with respect to the effluent was unexpected based on the estimated travel times of approximately one day. In part this may be due to the retention of some effluent in the unsaturated zone as "residual" moisture which would cause additional mixing and dispersion prior to reaching the water table. In addition, the conversion of organic nitrogen (retained in the upper portion of the unsaturated zone due to filtering) to soluble forms may be delayed due to cold temperatures. This would explain the higher nitrogen concentrations in well 5S in warmer months as shown in Figure 4.15. A further comparison of nitrogen peaks showed a 3 to 4 month lag between total N in the effluent and total N measured in Well 102. The flow time between the edge of the SW cell to well 102 was approximately 3.5 months based on an average K of 100 ft/day (determined for wells in that area), a gradient of .004 ft/ft and porosity of .35. This result supports the measured concentrations at well 102, assuming that the travel time within the unsaturated zone was negligible compared to travel time within the groundwater and that dilution and reduction were minimum.

Another observation of the data presented in Figure 4.15 is that during February and March of both 1985 and 1986 total N concentrations in well 102 fell below the background level due to the input of large volumes of low nitrogen content recharge water (effluent percolate) during the summer months. The delay was correlated to expected travel times in the groundwater.

Results from well 5S support these conclusions because tracer parameter concentrations (chloride and TDS) remained at levels near those in the wastewater during the period of study. Total nitrogen in the effluent and well 5S were presented with the lysimeter data in Figure 4.7.

Results of total N measurements in wells 8, 12, and 13 located 250 ft. downgradient of the seepage cells also reflected these temporal fluctuations. Levels were not as high at these wells suggesting some amount of dispersion, however estimated travel times again correlated with the arrival of higher and lower nitrogen concentrations.

The distribution of forms of nitrogen within the groundwater including NO_3-N, NH_3-N and organic N were evaluated to detect whether transformations were occurring both in the unsaturated and saturated zones. Figures 4.16 through 4.19 illustrate the speciation of nitrogen in the effluent, well 5S, well 102 and well B. These graphs in sequence represent a picture of the spatial (horizontal) and temporal changes in the distribution of the nitrogen forms at the 4 sampling points. These distributions are discussed below.

Ammonia levels were high in the effluent during the winter months while NO_3 was the dominant species during warmer summer months. This fluctuation was discussed earlier. Although similar fluctuations were evident from samples taken from well 5S, the peak total N concentrations in well 5S were generally lower than those in the wastewater for any given sampling date. A mass balance of input vs. output nitrogen over the unsaturated soil profile was performed to determine the overall difference over a 12 month period. By performing a nitrogen budget balance over 1 year, seasonal variations and redistribution of nitrogen due to temporary adsorption in the soil profile or mixing could be neglected.

Assumptions that were used to determine the mass differences included: 1) all of the effluent discharge to the seepage cells reached the water table (effects of precipitation or evaporation were ignored); 2) levels measured in well 5S were representative of the effluent percolating over the entire cell area and no mixing with the background water occurred; and 3) levels measured for the effluent and well 5S each month (one grab sample) were representative of the entire month.

The amount of nitrogen unaccounted for at the water table would be the amount removed within the unsaturated soil profile and attributed to denitrification. Loading rates (on a mass basis) were calculated using the analytical result for nitrogen for a particular month and the average volumetric flow rate in that month. These were summed over one year beginning December 1984 through November 1985. Tabular results of this estimate are presented in Appendix C.5.

In making the assumptions and calculations described above, the nitrogen loading rate to the seepage cells was estimated to be 16,990 lbs/year. The amount of nitrogen reaching the water table was estimated to be 15,900 lbs/year. This is a 6% reduction in the total nitrogen applied. Values in well 5S for some months exceeded those in the effluent due to reasons discussed previously (regarding Figure 4.15) and due to error inherent in using monthly "grab" samples for concentration and average flow rates. The error in making this mass balance calculation could

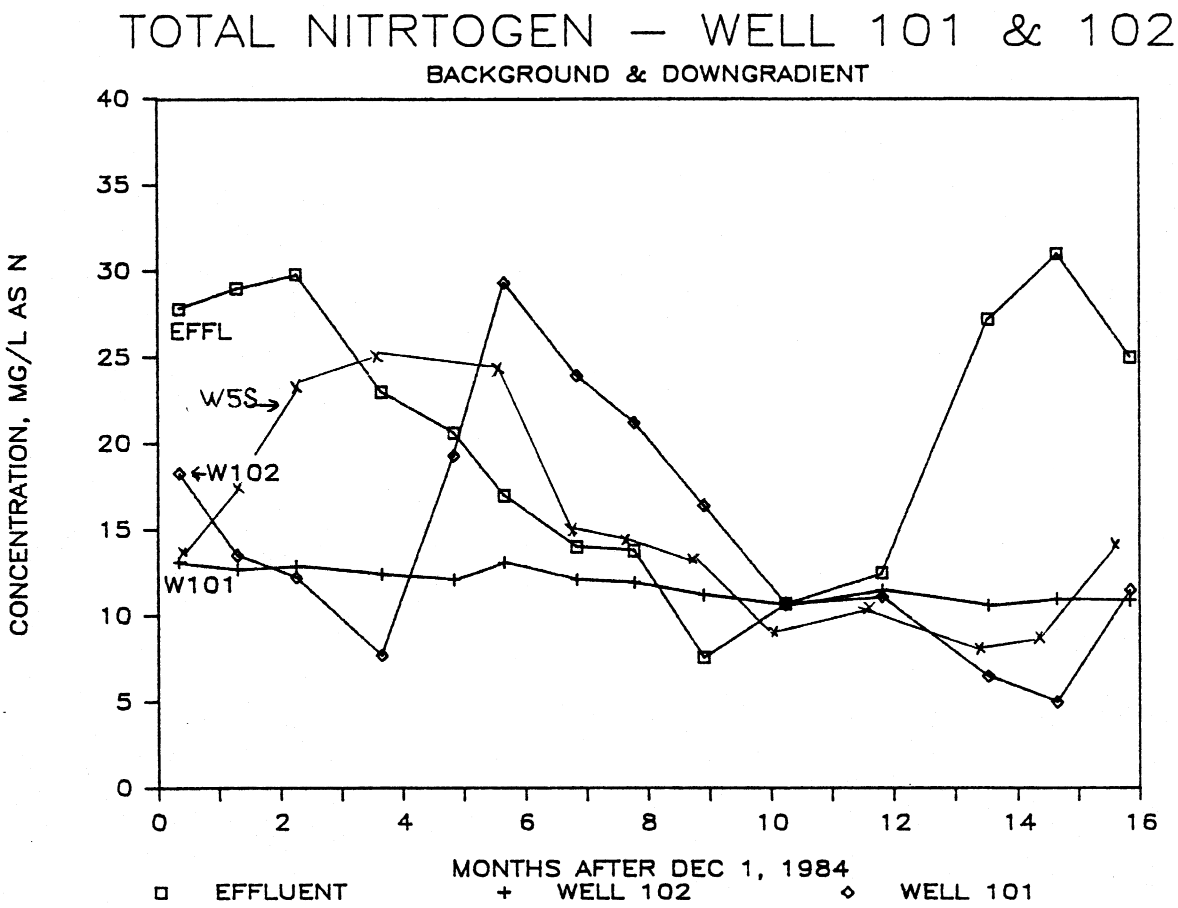

TOTAL NITRTOGEN — WELL 101 & 102
BACKGROUND & DOWNGRADIENT
CONCENTRATION, MG/L AS N
MONTHS AFTER DEC 1, 1984
EFFL
W5S
W102
W101
EFFLUENT
WELL 102
WELL 101
0
5
10
15
20
25
30
35
40
0
2
4
6
8
10
12
14
16

FIGURE 4.15

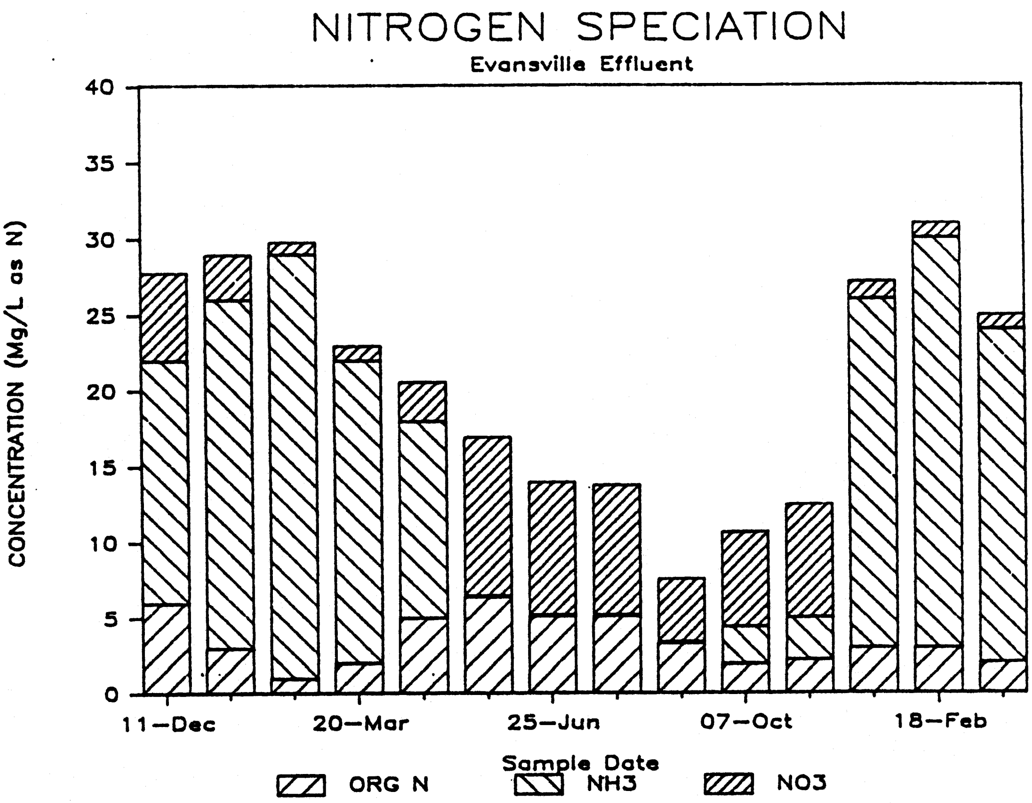


FIGURE 4.16

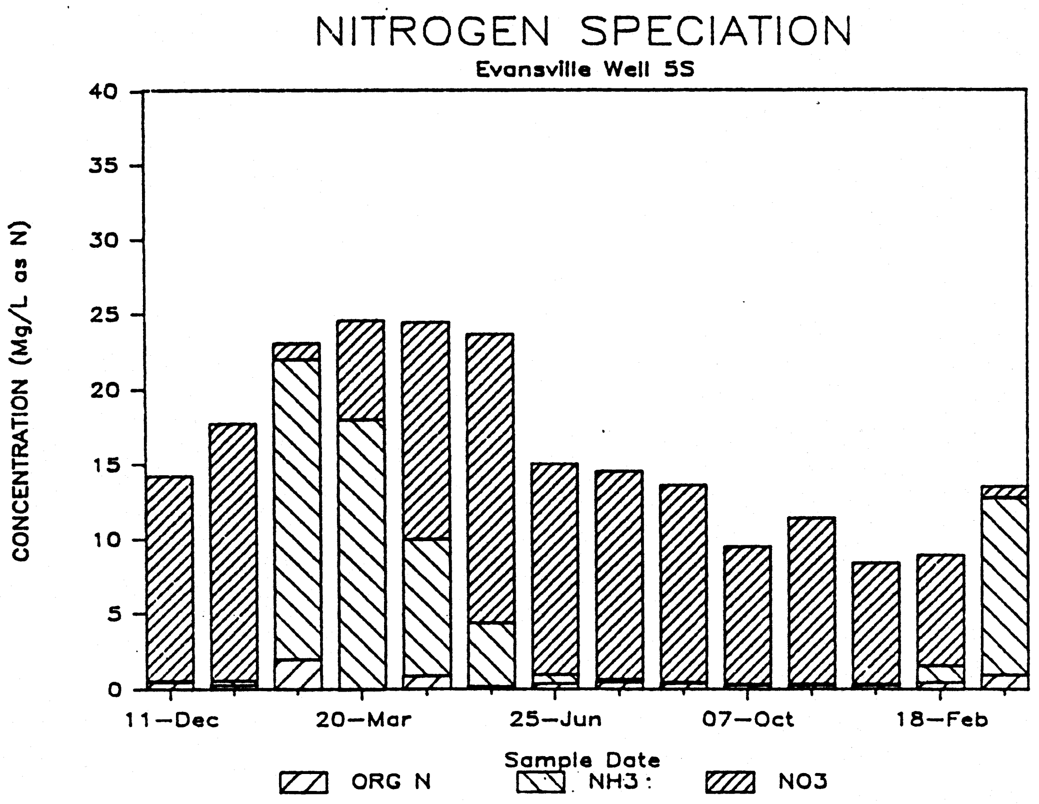


FIGURE 4.17

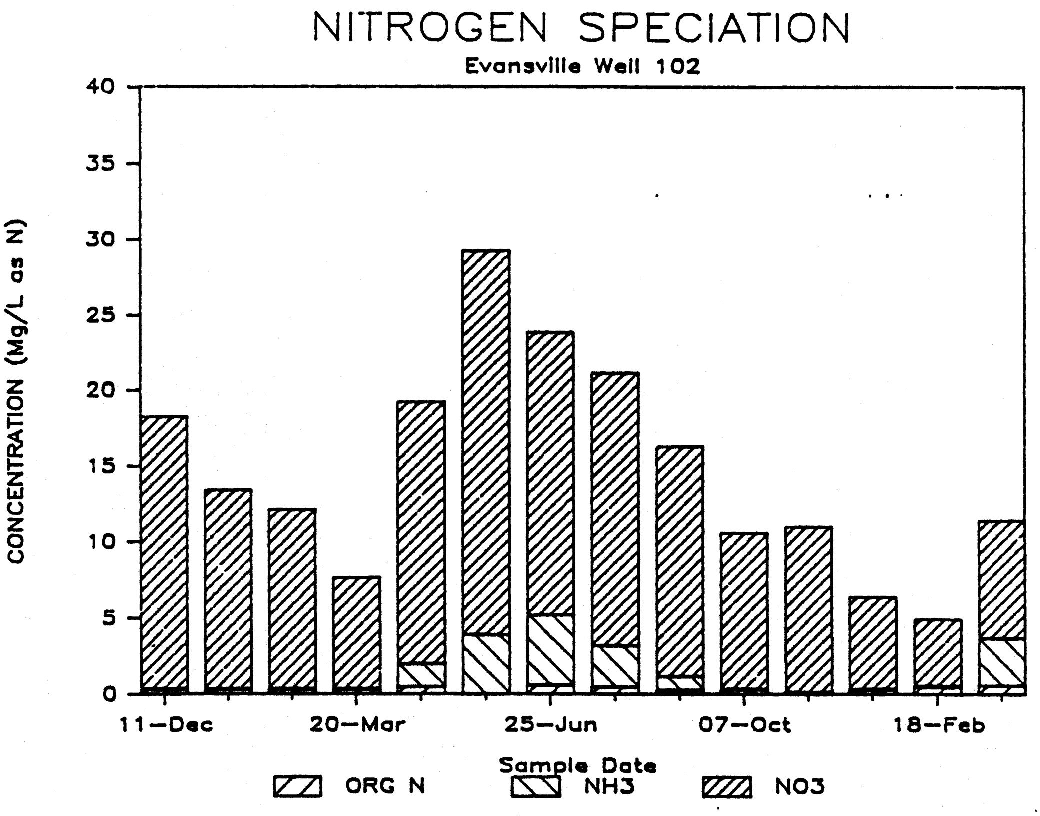


FIGURE 4.18

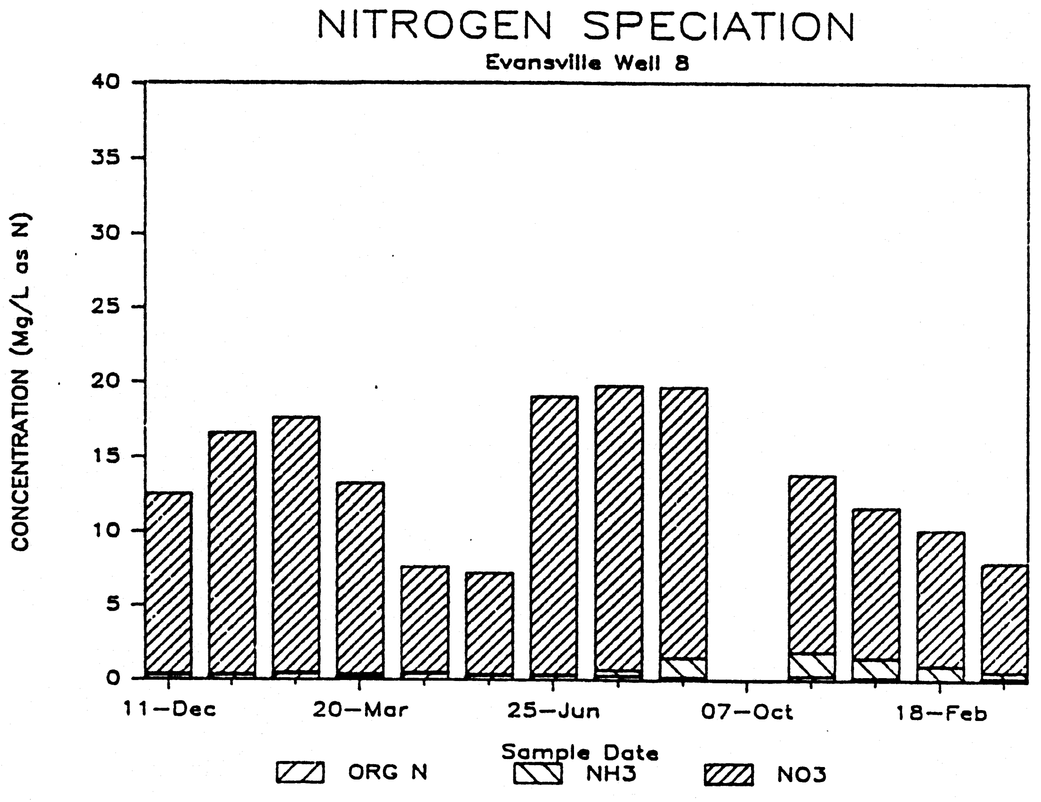


FIGURE 4.19

easily be in excess of +/-10%. It is therefore concluded that no measurable decrease in total N within the soil profile was detected. Additional work with lysimeters would aid in making a more accurate estimate of N removal rates.

The distribution of forms of nitrogen in the groundwater depicted by results from wells 5S, 102, and 8 reflected effluent concentrations to varying degrees depending on the spatial distance to the sampling point.

NH_3 levels in well 5S increase approximately 2 months after levels in the effluent increase. This increase is shown by Figures 4.16 and 4.17. This delay was attributed to nitrification in the unsaturated zone. Heat held in the soil warmed the incoming colder wastewater such that nitrifying bacterial continued to function in converting NH_3 to NO_3. The temperature in W5S was 72• F on 10/7/85. By February soil temperatures cooled to levels where nitrification slowed and eventually ceased (less than 40• F). The result was a breakthrough of NH_3 laden percolate which was then detected at the water table. Temperatures measured in well 5S were 72• F on 10/7/85, 60• F on 11/24/85, 40• F on 1/16/85 and decreased to 35• F on 2/18/86. Temperatures of the effluent were between 33• and 35• F from December 1985 through February 1986. Temperature data for the background water table well 101, the effluent, well 5S, and well 8 located 250 ft. downgradient are presented in figure 4.20. These data correlated with the breakthrough of NH_3 laden percolate in February. These data also suggest that temperatures in the soil column dropped to near freezing temperatures as well and therefore nitrogen reductions by way of denitrification is unlikely during winter months.

Figures 4.18 and 4.19 show the distribution of nitrogen in samples from wells 102 and 8 located 125 ft. and 250 ft. downgradient respectively. A delay of the NH_3 peak and reduction of NH_3 levels was seen at both locations.

The lag time was in part attributed to the previously calculated travel times of 3-4 months to well 102 and 6-8 months to well 8. These travel times correspond to the arrival of NH_3 as depicted in both Figures 4.18 and 4.19. The reduction or "attenuation" of the peak may result from nitrification of NH_3 in the shallow ground water system or dispersion (transverse, vertical, and longitudinal) or a combination of these two mechanisms. Due to the high nitrate levels in the background water it was difficult to separate out the effects of dilution.

From June 17, 1985 through June 28, 1985 an intensive 2 week sampling program was initiated to define the short term variability of wastewater and groundwater quality and to determine the effect of discrete effluent dosings on shallow groundwater flow and quality.

For this 2 week period samples were taken daily from the effluent, and wells 5S, 5M, 10 and 102. Water levels were taken daily at wells 5M, 5D, W4, W6, W7, W10, and W102 and continuously at well 5S. In-field testing of both NH_3 and chlorides was performed to aid in evaluating qualitative changes in the groundwater over time. This additional testing was not helpful because: 1) NH_3 levels in the wastewater and W5S had already dropped to less than detectable concentrations (nitrification was occurring in the lagoons due to warmer temperatures); and 2) chloride concentrations remained elevated in the wastewater and shallow well and therefore could not be used as a tracer for a particular wastewater dosing.

Little variability was measured in the wastewater and groundwater samples taken during this period. The significant result of this effort was the observed distribution of nitrogen species at the four wells which were sampled. These results are summarized in Table 4.5 for the 2 week period. NH_3 concentrations in wells 5S and 5M (20 ft. below water table) were low, reflecting the conditions

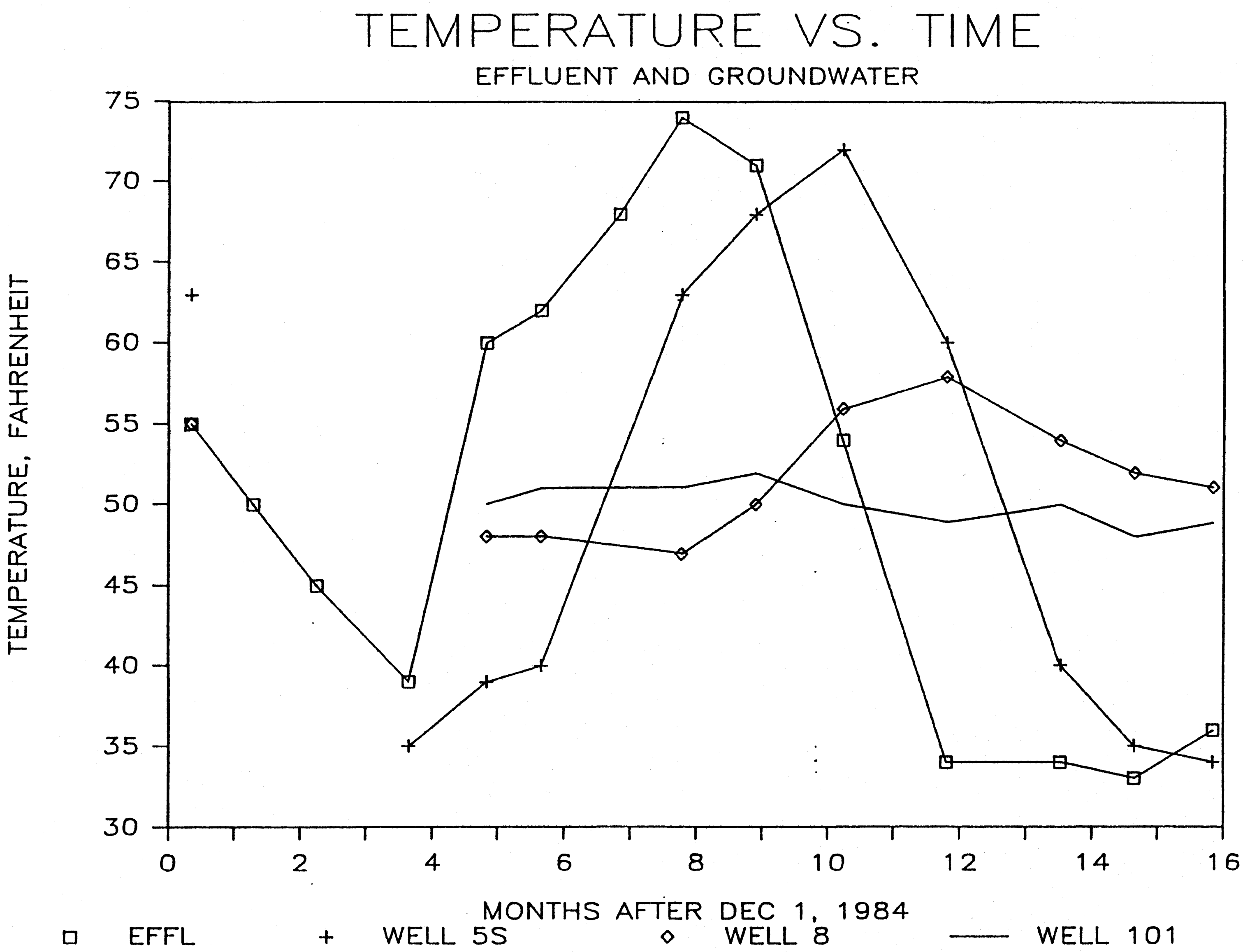

TEMPERATURE VS. TIME
EFFLUENT AND GROUNDWATER
TEMPERATURE, FAHRENHEIT
75
70
65
60
55
50
45
40
35
30
0
2
4
6
8
10
12
14
16
MONTHS AFTER DEC 1, 1984
EFFL
WELL 5S
WELL 8
WELL 101

FIGURE 4.20

TABLE 4.5

CHEMICAL DATA FROM JUNE INTENSIVE SAMPLING
EVANSVILLE RI SYSTEM

DATE ==>	WEEK 1: NW CELL LOADED 6/17 MON	6/18 TUE	6/19 WED	6/20 THU	6/21 FRI	WEEK 2: SW CELL LOADED 6/24 MON	6/25 TUE	6/26 WED	6/28 FRI
WASTEWATER									
VOLUME	0.858	-	0.691	-	0.676	0.574	-	0.531	0.554
NO3	10.1	-	9.8	-	9.6	9.0	-	8.8	9.0
NH3	<0.1	-	0.1	-	0.1	0.1	-	0.2	0.1
TKN	5.0	-	4.8	-	4.3	7.0	-	5.2	6.0
COD	94	-	71	-	76	95	-	74	98
CL	240.0	-	240.0	-	220	240	-	250	250
WELL 5S									
NO3	14.5	16.4	16.8	18	16.4	-	14.8	13.8	16.4
NH3	1.0	0.9	0.9	0.7	0.7	-	0.6	0.6	0.8
TKN	1.6	1.4	1.3	1.1	1.0	-	1	1.0	1.2
COD	-	11	13	12	9	-	8	8	18
CL	210	220	210	220	220	-	230	230	220
WELL 5M									
NO3	-	22.0	20	20.0	18.9	-	18.5	17.4	17.0
NH3	-	1.7	1.8	1.6	1.7	-	1.5	1.4	1.4
TKN	-	2.1	2.2	2.6	2.0	-	1.8	1.9	1.8
COD	-	10	10	10	9	-	6	8	13
CL	-	220	220	210	220	-	220	220	220
WELL 10									
NO3	-	26.0	27.0	29.0	28.0	-	26.0	19.8	18.0
NH3	-	7.9	8.2	7.5	7.7	-	7.3	5.4	7.4
TKN	-	8.0	8.2	7.5	8.0	-	7.6	6.5	7.5
COD	-	11	11	9	10	-	8	13	18
CL	-	210	210	210	210	-	220	230	220
WELL 102									
NO3	-	17.7	18.0	18.8	18.2	-	18.7	18.2	18.8
NH3	-	5.2	5.2	4.6	5.1	-	4.6	4.7	4.6
TKN	-	5.5	5.5	5.2	5.5	-	5.2	5.5	6.0
COD	-	11	11	10	11	-	9	9	16
CL	-	220	210	220	220	-	230	230	230

WASTEWATER VOLUME IN MILLION GALLONS
CONCENTRATIONS IN MG/L

TABLE 4.6
BACTERIOLOGICAL SAMPLING RESULTS SUMMARY

DATE==>		1985 6/25	7/24	8/28	10/7	11/25	1986 ==> 1/16	2/18	3/26
SAMPLE									
Effluent	TC:	3900	-	-	-	-	9600	5/5(u)	5/5(u)
	FC:	190	960	2200	-	1100	-	-	-
	FS:	LA	230	210	-	-	-	-	-
Well 5S	TC:	14	-	-	-	-	200	5/5(u)	(u)
	FC:	<2	<1	<10	-	<10	-	-	-
	FS:	1100	56	60	-	-	-	-	-
Well 5M	TC:	66	-	-	-	-	-	-	-
	FC:	<2	<1	-	-	-	-	-	-
	FS:	2000	1500	-	-	-	-	-	-
Well 10	TC:	28	-	-	5	-	-	-	-
	FC:	<2	<1	-	-	-	-	-	-
	FS:	1300	1100	-	-	-	-	-	-
Well 8	TC:	-	-	-	-	-	<10	3/5(u)	1/5(u)
	FC:	-	<1	-	-	<10	<10	-	-
	FS:	-	2100	-	-	-	-	-	-
Well 9	TC:	-	-	-	-	-	<100	5/5(u)	0/5(s)
	FC:	-	<1	<10	-	<10	-	-	-
	FS:	-	3000	610	-	-	-	-	-
Well 101	TC:	6	-	-	-	-	<100	0/5(s)	1/5(u)
	FC:	<2	<1	-	-	<10	-	-	-
	FS:	400	680	-	-	-	-	-	-
Well 11	TC:	-	-	-	-	-	-	-	-
	FC:	-	<1	-	-	-	-	-	-
	FS:	-	400	-	-	-	-	-	-
Well 12	TC:	-	-	-	-	-	-	-	0/3(s)
	FC:	-	-	-	-	-	-	-	-
	FS:	-	-	-	-	-	-	-	-
Well 13	TC:	-	-	-	-	-	-	-	1/3(u)
	FC:	-	-	-	-	-	-	-	-
	FS:	-	-	-	-	-	-	-	-
Well 102	TC:	-	-	-	-	-	-	-	1/5(u)
	FC:	-	-	-	-	-	-	-	-
	FS:	-	-	-	-	-	-	-	-

Units= #/100 ml; TC=total coliform, FC=fecal coliform, FS=fecal strepp

Fermentation Tube method used for total coliform group, for 2/18 and 3/26 samples, u = unsafe, s = safe (per drinking water standards) measured in the wastewater while NH_3 concentrations in water table wells 10, and 102 downgradient showed elevated concentrations. Previous sampling showed low levels of NH_3 in the wastewater since at least the third week in May. This suggests at minimum a 1 month response time for effects of the effluent in the vicinity of wells 10 and 10S. The lag time was 3-4 months based upon groundwater flow velocities.

Total nitrogen measured during this period in well 5M and well 10 were greater than at well 5S. These wells were also affected by the additional input of nitrate from mixing with upgradient groundwater.

The following discussion is a summary of results of additional chemical testing that was performed on the effluent and groundwater samples during the entire study. A summary of these and corresponding sampling frequencies was presented in Table 3.1 of Section 3.

BOD concentrations were generally below the detection limit of 3 mg/l in the groundwater with the exception of well 5S. A concentration of approximately 6 mg/l BOD was measured in well 5S in February and March, 1985 and 4.3 mg/l BOD_5 in June 1982. Otherwise levels were below 3.0 mg/l BOD_5 for the sampling period. It was concluded that although the sand and gravel soils were inefficient for removing nitrogen, they were effective in removing the amounts of BOD that were applied.

Phosphorus removal in the unsaturated zone was approximately 70%, based on concentrations ranging from 2.4 to 4 mg/l total P in the effluent and .44 to 1.9 mg/l total P in well 5S. Concentrations at distances of 250 ft. were below detection limit.

Bacteriological data including fecal coliform, total coliform and fecal strep were taken later in the study. Results of these tests are summarized in Table 4.6. Fecal coliforms were not detected in any of the wells. High values for fecal strep in the background wells (up to 3000/100 ml) indicated that the wells themselves were not suitable for bacteriological sampling. Total coliform tests (fermentation tube method) identified wells 5S, 8, 13 and 102 as being bacteriologically unsafe. This was based on two samplings except well 13 which was sampled only once. It is also worth noting that well 101 (background) was also identified as unsafe in one of two samplings while well 9 (deeper background) was safe for both samplings. No conclusion as to the impact from the seepage cell system can be made from so few samples and due to results from well 101.

SECTION 5 BRUNKOW - PRESENTATION AND DISCUSSION OF RESULTS

5.1 System and Site Description

The Brunkow Cheese factory located 5 miles northeast of Darlington, Wisconsin has utilized land application methods for disposing of washwater wastes for approximately 13 years. The site location is shown in Figure 5.1. In 1980 two absorption ponds were constructed to receive and dispose of the washwater wastes by seepage to the groundwater. In May 1984 a third pond was constructed to increase the capacity of the system. Prior to 1980 the washwater was spray irrigated to nearby agricultural fields. At this time the wastewater runoff from the spray irrigation system was held in the existing effluent ditch and two smaller ponds which were located just prior to the existing ponds.

During 1985 the factory received an average of 50,000 lbs of milk per day. During peak production in May and June the plant processed 70,000 lbs per day. Lowest production levels occurred in November and December when 40,000 lbs per day was received.

The cheese factory generated an average of 3000 gallons of wastewater per day. The wastewater consisted primarily of washwater from floors and equipment. The washwater was collected by floor drains around the plant which fed to a 2000 gallon cement holding tank. The wastewater was discharged through an overflow pipe from the tank to a pipe which ran east under the road and approximately 1000 ft. downslope to a valley. It then flowed into an open ditch, in which it travelled about 900 ft. south to the absorption ponds.

The cells were constructed in an alluvial valley one quarter mile southeast of the factory and 75 to 100 ft. west of a small, perennial stream which flows into Otter Creek.

Figure 5.2 is a plan view of the site showing the configuration of the absorption ponds and location of the monitoring equipment and stream. Pond 1 receives all of the wastewater and serves as both a settling and seepage pond. The sludge layer in pond 1 was 6 to 12 inches thick during this study. Initially the ponds were operated in series by overflow pipes. In June 1985, 2 inch inside diameter PVC outlet pipes with adjustable gate valves were installed. Discharge was then alternated between ponds 2 and 3 to operate the system on a rest and load cycle. Areas of the ponds measured on 9/10/85 were .13 acres for pond 1, .098 acres for pond 2, and .112 acres for pond 3.

Soils at the site consisted of 0 to 3 ft. of black loamy topsoil, over 7 to 10 ft. of silt and silty clay loam over 80 to 120 ft. of hard limerock (Galena limestone). The well log for the water supply well at the factory showed topsoil and clay from ground level down to 7 ft., hard limerock from 7 ft. down to 41 ft. (Galena Formation), Upper Platteville limerock from 41 to 75 ft., Trenton limestone from 75 to 130 ft., St. Peter sandstone from 130 ft. to 196 ft. and Magnesia limestone from 196 down to 284 ft. The well was screened in the Magnesia limestone.

The ponds were excavated into the silt layer. Cross sections based on boring logs from this study are shown in Figure 5.3. Soils data taken from boreholes during well drilling showed 30 to 44% silt, 4 to 30% clay, 10 to 56% sand, a nitrogen content of .18%, and organic content of .2 to 4.4% and a CEC ranging from 4 to 12 meq/100g. Soils with higher percentages of sands occurred in samples taken just above the limestone bedrock. These data and corresponding sampling depths are presented in Appendix B.2. Borings taken just west of the pond showed 10 to 13 ft. of silt loam above bedrock with no water table encountered.

The water table was located 3 to 5 ft. below the ground surface near the stream. The depth to the water table was estimated to be 5 to 6 ft. below the base of the ponds from borings taken downgradient and around the ponds.

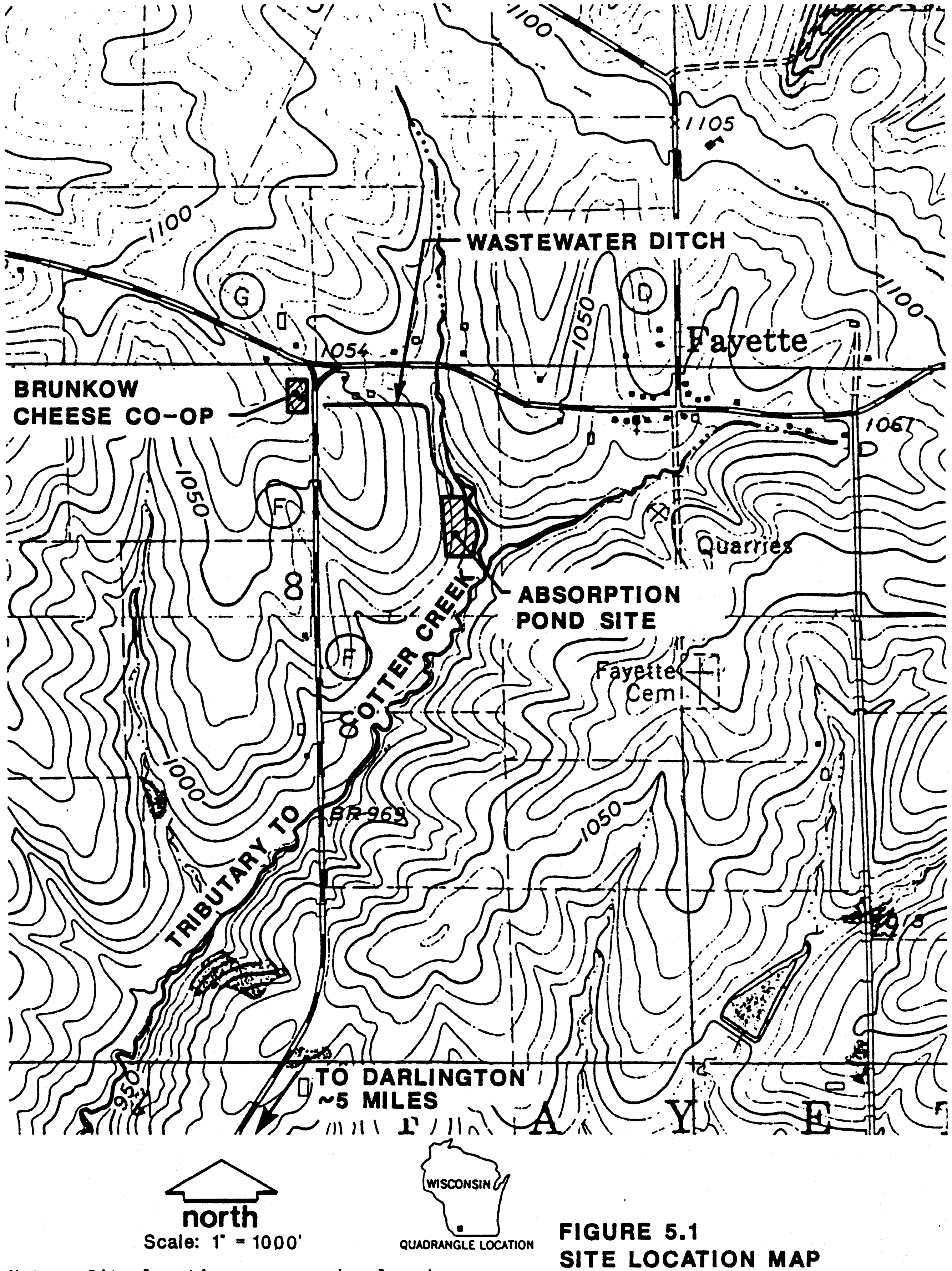


FIGURE 5.1
SITE LOCATION MAP
BRUNKOW CHEESE CO-OP
ABSORPTION POND SITE
DARLINGTON
LAFAYETTE COUNTY, WISCONSIN

Note: Site location map was developed from the U.S.G.S. 7½ minute quadrangle maps, Waldwick and Darlington, Wisconsin, dated 1962.

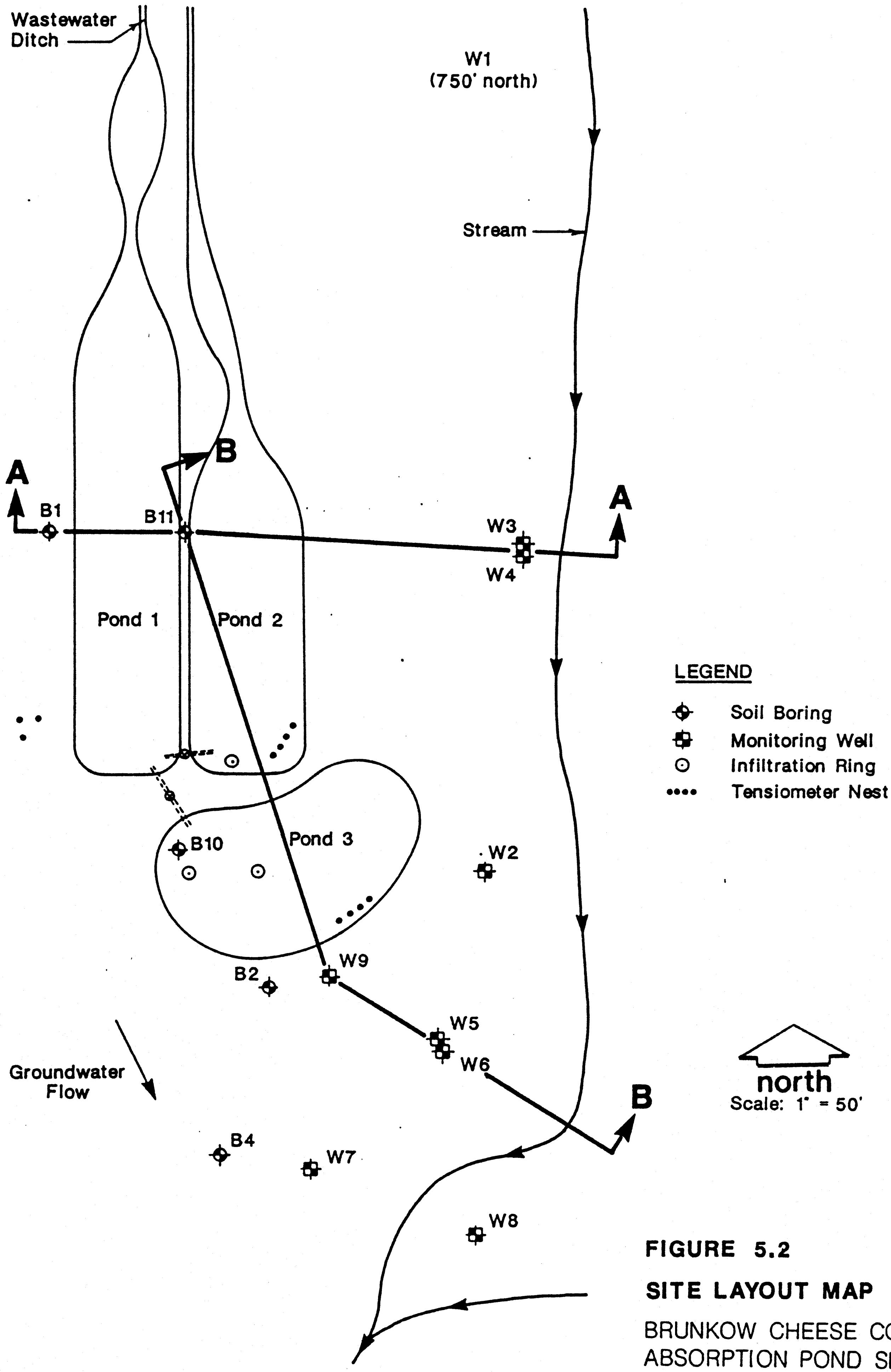


FIGURE 5.2

SITE LAYOUT MAP

BRUNKOW CHEESE CO-OP
ABSORPTION POND SITE

5.2 Wastewater Characterization

At the outset of the study very little was known about the quality or quantities of wastewater being discharged to the absorption ponds. Monthly grab samples were taken from the tank at the plant (untreated wastewater) and from the 3 absorption ponds.

In general the wastewater exhibited BOD concentrations ranging from 3500 mg/1 to 15,000 mg/1 with a mean of 8100 mg/1 BOD_5. TKN's ranged from 150 to 590 mg/l as N with a mean of 280 mg/l as N. Organic nitrogen was the dominant form present in the wastewater, and the pond wastewater comprising 91 to 95% of the total nitrogen content.

Mean values of additional selected parameters for the untreated plant wastewater are listed in Table 5.1. Wastewater quality data for all parameters tested is listed in Appendix B.1. For comparison, parameter averages from the pond samples are also listed in Table 5.1.

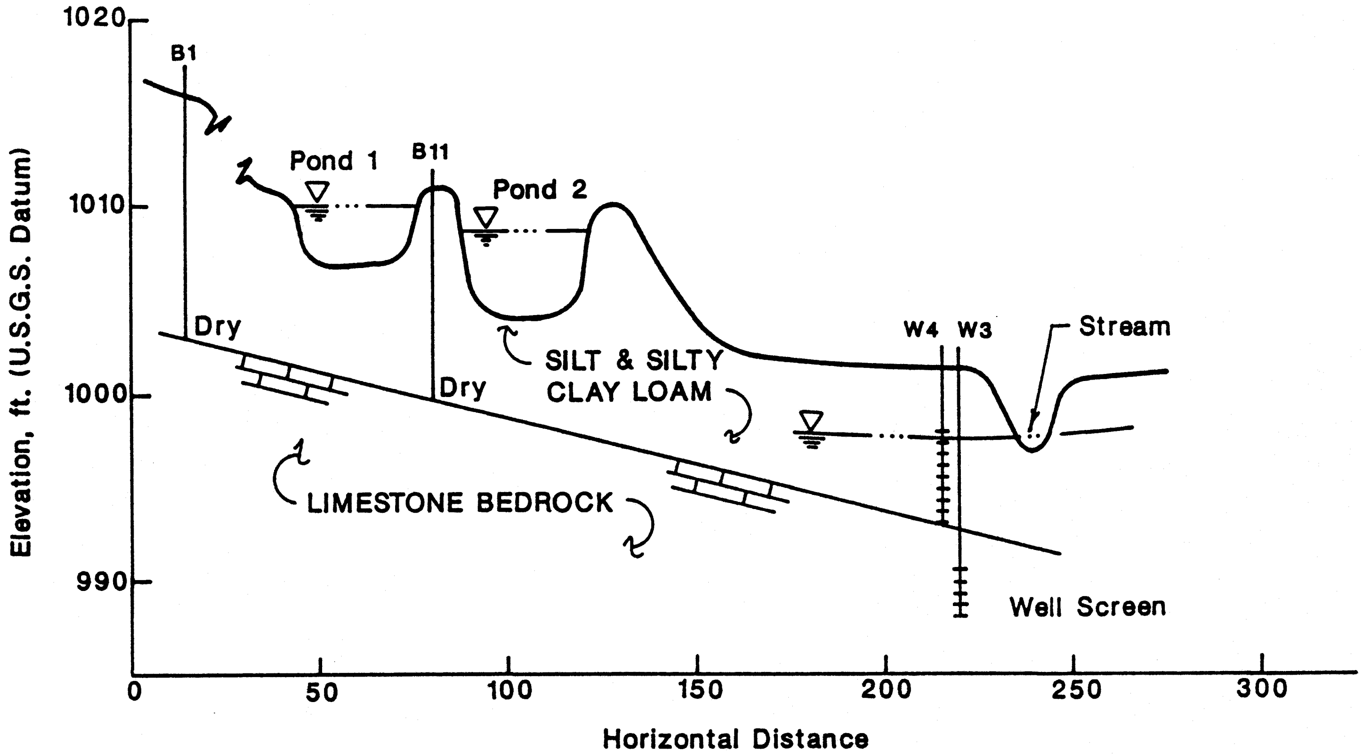
In addition to the overall high strength of the waste compared with other dairy wastes, two general observations of the wastewater data can be made. First there is great variability in the data as depicted by the standard deviations. Part of this variability results from "grab" sampling. Second, there is an increase in total nitrogen in ponds 1 and 2 over the total nitrogen concentration in the wastewater. This result is discussed below.

Pond samples were taken .5 to 1 ft. below the surface and 3 to 4 ft. from the pond edges. Total BOD_5, COD and chloride concentrations in pond 1 were similar to the untreated plant wastewater. Levels of these parameter decreased from 2 to 5 times in ponds 2 and 3. The decrease in BOD_5 was attributed to settling in pond 1 since the suspended solids level decreased by a factor of about 4 compared to untreated plant wastewater concentrations.

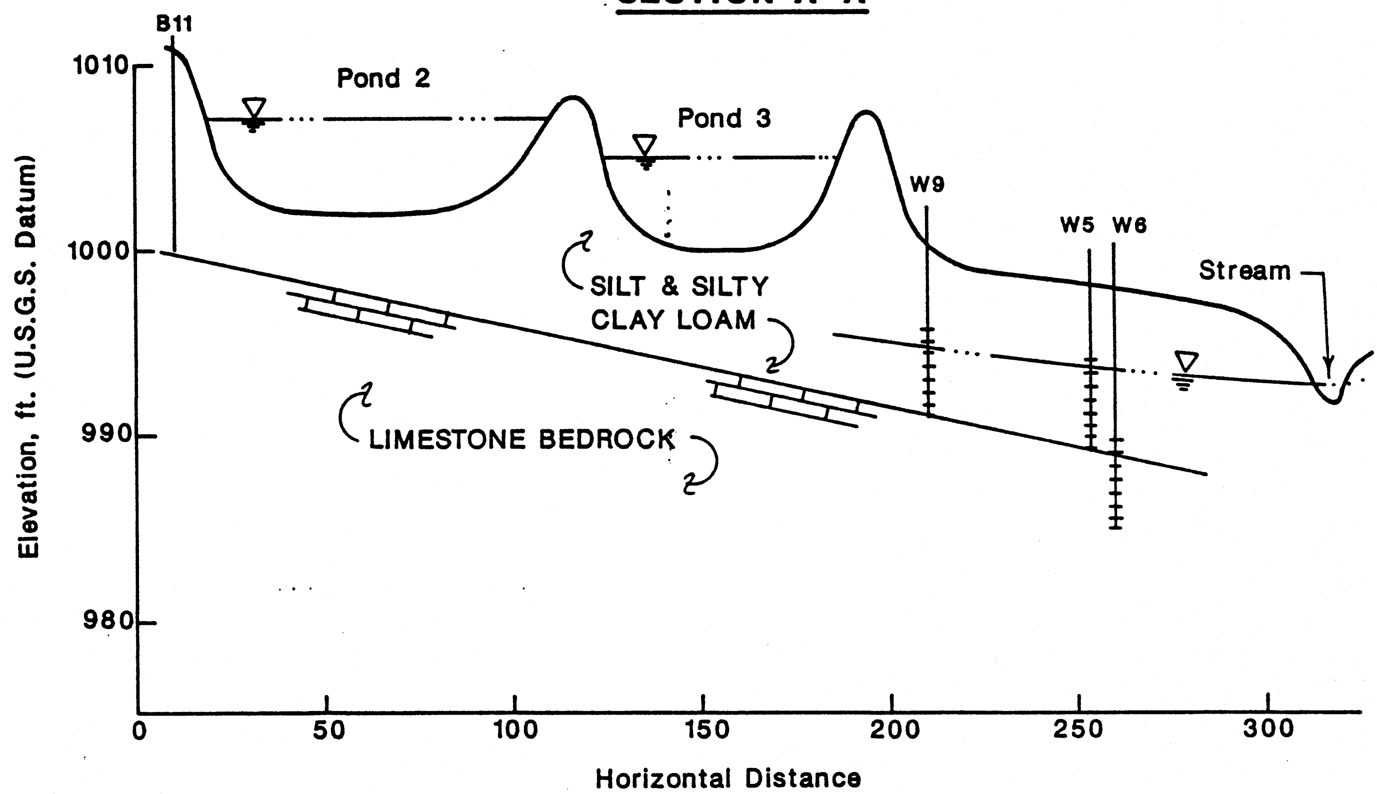
Pond samples were taken .5 to 1 ft. below the surface and 3 to 4 ft. from the pond edges. Total BOD_5, COD and chloride concentrations in pond 1 were similar to the untreated plant wastewater. Levels of these parameters decreased from 2 to 5 time in ponds 2 and 3. The decrease in BOD_5 was attributed to settling in pond 1 since the suspended solids level decreased by a factor of about 4 compared to untreated plant wastewater concentrations.

Total nitrogen in pond 1 increased over the levels measured in the untreated plant wastewater Figure 5.4 illustrates the total nitrogen measured at all wastewater sampling points. These higher concentrations were attributed to additional nitrogen inputs from overflow of a manure storage pit at a nearby hog rearing barn and runoff from surrounding crop fields. The barn was 500 ft. upslope of the effluent ditch. Overflows from the manure storage pit ran downhill and were intercepted by the ditch. Farm fields on the hill upslope from the ditch and ponds were periodically spread with whey for fertilizer. Both of these operations contributed nitrogen to the pond water. By mid summer the hog barn manure pit was pumped weekly to eliminate the overflows.

NH_3 was the dominant form of nitrogen in the pond water. There was little to no NO_3 detected in the pond water. Due to the anaerobic conditions, there was essentially no nitrification in the ponds.



SECTION A-A



SECTION B-B

Scale

Vertical: 1" = 10'

Horizontal: 1" = 50'

NOTES

1. Water table elevations measured on 6/27/85.
2. Pond elevations are approximate.

FIGURE 5.3
CROSS SECTIONS
A-A AND B-B
BRUNKOW CHEESE CO-O
ABSORPTION POND SITE

Table 5.1
Brunkow Wastewater Parameter Values
Mean and standard Deviations
(Dec. 1984 - Oct. 1985)

Parameter	Concentrations (mg/l)* Plant	Pond 1	Pond 2	Pond 3
BOD_5	8100 (+/-3000)	8300 (+/-4000)	4000 (+/-1500)	2200 (+/-1100)
COD	13000 (+/-5400)	10500 (+/-2600)	5700 (+/-2100)	2700 (+/-1300)
TSS	1190 (+/-470)	290 (+/-75)	120 (+/-60)	110 (+/-80)
Chloride	760 (+/-440)	760 (+/-160)	680 (+/-190)	540 (+/-160)
TKN	280 (+/-112)	426 (+/-160)	353 (+/-200)	172 (+/-85)
NH_3	20 (+/-7)	226 (+/-100)	293 (+/-150)	172 (+/-70)
NO_3	<1	<1	<1	<1

*Values based on 10 samples fro Raw WW, 11 samples for pond 1, 9 samples for pond 2, and 8 samples for pond 3.

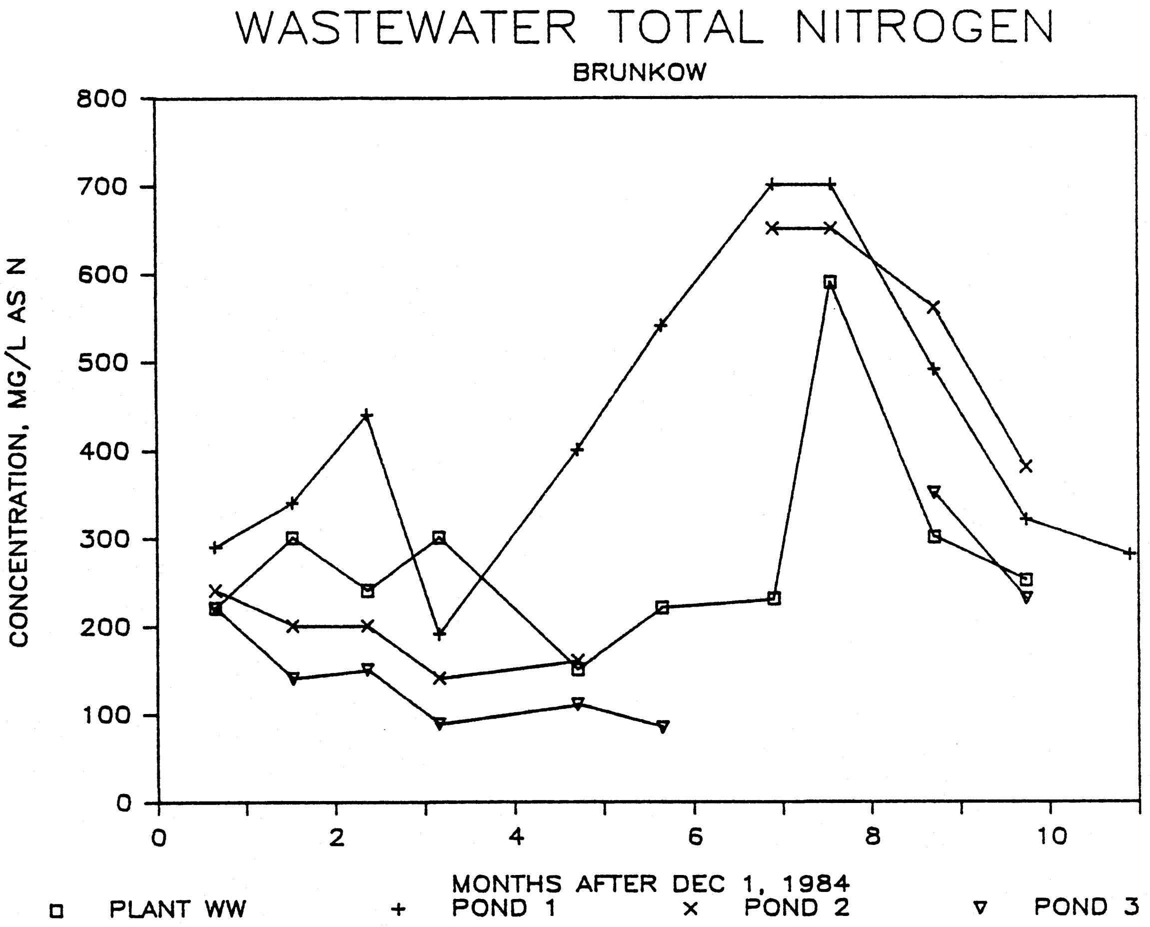

WASTEWATER TOTAL NITROGEN
BRUNKOW
CONCENTRATION, MG/L AS N
800
700
600
500
400
300
200
100
0
0
2
4
6
8
10
MONTHS AFTER DEC 1, 1984
PLANT WW
POND 1
POND 2
POND 3

FIGURE 5.4

5.3 Loading to the System

Due to the low and variable flows it was difficult to accurately determine volumetric flow rates to the absorption ponds. The wastewater flow in the effluent ditch was too low to be measured by a meter or weir. Flows were reported to DNR based on monthly averages of the amount of water used from the water supply well at the plant (with a correction for the amount used for cooling and the pasteurizing process). This cooling water was discharged to a surface water stream and was not part of the wastewater flow. The plant used an average of 5500 gpd of well water.

To estimate actual wastewater flows the tank at the plant was emptied and the time to refill the tank was recorded on 9/13/85. The day the tank was pumped represented an average operating day at the plant for both wastewater generation and water usage. The amount of wastewater generated that day was 3000 gallons. The average amount of water used in September (as measured from the metered water supply well) was 5350 gpd.

The amount of wastewater generated (3000 gpd) was 56% of the total amount of water used from the well. Then a 1985 average usage rate of 5500 gpd corresponds to a wastewater flow rate of 3080 gpd.

Because of seasonal fluctuations in the amount of water used for cooling, monthly estimates of wastewater generation rates based on a single measurement in September could be misleading. Thus an average of 3000 gpd as determined above was used for the loading rate calculations to follow.

To summarize the loading to the system for 1985 the average concentration of selected parameters from TAble 5.1 were converted to mass loadings by using the average daily volumetric rate of 3000 gpd. These are given in Table 5.2. Unit area loading rates are calculated over the total pond area (includes all 3 ponds) of .33 acres.

TABLE 5.2

System Loading Rates - Brunkow
(1985 Average)

Parameter	Loading
Flow-gal/day (gpd)	3000 (9090)
BOD-lb/day (lb/ac/day)	203 (615)
TSS-lb/day (lb/ac/day)	29 (87)
Total N-lb/day (lb/ac/day)	7 (21)

From these data it was concluded that organic loading rates were excessive for this type of system as compared to system design requirements listed in Table 2.1. Observations that were made during field visits offer some insight as to the reason for the high organic strength of the waste. First it was noted that drains located near the whey holding tank received spillages of whey when trucks were loaded. Second, there were occasional milk spills at the plant which were washed down to floor drains. Both of these sources have low volumes yet very high BOD concentrations.

5.4 Pond Seepage Rates

To evaluate groundwater contamination it was important to measure seepage (pond outflow) rates as well as estimate pond inflow rates. The ponds were hydraulically overloaded and had to be pumped approximately once per year. In May 1985 liquid and unknown quantities of sludge in the ponds were pumped out to avoid over topping the berms. In May 1986 the ponds were pumped again although some storage remained in pond 3 (approximately one third of the volume).

A water balance was performed to estimate seepage rates to the groundwater. Flow to the ponds was blocked off from September 11 through September 16, 1985 by placing a barrier in the ditch. The wastewater accumulated in the ditch. For these 6 days pond level measurements were taken from staff gauges. A longer study period was not possible due to rising levels of wastewater in the ditch upstream of the barrier. Evaporation estimates of .1 inch per day were made for the 6 day period based on average daily temperatures taken at Darlington and solar radiation data from Madison. The method used to estimate evaporation was developed by Priestly and Taylor and referenced by Tanner (1983). Advective losses were not considered. An outline of the method used is included in Appendix D.2. The amount of seepage in each pond was determined by the total drop in pond level minus the depth decrease due to evaporation. There was no precipitation between 9/11/85 and 9/16/85. The results are discussed below. Calculations are presented in Appendix D.2.

Levels decreased at the rate of 0.15 inch/day, 0.35 inch/day, and 0.2 inch/day in ponds 1, 2 and 3 respectively. Subtracting 0.1 inch/day due to evaporation losses resulted in estimated seepage rates of 0.05 inch/day, 0.25 inch/day, and 0.1 inch/day in ponds 1, 2 and 3 respectively. The water depths in ponds 1, 2 and 3 were 3 ft, 4 ft, and 2 ft, respectively. The seepage in pond 1 was the smallest as it had the thickest solids layer on the bottom. The water depth of only 2 ft. in pond 3 may have caused its low seepage rate. These seepage estimates are subject to error in both measurement and estimation of evaporation.

The seepage rates were converted to volumetric rates using the pond areas measured on 9/10/85. Pond 1 was seeping at a rate of 176 gpd, pond 2 at 667 gpd and pond 3 at 305 gpd. The total estimated seepage was 1150 gpd. The change in area due to the decreases in pond levels was assumed negligible, as the total pond level decrease was less than 1.75 inches during the test period. For an inflow of 3000 gpd and an outflow of 1150 gpd (equivalent to the seepage rate), the detention time in the pond system, with a total volume of 540,000 gallons, was approximately 290 days. This assumes evaporation and precipitation cancel over one year. From information provided by the owner of the cheese factory the seepage ponds have to be pumped every 12 to 14 months. This discrepancy of 2 to 4 months could be due to under estimation of seepage rates, over estimation of inflow rates, the assumption that precipitation equalled evaporation, or under estimation of the total pond volumes.

Nitrogen loading rates to the groundwater, based on the estimated seepage rates and the average measured concentration in each pond, are summarized in Table 5.3.

TABLE 5.3

Nitrogen Loading Rates - Brunkow
Based on Seepage Estimates from 9/83

	Seepage Rate (gpd)	Nitrogen Loadings (lb/day	(lb/acre day)
Pond 1	175	.57	4.4
Pond 2	670	1.77	18.0
Pond 3	305	.41	3.7
Total	1150	2.75	

These calculations were made using TKN values (NH_3 + organic N), NO_2 + NO_3 concentrations were below detection limit and thus were assumed negligible. Dissolved TKN's were used in determining the loading rates to enable comparison with the groundwater samples, which were tested for dissolved parameters (filtered). The dissolved TKN fraction of the wastewater was approximately 90% of the total TKN. Details of the nitrogen loading calculations are listed in Appendix D.2.

Nested tensiometers placed at depths ranging from 1 to 5 ft. below the ponds showed tensions at or very close to zero most of the time. The deepest tensiometers always read "0" centibars (no tension or nearly saturated conditions). Tensiometers that were placed upgradient of the ponds measured tensions ranging from 30 to 80 centibars (10.2 to 27.2 ft. H_2O). These data suggested that the soil beneath the ponds was at or very near saturation and that wastewater from the ponds was flowing to the groundwater. This conclusion was confirmed by the high chloride concentrations in wells 5 and 9 (discussed later). An estimate of vertical flow conditions below the ponds can be made by assuming that the measured infiltration rates govern the flow rate. Assuming a porosity of .40 (for saturated conditions), a unit gradient, and an average infiltration rate of .15 inches/day (based on measured values ranging from .05 to .25 in/day) yields a pore velocity of .375 inches/day. Over a 5 ft. depth of soil the ravel time would be 160 days.

5.5 Groundwater Flow System

In general groundwater flow was in the south to southeast direction. Figure 5.5 shows water table contours based on measurements taken September 22, 1985. Although seasonal fluctuations in elevations were observed between November 1984 and October 1985, the general pattern remained unchanged.

A water table fluctuation of 2.14 ft. was measured in the background well (W1) during the period of study. This was caused by seasonal recharge.

Horizontal gradients within the silt layer averaged 0.02 ft/ft between wells 4 and 2. 0.014 ft/ft between wells 2 and 5, 0.015 ft/ft between wells 9 and 5, and 0.02 ft/ft between wells 9 and 7. Horizontal gradients between wells 3 and 6 screened in the bedrock were 0.023 ft/ft. Hydraulic conductivities determined from slug test ranged from .42 ft/day (1.5×10^{-4} cm/sec) to 0.98 ft/day (3.5×10^{-4} cm/sec) in the silt layer and were 0.86 ft/day (3.0×10^{-4} cm/sec) in the bedrock layer. Results of the slug tests are listed with well specifications in Appendix B.2.

For an average horizontal gradient of .02 ft/ft, the horizontal flow of the groundwater in the silt layer ranged from 8.4×10^{-3} (.25 cm/day) to 1.96×10^{-2} ft/day (.59 cm/day).

For an assumed porosity of .40 the pore velocity ranged from .021 ft/day (.63 cm/day) to 0.49 ft/day (1.5 cm/day). At these rates it would take 200 to 474 days for the groundwater to travel 10 feet. From this it is apparent that the calculated travel times could be in error by a year due to the variability in hydraulic conductivities as determined by the slug tests. This result will be discussed further in connection with the solute concentration results in the next portion of this section.

The two sets of nested wells showed vertical gradients but in opposite directions. Nested wells 3 and 4 located east of pond 2 (approximately 80 ft.) showed upward vertical gradients ranging from .019 to .028 ft/ft. Well 4 was screened at the water table in the silt layer (5 ft. screen). Well 3 was screened 6 ft. into the bedrock layer (2.5 ft. screen).

Nested wells 5 and 6 located 50 ft. southeast of pond 3 showed downward gradients ranging from .05 ft/ft to .09 ft/ft. Well 5 was screened in the silt layer. Well 6 was screened 10 ft. into the bedrock. Elevation data for wells 3, 4, 5, and 6 for the sampling period are illustrated in Figure 5.6. The cause for this shift in the direction of the gradient between the two nested wells (185 ft. apart) is not entirely clear from the data available. A correlation of the chemical data with elevation data explains in part the significance of these gradients (discussed in the next portion of this section).

Examination of the water table contours in Figure 5.5 does not reveal any evidence of mounding due to recharge from the ponds. Limitations as to where wells could feasibly be located prevented a more complete analysis of the effects of the ponds on groundwater flow.

Elevation data and soils data suggest that the groundwater in the silt layer feeds directly to the stream. From soil boring data and stream elevations, the depth to bedrock below the stream south of the ponds is approximately 5 ft.

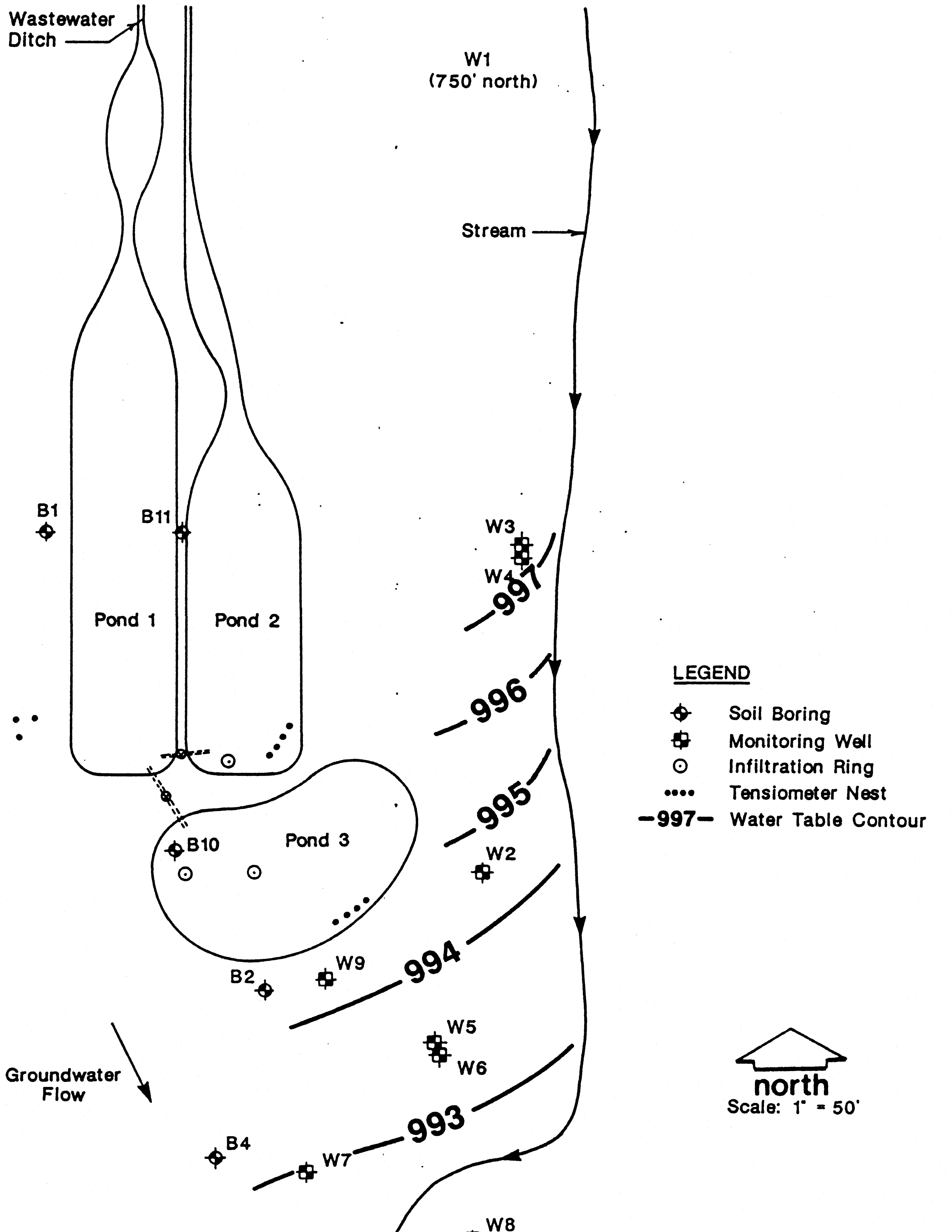


FIGURE 5.5
WATER TABLE CONTOURS
SEPTEMBER 22, 1985
BRUNKOW CHEESE CO-OP
ABSORPTION POND SITE

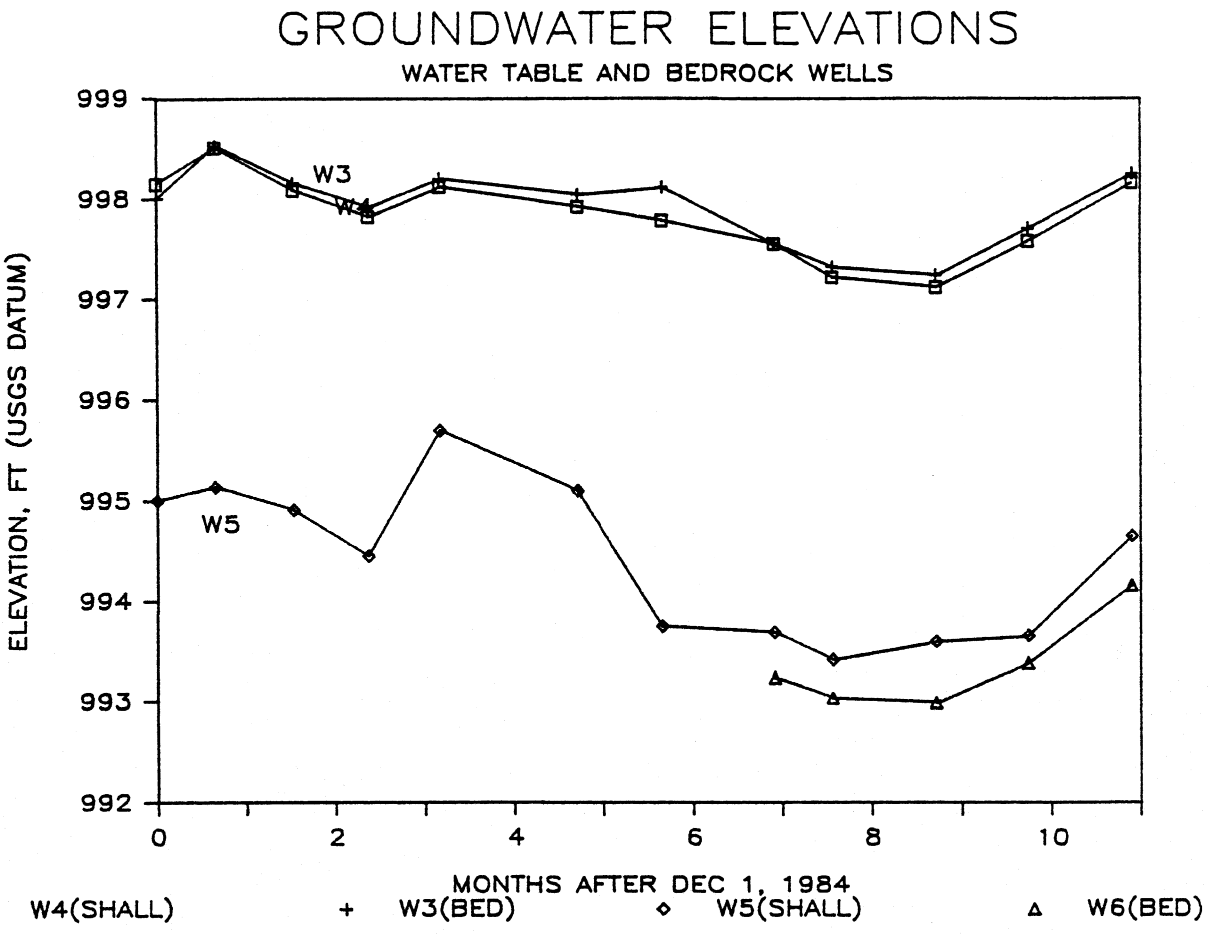

GROUNDWATER ELEVATIONS
WATER TABLE AND BEDROCK WELLS
ELEVATION, FT (USGS DATUM)
999
998
997
996
995
994
993
992
0
2
4
6
8
10
W3
W4
W5
MONTHS AFTER DEC 1, 1984
W4(SHALL)
+ W3(BED)
◇ W5(SHALL)
Δ W6(BED)

FIGURE 5.6

5.6 Groundwater Solute Distribution

Mean values for selected parameters from the groundwater monitoring wells are listed in Table 5.4. The raw data for all wells and parameters tested at the Brunkow site are listed in Appendix B.1.

Iso-concentration contours representing the spatial distribution of chlorides in the silt layer are illustrated in Figure 5.7. The contours are drawn from data taken on June 27, 1985. A specific data was selected due to the change in chloride concentrations that was observed over time. To illustrate this temporal change, chloride concentrations vs time are shown in Figure 5.8 for three wells (W2, W4, and W5). There was a significant decrease in the chlorides in all three wells, beginning in May 1985 for wells 2 and 4 and in June 1985 for well 5. Additional data for consecutive years would be required to determine the cause of this decrease.

Wells 9, 5 and 7 showed the most effect from the wastewater as indicated in the chloride contour plot (Figure 5.5) These wells were located directly downgradient of the ponds. Samples from well 9 exhibited an average chloride concentration of 430 mg/l compared to an average chloride concentration of 540 mg/l in Pond 3. This suggests a dilution of approximately 20% at this location (20 ft. downgradient of pond 3).

Wells 1, 3, 6 and 8 were unaffected by the wastewater. This was evident from the lower chloride and dissolved solids concentrations. Wells 1 and 8 were screened at the water table and represented background conditions. Chemical parameters in the bedrock wells (W3 and W6) showed concentrations similar to the background wells.

As was discussed previously, the vertical gradient at the W4 and W3 nest was upward in direction and the vertical gradient at the W5 and W6 nest was downward. From the chemical data it appears that the bedrock groundwater quality is unaffected by the shallow system regardless of gradient. This negligible exchange between the silt and bedrock strata suggests that the two are hydraulically disconnected.

The background and bedrock NO_3 concentrations were around 10 mg/l while the NO_3 concentrations were below detection limits in wells in the silt layer downgradient of the ponds (W5, and W7). Figure 5.9 depicts the distribution of the dominant nitrogen species in the shallow silt layer as compared to the bedrock groundwater. A similar pattern was measured in nested wells 3 and 4. It was concluded that the effect of the wastewater plume on the shallow groundwater system was an overall increase in most parameters except for a decrease in the NO_3 concentrations. This lower NO_3 concentration was attributed to the anaerobic conditions present below the ponds and in the wastewater plume.

A further evaluation of the NH_3 concentrations supported this conclusion. Figure 5.10 depicts the spatial distribution of NH_3 in the shallow groundwater system (based on wells screened in the silt layer). These contours were drawn based on results from samples taken on June 27, 1985. Similar patterns existed for other monthly samplings. NO_3 concentrations were generally at or below the detection limit in wells 9, 5 and 7. Organic nitrogen typically comprised 5 to 10% of the TKN.

These data suggest that there was a significant reduction in the NH_3 and other constituent concentrations in the groundwater samples in the silt layer, compared to the wastewater samples. To illustrate this result, average concentrations of selected parameters are summarized in Table 5.5 for pond 3 and the two most impacted wells.

Table 5.4

GROUNDWATER CHEMICAL PARAMETERS - BRUNKOW
Mean and Standard Deviations
November 1984 - October 1985

Well		BOD	COD	TDS	TKN	NH_3-N	NO_3-N	Cl
1	mean:	3.2	8	535	0.43	0.13	9.5	50
	std dev:	(1.5)	(4)	(185)	(0.07)	(0.09)	(4.3)	(20)
2	mean:	3.9	16.4	630	1.0	0.6	0.6	190
	std dev:	(1.2)	(9.8)	(240)	(0.4)	(0.3)	(0.8)	(100)
3	mean:	3.6	7.5	510	0.6	0.3	7.6	55
	std dev:	(1.1)	(3.3)	(160)	(0.3)	(0.3)	(2.7)	(22)
4	mean:	8.3	19	670	3.2	2.8	0.2	170
	std dev:	(7.4)	(12)	(260)	(0.4)	(0.4)	(0.1)	(82)
5	mean:	6.3	23	1020	5.5	5.0	0.1	320
	std dev:	(3.3)	(8)	(240)	(0.9)	(1.0)	-	(100)
6	mean:	<3.0	6	554	0.5	0.2	8.0	70
	std dev:	-	(2)	(41)	(0.2)	(0.1)	(1.8)	(20)
7	mean:	4.7	17	870	1.9	1.3	0.1	300
	std dev:	(1.4)	(2)	(160)	(0.4)	(0.3)	-	(30)
8	mean:	3.3	7	500	0.2	0.1	8.8	60
	std dev:	(0.3)	(3)	(16)	(0.1)	-	(1.0)	-
9	mean:	12.8	40	1260	13.9	12.1	3.1	430
	std dev:	(4.7)	(9)	(42)	(4.2)	(3.2)	(6.0)	(20)

Table 5.5

Comparison of Wastewater Parameters
to the Groundwater

Parameter* (mg/l)	Pond 3	Well 9	Well 5
Chloride	540	430	320
BOD	2200	13	6
COD	2700	40	23
TKN	172	14	5.5
NH_3	148	12	5.0

(NO_3 was less than detection at all three locations).
*Values are averages for 12/84-9/85.

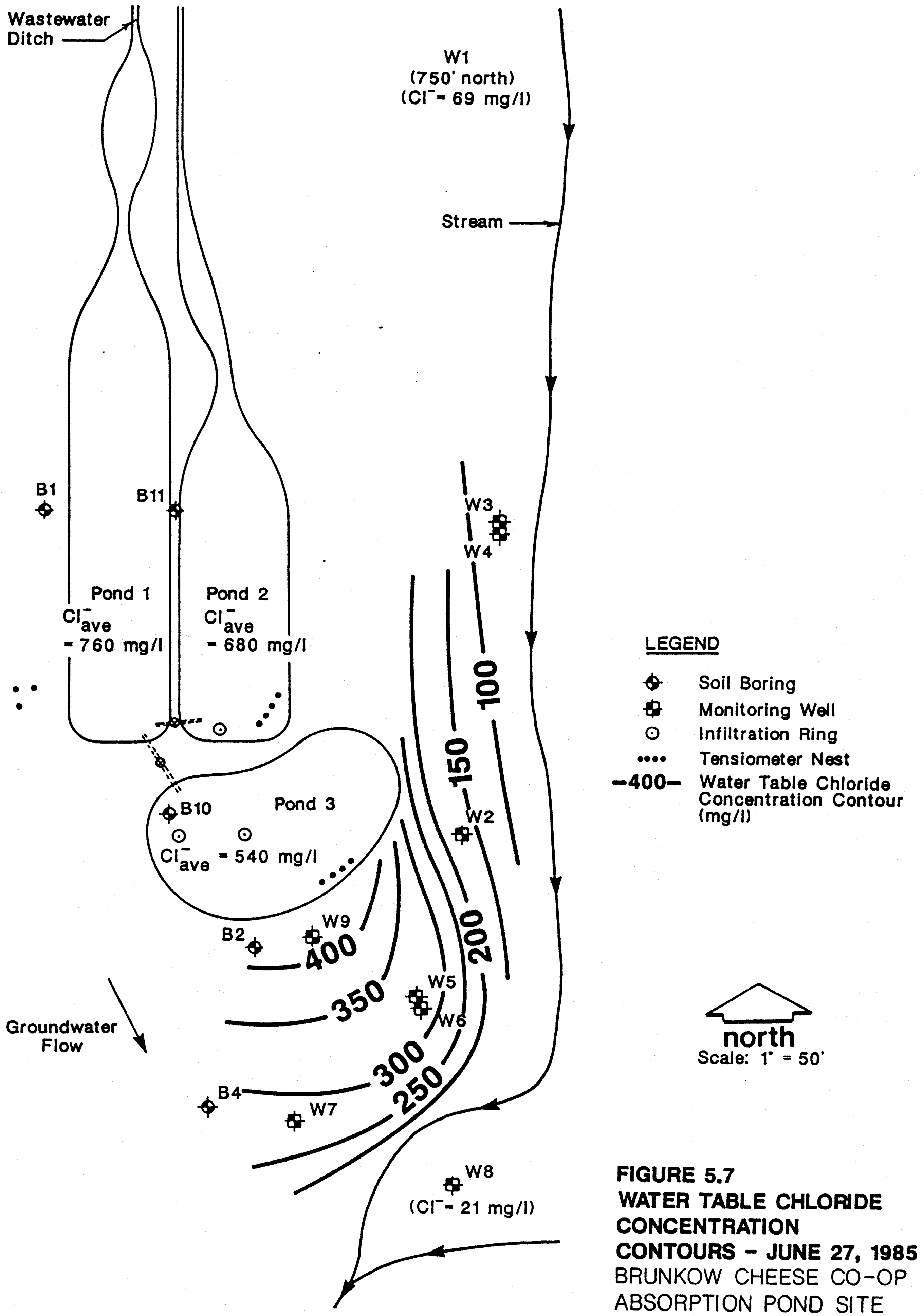


FIGURE 5.7
WATER TABLE CHLORIDE CONCENTRATION CONTOURS - JUNE 27, 1985
BRUNKOW CHEESE CO-OP ABSORPTION POND SITE

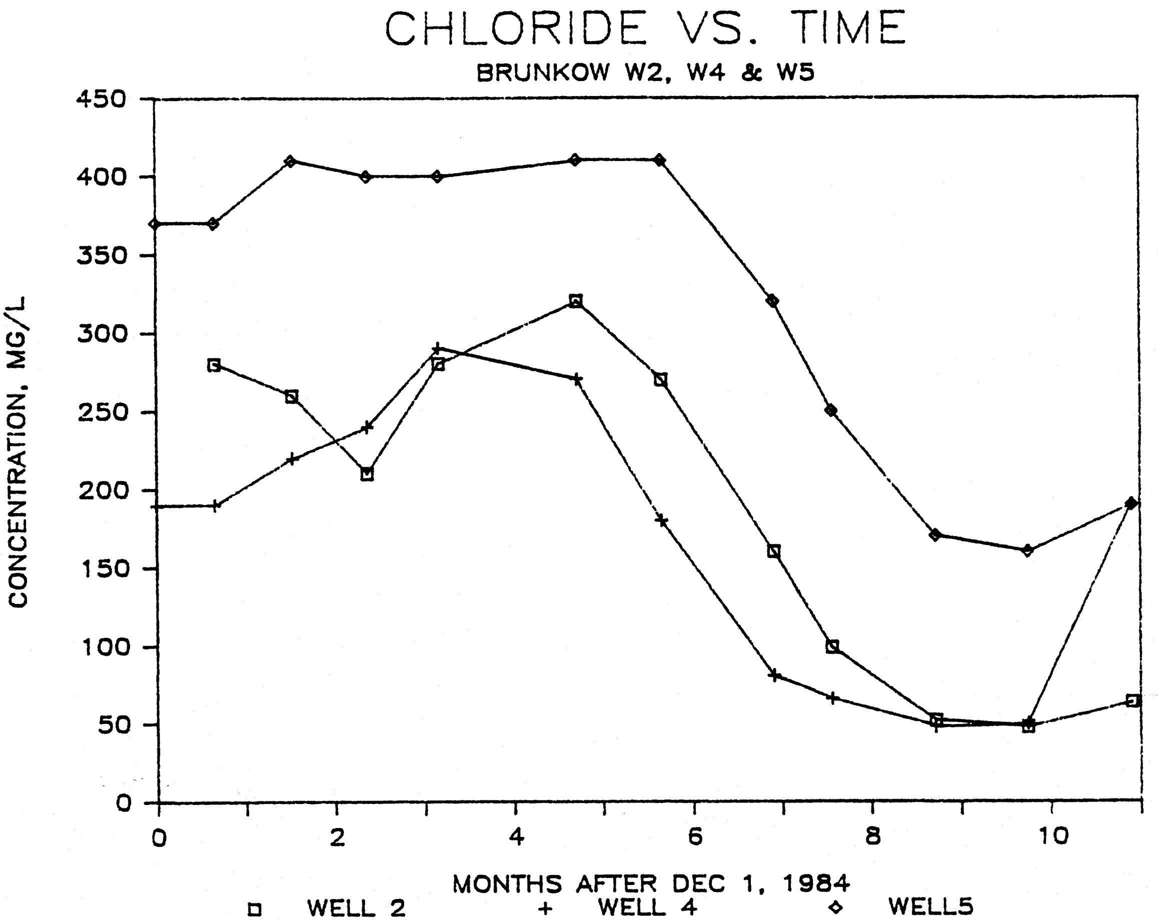

CHLORIDE VS. TIME
BRUNKOW W2, W4 & W5
CONCENTRATION, MG/L
450
400
350
300
250
200
150
100
50
0
0
2
4
6
8
10
MONTHS AFTER DEC 1, 1984
WELL 2
WELL 4
WELL5

FIGURE 5.8

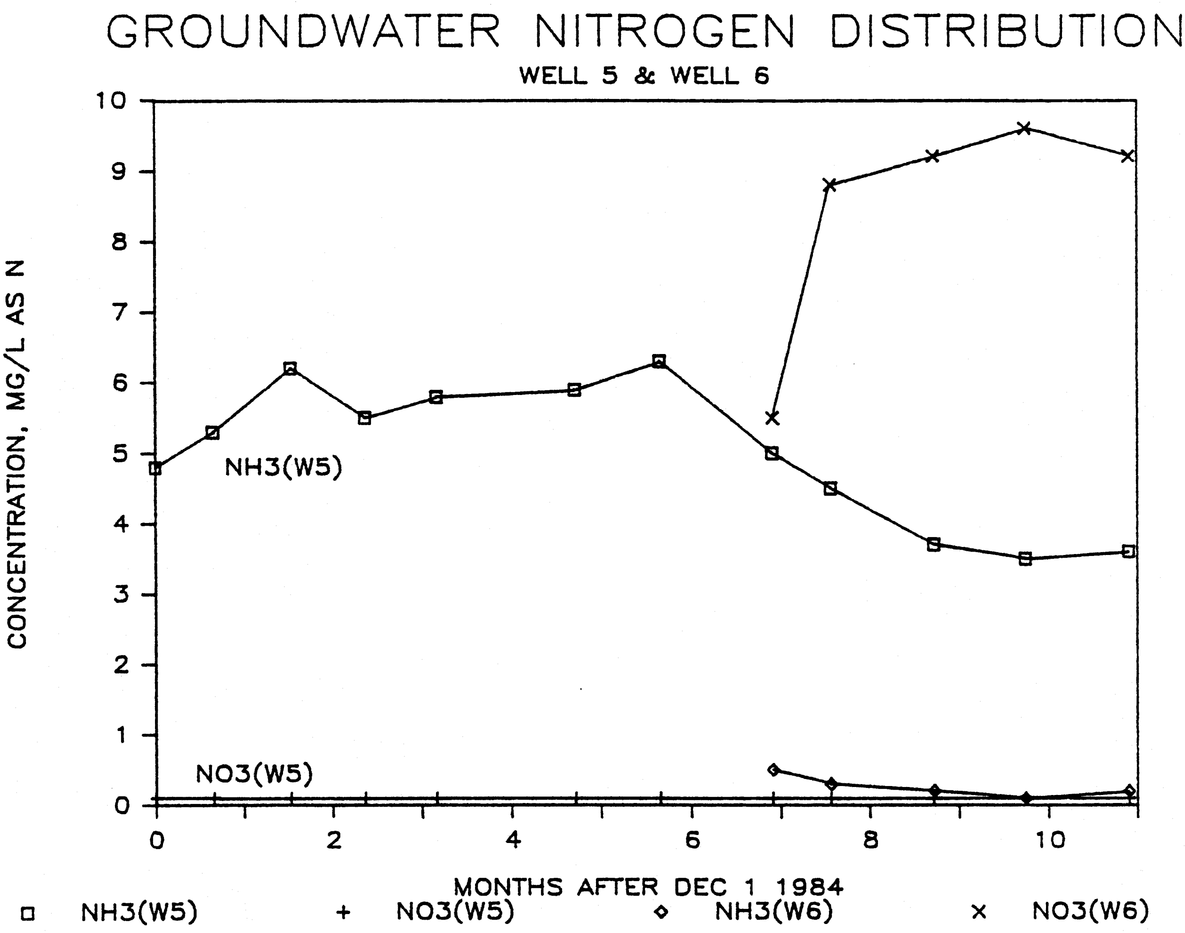

GROUNDWATER NITROGEN DISTRIBUTION
WELL 5 & WELL 6
CONCENTRATION, MG/L AS N
10
9
8
7
6
5
4
3
2
1
0
NH3(W5)
NO3(W5)
0
2
4
6
8
10
MONTHS AFTER DEC 1 1984
NH3(W5)
NO3(W5)
NH3(W6)
NO3(W6)

FIGURE 5.9

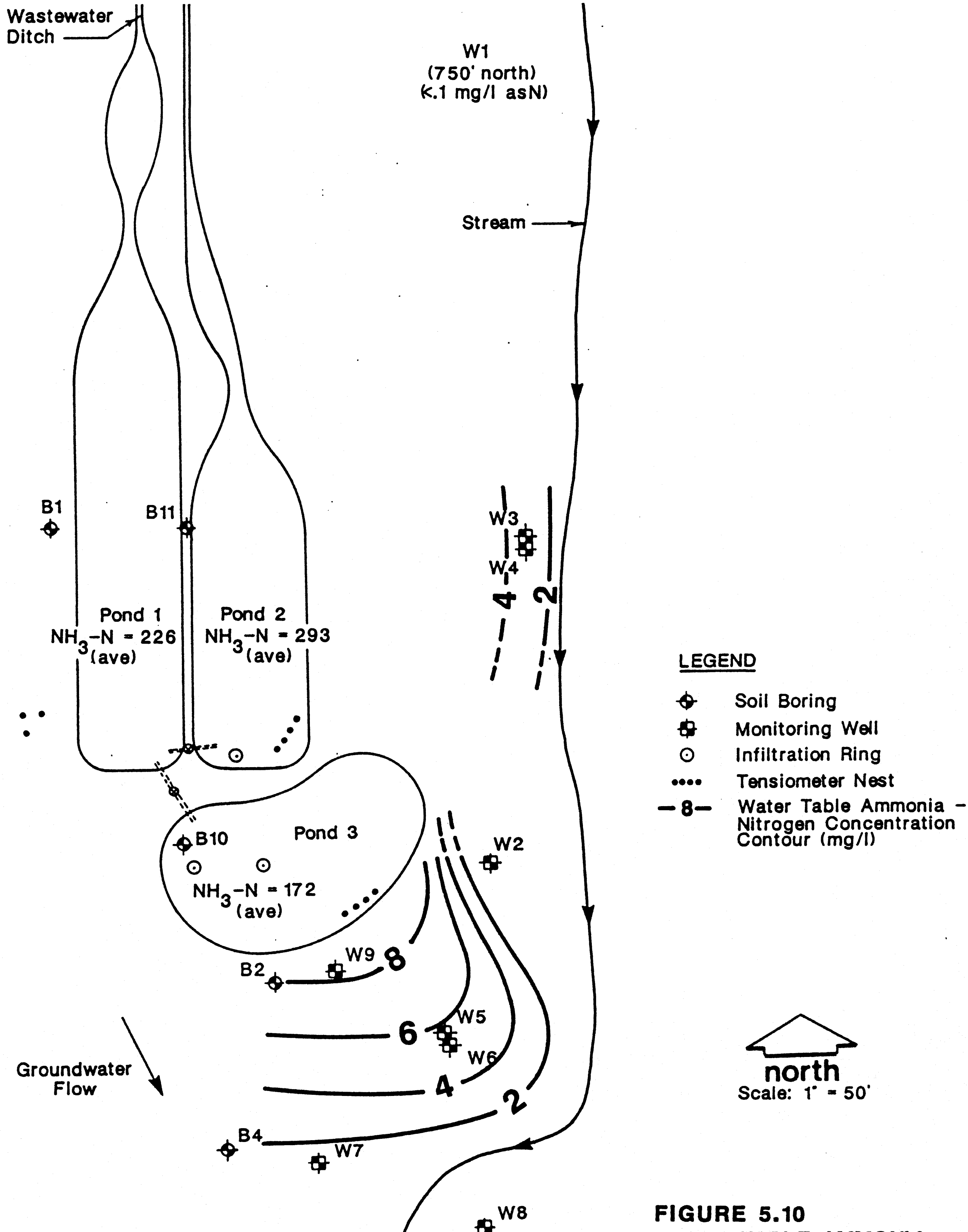


FIGURE 5.10
WATER TABLE AMMONIA - NITROGEN CONCENTRATION CONTOURS - JUNE 27, 1985
BRUNKOW CHEESE CO-OP ABSORPTION POND SITE

Evidence of the wastewater plume in the well samples was seen from elevated chloride concentrations. Travel times from pond 3 to well 9 could only be inferred from measured gradients between wells 9 and 7 due to lack of information on water table elevations below the ponds. Using an average linear velocity of 10 ft/year, the travel time between pond 3 and well 9 (a distance of 20 ft.) was approximately 2 years. Pond 3 was constructed 1 year prior to when the first samples were collected from well 9. Although the center of mass of the plume may not have reached the location of well 9 it was assumed for comparative purposes that a 20% decrease in concentrations could be attributed to dilution (by mixing mechanisms).

The TKN concentrations in well 9 were 90% less than the average concentration measured in pond 3 water. Allowing for a possible dilution of 20%, there was roughly a 85% decrease in total nitrogen in well 9. This loss may be partially due to settling in the sludge layer, adsorption in the soil profile or denitrification. The adsorptive capacity of the soil profile or denitrification. The adsorptive capacity of the soils was high (CEC's as high as 12 meq/100 gm). The amount of reduction due to denitrification alone was unknown. Wet conditions below the ponds, as depicted by the tensiometers, suggested that little oxygen was present for the necessary conversion of NH_3 to NO_3 prior to denitrification. Installation of gas probes to determine oxygen levels in the soil would have aided in this interpretation.

As the adsorptive capacity of the soils is reached, NH_3 levels in the groundwater will increase. Continued sampling of pond water and well 9 to identify changes in NH_3 concentrations would aid in determining the mechanisms responsible for the measured reductions in nitrogen concentrations.

SECTION 6 - SITE CONCLUSIONS AND RECOMMENDATIONS

Based on the field and monitoring work performed during this study, the following conclusions are drawn regarding the performance of the absorption ponds at the Evansville and Brunkow Cheese Coop sites. Site specific conclusions are summarized for each system separately. Additional conclusions which have more general application and implications of the results at both sites for design of future systems are discussed in Section 7.

6.1 Evansville Absorption Pond System

The land disposal portion of the wastewater treatment system for the City of Evansville was operated at design capacity during the period of study from October 1984 to March 1986. This condition was achieved by using only 2 of the 4 existing seepage cells since the plant was operating at half its design capacity. Under this mode of operation major conclusions of this study are summarized below.

1. Impacts from effluent discharges were detected in all monitoring wells within 250 ft. downgradient of and within 35 ft. below the infiltration basins. Wells W11 (water table) and W5D (deep) screened 450 ft. downgradient and 70 ft. below the basins respectively, showed no impact from the effluent plume.

2. The primary form (species) of nitrogen in the effluent applied to the seepage cells was seasonally dependent. NO_3 was the dominant species from May through November, while NH_3 was the dominant species from December through April. This variation was temperature dependent. From August 1985 through October 1985 the effluent contained total nitrogen concentrations below background levels, which resulted in reduced levels (below background) in wells immediately downgradient of the seepage cells. This dilution of the higher background nitrogen concentrations by the effluent plume occurred for approximately 3 months.

4. Temporal fluctuations in the nitrogen species, measured in wells immediately below the seepage cells (well W5S) and at distances of 125 ft. (well W102) and 250 ft. (well W8) downgradient, were correlated to fluctuations in the effluent concentration. Delays in arrival of NH_3 peaks coincided with the approximate required travel time to specific wells; however, the NH_3 concentrations downgradient were lower. This result is attributed primarily to transformation of NH_3 to NO_3 within the effluent plume. A maximum of 19.2 mg/l NO_3 and minimum of 6.9 mg/l NO_3 and minimum of 0.8 mg/l was measured in well W5S and; a maximum of 10.5 mg/l NO_3 and minimum of 0.8 mg/l NO_3 was measured in the effluent.

5. Conservative tracer parameters (chlorides and TDS) were detected at concentrations near those in the wastewater effluent at a distance of 250 ft. downgradient. A dilution of 10% was estimated; thus, the effects of mixing with the background water was minimal at this location.

6. Under winter operation a breakthrough of NH_3 laden percolate was measured in the groundwater immediately below the cells and correlated directly to the temperature drop of 35• F in the groundwater.

7. During the period December 1984 to November 1985 removal of other effluent wastewater constituents (within the unsaturated zone) was estimated to be: 75% for COD; more than 95% for BOD_5; and 70% for total phosphorus.

8. The resting and loading schedule of the cells was intended to be 10 days of flooded conditions and 10 days rest. This schedule was not realized due to the high infiltration rates, ranging from 4 to 9 ft/day. Thus, there was approximately a six hour total flooding time and a 42 hour rest period from April through November. During winter months surface infiltration rates slowed considerably such that the effluent remained ponded from December through March.

9. Groundwater mounding occurred on a transient basis with a maximum estimated mound height of 2 ft. at the center of either cell and a measured mound height of 1.5 ft. at the edge of the SW cell based on elevations taken at well 5S.

Based on the literature review, sampling results, and conclusions of the absorption pond study at Evansville, the following factors which limit effective nitrogen removal are:

- insufficient residence time in the unsaturated zone;
- inability to maintain a ponded surface within the basins (to create anaerobic conditions);
- cold temperatures during winter months;
- lack of sufficient carbon source during late summer (inferred from low BOD_5 concentrations in the effluent).

These factors combined are responsible for the minimal net loss of nitrogen through the soil profile, estimated for the one year period of this study. Aspects of each factor and corresponding recommendations for achieving better removal rates are discussed below.

The lack of residence time in the unsaturated zone resulted from both the high hydraulic conductivity of the soils and the large volumes of effluent applied in a given day. To enhance nitrogen removal, the infiltration rates should be decreased to promote longer ponded or wet conditions. Several methods for achieving reduced infiltration rates were discussed by Reed (1982). Mixing fine soils or other media at the surface proved unsuccessful for the cases cited. The use of water tolerant vegetation proved most successful. This along with a mixture of finer soils might be feasible at the Evansville site. It would be necessary, however, to also reduce application rates from those currently used, to maintain an even distribution of the effluent, and to cut the vegetative cover and remove clippings periodically.

If slower infiltration rates could be achieved, dosing schedules should be developed dependent on the dominant nitrogen species present in the effluent. For nitrified effluent the rest period should be as short as possible relative to the flooding period since anaerobic conditions are desired. An example would be 2 days rest and 5 days of flooding. It was observed that infiltration rates slowed if the cells were not disked. Although it was necessary to remove some weeds periodically, the seepage cells should be disked (or plowed) as little as possible.

During periods when NH_3 is the dominant form of nitrogen in the effluent, loading and resting schedules should be equal. One week rest and load cycles have been recommended by some researchers.

The removal capabilities of the system were reduced due to colder temperatures in the wastewater and its effects on the temperatures in the unsaturated zone and shallow groundwater system. To alleviate this problem the existing system may have to be modified by covering the aeration lagoons and possibly the infiltration basins. This would not be an appropriate solution unless other limiting factors discussed previously were adequately addressed.

Application of effluent with a higher carbon content would enhance the denitrification process. This condition only would be effective if the travel time to the water table also was decreased. Under slower infiltration rates the pretreatment system could be operated to produce effluent concentrations closer to the effluent limit of 50 mg/l BOD_5 by using less aeration. A carbon to nitrogen ratio closer to the minimum required for denitrification (C:N of 2:1) would then occur to a greater depth within the soil, and would promote denitrification. More research on optimal operation of aerated lagoons to produce the required carbon to nitrogen ratio is needed.

Future operation of the absorption pond system as it exists presently should involve discharging to all 4 cells. One recommended mode of operation would be to load the cells alternately in a pattern which will optimize dispersion and mixing, thus minimizing groundwater impacts, if no other action is taken in the near future. This condition can be accomplished by keeping the long axis of the cells receiving effluent perpendicular to the groundwater flow. An example would be the sequence of first loading the NE cell, then SE cell, then NW cell and then Sw cell or a similar sequence. A second mode of operation would be to use 2 cells for a single dosing and alternate between pairs of cells; this plan would distribute the load to the unsaturated zone over a greater area.

Recommendations for future investigative work at Evansville include:

- Continue sampling the wastewater and selected wells to detect changes in the impact on the local groundwater system for changes in the mode of operation of the system as discussed in the above recommendations.

- Install lysimeters than can be placed in closer contact with the soil (ceramic tip with no silica pack) and at deeper depths. The teflon lysimeters are not recommended because the silica pack was not representative of the natural soil and may have influenced the nitrogen distribution as samples passed through the pack (due to retention time, bacterial growth, filtering, etc.).

- Continuous sampling in the unsaturated soil profile twice weekly for at least two months would help characterize more clearly the nitrogen transformations that occur.

- Testing for total or soluble organic carbon and installation of gas probes to measure O_2 levels would aid in the interpretation of the nitrogen data.

- Placement of water table wells with smaller screen lengths (2 ft.) immediately below the basins would aid in evaluating effluent percolate quality.

- Additional evaluation of travel time in the unsaturated zone to determine more accurately the unsaturated flow characteristics, solute transport and the correlation between them.

- Continue measuring groundwater elevations to characterize the transient water table mound that develops below the site under various effluent loading schemes. The collective data from this study and further elevation measurements could be used to verify analytical or numerical models for predicting water table mounding due to pond recharge. In addition, this would aid in predicting more accurately the depth of penetration, spreading and dilution of the plume downgradient of the site.

6.2 Brunkow Cheese Coop Absorption Ponds

The absorption ponds served as a means of disposal of the washwater wastes from the Brunkow Cheese factory. Located a quarter mile southeast of the plant, the three cell absorption pond system received on average 3000 gal/day of wastewater. Specific to this site the following conclusions are made.

1. The highest impacts on the groundwater due to the absorption ponds are concentrated immediately downgradient (southeast of the ponds) within the shallow (surface) silt layer. At a distance of 50 ft. downgradient (well 5) the level of impact was characterized by an increase in chlorides to 320 mg/l, compared to background levels around 50 mg/l chloride. Total nitrogen concentrations were below background levels in all wells (affected by the ponds) with the exception of well 9. This result was attributed to anaerobic conditions that developed in the silt layer in areas affected by the wastewater.

2. From an analysis of conservative parameters measured in well 9 (located 20 ft. downgradient), it was concluded that a dilution of 20% was occurring over the measured concentrations in pond 3. Analysis of the nitrogen data in well 9 showed reductions of 90%, compared to total nitrogen concentrations in pond 3. With the 20% dilution, this result corresponds to an 85% decrease (removal) of the total nitrogen.

3. The groundwater in the bedrock limestone layer showed no impact from the ponds based on samples taken from wells 3 and 6. An analysis of the groundwater elevations and chemical data in these wells showed that there was no flow occurring vertically into the bedrock at the sampling locations. This result also was supported by soil borings, taken during well drilling, which showed a clay residual layer 4 inches thick above the bedrock layer.

4. The .33 acre system was hydraulically overloaded for the plant operating conditions during the study. It was necessary to pump the ponds in the spring of both 1985 and 1986. Over the period of study pond levels rose slowly. This overload was in part due to immeasurable sources, including surface runoff, overflows from a trough used to feed whey to hogs and overflows from a manure storage pit at a barn upgradient of the ponds. A water balance done on the ponds during a week when there was no wastewater inflow yielded infiltration rates of .05 in/day in pond 1, .25 in/day in pond 2, and .1 in/day in pond 3. These rates were not large enough to accommodate the inflow volumes.

5. The system could not successfully be operated on a rest and load basis by alternating discharges to ponds 2 and 3. Neither pond would drain dry during a rest period of one month.

6. The organic loading (BOD) to the absorption ponds was high, averaging 600 lb/acre/day BOD_5. This is far in excess of the required design criteria of 25 lb/acre/day contained in Wisconsin Administrative Code NR 214 for these types of systems.

7. There was no detectable impact on the stream from the absorption ponds. The effect would be hard to characterize since the stream runs through cow pastures and receives both fertilizer and manure runoff.

Based on the results and conclusions of the absorption pond study at Brunkow Cheese Coop site the following recommendations are offered:

1. Pretreatment of the wastewater at Brunkow would reduce the solids and lower the levels of BOD and nitrogen being discharged to the absorption ponds. Alternatively or in conjunction with the above, operations at the factory should be modified to reduce the organic strength of the washwater wastes. One example of this would be containment of whey spillage when trucks are loaded so that washwater drains near the loading area do not receive this spillage. Secondly, spoiled milk and substantial milk spills should be handled with the whey disposal operation. Segregation of these high strength waste sources would reduce the organic strength of the washwater wastes.

2. To increase the hydraulic capacity of the system, surface runoff should be diverted around the system by building up the dike along the wastewater ditch. The dikes along the east side of the ditch should also be built up and repaired in some areas to prevent seepage through them.

3. The possibility of spray irrigating the wastewater in summer should be considered to draw down pond levels and create additional winter storage. At minimum a controlled schedule for pumping the ponds should be maintained. Disposal of the pumped wastewater should be to crop fields, utilizing an effective means for evenly distributing the wastewater.

4. Chloride concentrations in the wastewater should be reduced to lower the amount discharged to the groundwater. It was assumed that the high chlorides resulted from use of salt in the cheese making process. The immediate source of the high chloride concentrations is unknown. Operations at the factory should be surveyed to identify potential sources and an effort made to reduce the amount of chloride going to the washwater holding tank.

5. The ponds should be operated on a rest and load basis with a rest period at least as long as the loading period.

7. Long term impacts on the groundwater are difficult to assess due to the low groundwater velocities and resulting travel times of approximately 10 ft/yr. Selected monitoring wells southeast and south of pond 3 should be monitored at least semi-annually to evaluate long term impacts.

SECTION 7 - IMPLICATIONS FOR FUTURE DESIGN

The findings and conclusions discussed in the previous sections are specific to the two systems studies. By way of comparison the two systems are very different in design and operation and in the type of wastewater handled. The parameters governing the performance of each system, however, are the same. Considering this, the following is a general discussion regarding the design of absorption ponds.

A major design problem is one of balancing hydraulic capacity with treatment capability. At each of the two sites studied only one of these objectives was met. The Brunkow site lacked hydraulic capacity but provided substantial treatment capability. The Evansville site had exceptional hydraulic capacity but little treatment capability. Further work to identify the type of soils which will meet both design objectives is needed. Design of larger systems to provide nitrogen removal may be unrealistic due to the large land areas that would be required. To address the problem of excessive nitrogen input to groundwater it may be necessary to provide nitrogen removal in the pretreatment system, prior to land disposal.

Impacts on the groundwater were documented at both absorption pond sites. Levels of critical parameters measured in the groundwater were dependent on mixing and dilution mechanisms for attenuation. Another approach to minimizing effects on the groundwater would be to use these mechanisms to reduce contaminant concentrations downgradient. In the design of future systems, configuration of the ponds and their loading sequence in relation to local groundwater flow patterns should be considered in order to enhance dilution and minimize the effects on the groundwater. This would require detailed hydrogeologic information for the site.

In the operation of absorption pond systems characteristics of the wastewater, soil properties, the groundwater, and both warm and cold climate conditions must be considered. During the winter it was important at both sites to maintain an ice cover over the ponded wastewater to keep the system operating hydraulically. This constraint offered little possibility for maintaining load and rest cycles in the winter.

For larger systems alternatives to land disposal during the winder should be considered to minimize the overall impacts on groundwater quality (specifically regarding nitrogen). This could include capabilities to discharge to surface water during the winter or storage in holding ponds over critical months when temperatures are coldest and treatment within the soil is a minimum.

The frequency and distribution of saturated and unsaturated zone monitoring required depends on the pore water and groundwater flow velocities, travel times, soil characteristics, depth to groundwater, and loading rates. At Brunkow there were no significantly monthly trends in the groundwater data. Sampling on a semi-annual basis would provide sufficient data to assess the long term impacts on groundwater. At Evansville monthly or bimonthly sampling would be needed due to the higher groundwater velocities and seasonal variability of the effluent quality.

Detailed assessment of the overall removal efficiencies within the unsaturated zone was (for the most part) limited to the use of data from shallow wells. More reliable means for sampling the soil pore water (quantity and quality) are needed. Installation of shallow wells with small screen lengths beneath the absorptions would provide valuable information regarding the quality of wastewater percolate.

Finally, it is recommended that this type of investigative work continue so that the collective results can be used to improve current design criteria for absorption pond systems. There is a need to intensify the data collection effort at selected sites to allow a more comprehensive study of systems and response to operational variations.

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APPENDIX A - EVANSVILLE DATA

A.1 Wastewater and Groundwater Data

A.2 Lysimeter Data

A.3 Soils Data

APPENDIX A.1

EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WASTEWATER EFFLUENT

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	BOD5	TEMP F
11-Dec-84	5.8	16.0	22.0	6.0	27.8	240.0	76.0	21.00	55
09-Jan-85	3.0	23.0	26.0	3.0	29.0	260.0		29.00	50
07-Feb-85	0.8	28.0	29.0	1.0	29.8	290.0		28.00	45
20-Mar-85	1.0	20.0	22.0	2.0	23.0	220.0	64.0	25.00	39
25-Apr-85	2.6	13.0	18.0	5.0	20.6	210.0	59.0	37.00	60
16-May-85	10.5	0.1	6.5	6.4	17.0	230.0		37.00	62
25-Jun-85	8.8	0.1	5.2	5.1	14.0	250.0	74.0	31.00	68
24-Jul-85	8.6	0.1	5.2	5.1	13.8	250.0	81.0	31.00	74
28-Aug-85	4.2	0.1	3.4	3.3	7.6	250.0	55.0	30.00	71
07-Oct-85	6.3	2.5	4.4	1.9	10.7	250.0	41.0	18.00	54
24-Nov-85	7.5	2.8	5.0	2.2	12.5	220.0	49.0	11.00	34
16-Jan-86	1.2	23.0	26.0	3.0	27.2	260.0	57.0	14.00	34
18-Feb-86	1.0	27.0	30.0	3.0	31.0	260.0	75.0	29.00	33
26-Mar-86	1.0	22.0	24.0	2.0	25.0	220.00	68.0	16.00	36
AVERAGE	4.5	12.7	16.2	3.5	20.6	243.6	63.5	25.5	
STD DEV	3.3	10.8	10.2	1.6	7.6	20.9	12.0	7.8	14.0

WELL 5S

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	13.7	0.1	0.6	0.5	14.3	240.0		894.28	63
09-Jan-85	17.2	0.3	0.6	0.3	17.8	210.0	17.0	893.61	
07-Feb-85	1.1	20.0	22.0	2.0	23.1	300.0	33.0	893.07	
20-Mar-85	6.6	18.0	18.0	0.0	24.6	200.0	23.0	894.73	35
25-Apr-85	14.5	9.1	10.0	0.9	24.5	210.0	12.0	894.54	39
16-May-85	19.3	4.2	4.4	0.2	23.7	220.0	13.0	894.30	40
25-Jun-85	14.1	0.6	1.0	0.4	15.1	230.0	8.0	893.55	
24-Jul-85	13.9	0.2	0.7	0.5	14.6	240.0	12.0	892.93	63
28-Aug-85	13.2	0.1	0.5	0.4	13.7	250.0	10.0	892.98	68
07-Oct-85	9.2	0.1	0.4	0.3	9.6	240.0	10.0	892.60	72
24-Nov-85	11.1	0.1	0.4	0.3	11.5	230.0	11.0	893.94	60
16-Jan-86	8.1	0.1	0.4	0.3	8.5	210.0	9.0	893.48	40
18-Feb-86	7.4	1.1	1.6	0.5	9.0	240.0	13.0	893.57	35
26-Mar-86	0.8	11.8	12.8	1.0	13.6	220.0	24.0	895.40	34
AVERAGE	10.7	4.7	5.2	0.5	16.0	231.4	15.0	893.78	
STD DEV	5.3	6.8	7.1	0.5	5.6	23.9	7.1	0.8	14

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 5M

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	GW ELEV	TEMP F
11-Dec-84	13.5	0.7	1.2	0.5	14.7	220.0		894.21	60
09-Jan-85	19.8	0.1	0.4	0.3	20.2	230.0	10.0	893.60	
07-Feb-85	2.9	0.3	1.0	0.7	3.9	290.0	21.0	892.94	
20-Mar-85	2.1	20.0	20.0	0.0	22.1	250.0	32.0	894.69	35
25-Apr-85	20.0	17.0	18.0	1.0	38.0	250.0	13.0	894.45	39
16-May-85	14.2	6.8	7.4	0.6	21.6	210.0		894.23	40
25-Jun-85	18.5	1.5	1.8	0.3	20.3	220.0	6.0	893.48	
24-Jul-85	15.7	1.0	1.4	0.4	17.1	220.0	9.0	892.98	54
28-Aug-85	16.9	0.8	1.1	0.3	18.0	240.0		892.96	58
07-Oct-85	11.5	0.5	0.8	0.3	12.3	250.0	10.0	892.56	52
24-Nov-85	10.3	0.1	0.3	0.2	10.6	230.0		893.98	53
16-Jan-86	8.8	0.1	0.4	0.3	9.2	230.0	9.0	893.65	49
18-Feb-86	2.7	0.1	0.4	0.3	3.1	240.0		893.50	46
26-Mar-86	0.3	0.4	1.0	0.6	1.3	250.0	26.0	895.33	34
AVERAGE	11.2	3.5	3.9	0.4	15.2	237.9	15.1	893.75	
STD DEV	6.7	6.4	6.4	0.2	9.3	19.3	8.5	0.7	15

WELL 5D

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	GW ELEV	TEMP F
11-Dec-84	4.5	0.1	0.2	0.1	4.7	10.0		894.22	
09-Jan-85	4.3	0.1	0.2	0.1	4.5	7.9	<5.0	893.64	
07-Feb-85	5.0	0.1	0.2	0.1	5.2	11.0		893.01	
20-Mar-85	5.2	0.7	0.8	0.1	6.0	17.0	6.0	894.72	
25-Apr-85	5.0	0.3	0.6	0.3	5.6	16.0		894.02	39
16-May-85	5.1	0.1	0.2	0.1	5.3	11.2	<5.0	894.29	40
25-Jun-85	5.0	0.1	0.2	0.1	5.2	37.0		893.58	
24-Jul-85	5.0	0.1	0.2	0.1	5.2	22.0	<5.0	893.01	54
28-Aug-85	4.7	0.1	0.2	0.1	4.9	6.6		892.97	58
07-Oct-85				0.0	0.0	5.3		892.62	52
24-Nov-85	3.9	0.1	0.6	0.5	4.5	6.1		894.00	53
16-Jan-86				0.0	0.0	8.6		893.70	49
18-Feb-86	4.8	0.1	0.2	0.1	5.0	6.2		893.53	46
26-Mar-86	6.0	0.1	0.2	0.1	6.2	10.0	<5.0	895.41	34
AVERAGE	4.9	0.2	0.3	0.1	4.5	12.5	<5.0	893.77	
STD DEV	0.5	0.2	0.2	0.1	1.9	8.2	2.4	0.7	16

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 102

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	17.9	0.1	0.4	0.3	18.3	180.0		892.32	
09-Jan-85	13.1	0.1	0.4	0.3	13.5	230.0	14.0	892.22	
07-Feb-85	11.8	0.1	0.4	0.3	12.2	250.0	11.0	891.42	
20-Mar-85	7.3	0.1	0.4	0.3	7.7	290.0	14.0	892.92	
25-Apr-85	17.3	1.5	2.0	0.5	19.3	180.0	13.0	892.95	43
16-May-85	25.4	3.9	3.9	0.0	29.3	200.0	8.0	892.75	39
25-Jun-85	18.7	4.6	5.2	0.6	23.9	230.0	9.0	892.14	
24-Jul-85	18.0	2.7	3.2	0.5	21.2	240.0	10.0	891.64	62
28-Aug-85	15.2	0.9	1.2	0.3	16.4	250.0	10.0	891.81	67
07-Oct-85	10.3	0.1	0.4	0.3	10.7	240.0	9.0	891.51	59
24-Nov-85	10.9	0.1	0.2	0.1	11.1	210.0		892.48	54
16-Jan-86	6.1	0.1	0.4	0.3	6.5	280.0	13.0	891.57	42
18-Feb-86	4.4	0.1	0.6	0.5	5.0	260.0		891.77	42
26-Mar-86	7.8	3.1	3.7	0.6	11.5	260.0	13.0	892.82	42
AVERAGE	13.2	1.3	1.6	0.4	14.8	235.7	11.3	892.17	
STD DEV	5.7	1.6	1.6	0.2	6.7	32.5	2.1	0.53	18

WELL 8

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	12.2	0.1	0.4	0.3	12.6	190.0		892.03	55
09-Jan-85	16.3	0.1	0.4	0.3	16.7	220.0	12.0	891.87	
07-Feb-85	17.2	0.1	0.5	0.4	17.7	200.0	10.0	891.12	
20-Mar-85	12.9	0.2	0.4	0.2	13.3	220.0	9.0	892.41	
25-Apr-85	7.2	0.1	0.5	0.4	7.7	230.0	12.0	892.18	48
16-May-85	6.9	0.1	0.4	0.3	7.3	210.0	6.0	892.05	48
25-Jun-85	18.7	0.1	0.4	0.3	19.1	200.0	10.0	891.51	
24-Jul-85	19.1	0.4	0.7	0.3	19.8	200.0	8.0	891.07	47
28-Aug-85	18.2	1.3	1.5	0.2	19.7	220.0	11.0	891.22	50
07-Oct-85				0.0	0.0	230.0	11.0	890.94	56
24-Nov-85	12.0	1.6	1.9	0.3	13.9	230.0			58
16-Jan-86	10.2	1.3	1.5	0.2	11.7	200.0	13.0	891.49	54
18-Feb-86	9.2	1.0	1.0	0.0	10.2	180.0		891.39	52
26-Mar-86	7.4	0.4	0.6	0.2	8.0	180.0	8.0	892.47	51
AVERAGE	12.9	0.5	0.8	0.2	12.7	207.9	10.0	891.67	
STD DEV	4.4	0.5	0.5	0.1	5.5	17.0	2.0	0.50	15

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 101 (background-shallow)

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	12.5	0.1	0.6	0.5	13.1	42.0		895.89	56
09-Jan-85	12.5	0.1	0.2	0.1	12.7	42.0		895.93	
07-Feb-85	12.7	0.1	0.2	0.1	12.9	41.0		895.22	
20-Mar-85	12.2	0.1	0.2	0.1	12.4	45.0		896.81	
25-Apr-85	11.9	0.1	0.2	0.1	12.1	42.0		896.52	50
16-May-85	12.7	0.1	0.4	0.3	13.1	44.0		896.22	51
25-Jun-85	11.9	0.1	0.2	0.1	12.1	42.0		895.75	
24-Jul-85	11.7	0.1	0.2	0.1	11.9	39.0		895.39	51
28-Aug-85	11.0	0.1	0.2	0.1	11.2	37.0		895.12	52
07-Oct-85	10.4	0.1	0.2	0.1	10.6	37.0		894.87	50
24-Nov-85	11.3	0.1	0.2	0.1	11.5	42.0	<5.0	895.16	49
16-Jan-86	10.4	0.1	0.2	0.1	10.6	35.0		895.72	50
18-Feb-86	10.8	0.1	0.2	0.1	11.0	37.0		895.79	48
26-Mar-86	10.70	0.1	0.2	0.1	10.9	43.0	<5.0	897.02	49
AVERAGE	11.7	0.1	0.2	0.1	11.9	40.6	<5.0	895.82	
STD DEV	0.8	ERR	0.1	0.1	0.9	2.9	0.0	0.63	15

WELL 9 (background-deep)

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	7.5	0.1	0.2	0.1	7.7	12.0		895.57	56
09-Jan-85	8.1	0.1	0.2	0.1	8.3	13.0	5.0	895.68	
07-Feb-85	7.7	0.1	0.2	0.1	7.9	14.0		894.87	
20-Mar-85	7.3	0.1	0.2	0.1	7.5	15.0	6.0	896.45	
25-Apr-85	7.4	0.1	0.2	0.1	7.6	14.0		896.14	50
16-May-85	7.3	0.1	0.4	0.3	7.7	14.0	6.0	895.86	50
25-Jun-85	6.9	0.1	0.2	0.1	7.1	13.0	5.0	895.39	
24-Jul-85	7.2	0.1	0.2	0.1	7.4	14.0		895.05	53
28-Aug-85	7.1	0.1	0.2	0.1	7.3	16.0	5.0	894.76	53
07-Oct-85	7.2	0.1	0.3	0.2	7.5	16.0	5.0	894.55	50
24-Nov-85	7.1	0.1	0.2	0.1	7.3	14.0		895.74	49
16-Jan-86	6.9	0.1	0.5	0.4	7.4	13.0	10.0	895.41	50
18-Feb-86	7.2	0.1	0.3	0.2	7.5	14.0		895.39	49
26-Mar-86	7.0	0.1	0.1	0.0	7.1	12.5	<5.0	896.57	50
AVERAGE	7.3	0.1	0.2	0.1	7.5	13.9	5.3	895.53	
STD DEV	0.3	ERR	0.1	0.1	0.3	1.1	2.5	0.58	15

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 4

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	16.5	0.1	0.5	0.4	17.0	190.0		893.41	54
09-Jan-85	16.8	0.1	0.3	0.2	17.1	180.0		893.13	
07-Feb-85	12.1	0.1	0.3	0.2	12.4	120.0		892.43	
20-Mar-85	8.4	0.3	0.6	0.3	9.0	53.0		894.05	
25-Apr-85	10.8	0.7	1.2	0.5	12.0	140.0		893.82	48
16-May-85	15.8	2.6	3.0	0.4	18.8	170.0		893.63	40
25-Jun-85	13.6	2.8	3.4	0.6	17.0	160.0		892.99	
24-Jul-85	11.2	1.6	2.1	0.5	13.3	130.0		892.55	53
28-Aug-85	10.6	0.8	1.1	0.3	11.7	150.0		892.53	60
07-Oct-85	8.7	0.2	0.6	0.4	9.3	180.0		892.16	64
24-Nov-85	10.4	0.1	0.4	0.3	10.8	200.0		893.47	60
16-Jan-86	8.6	0.1	0.3	0.2	8.9	160.0		892.91	50
18-Feb-86	6.6	0.4	1.0	0.6	7.6	240.0		892.90	40
26-Mar-86	0.7	9.2	9.6	0.4	10.3	250.0		894.59	47
AVERAGE	10.8	1.4	1.7	0.4	12.5	165.9		893.18	
STD DEV	4.1	2.3	2.4	0.1	3.5	47.7		0.66	17

WELL 6

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	10.7	0.1	0.6	0.5	11.3	240.0		894.41	61
09-Jan-85	10.5	0.1	0.4	0.3	10.9	250.0		894.08	
07-Feb-85	10.6	0.1	0.5	0.4	11.1	190.0		893.37	
20-Mar-85	8.8	0.1	0.4	0.3	9.2	170.0		895.04	
25-Apr-85	4.9	0.1	0.6	0.5	5.5	240.0		894.80	
16-May-85	3.2	0.1	0.6	0.5	3.8	260.0		894.56	38
25-Jun-85	4.7	0.1	0.4	0.3	5.1	150.0		893.83	
24-Jul-85	10.6	0.1	0.4	0.3	11.0	75.0		893.35	48
28-Aug-85	10.7	0.1	0.2	0.1	10.9	29.0		893.27	49
07-Oct-85	9.3	0.2	0.4	0.2	9.7	22.0		893.93	52
24-Nov-85	8.9	1.8	1.9	0.1	10.8	24.0		894.30	50
16-Jan-86	9.1	0.5	0.8	0.3	9.9	25.0		893.81	50
18-Feb-86	9.1	0.3	0.6	0.3	9.7	22.0		893.83	46
26-Mar-86	1.8	0.5	1.0	0.5	2.8	200.0		895.64	48
AVERAGE	8.1	0.3	0.6	0.3	8.7	135.5		894.16	
STD DEV	2.9	0.4	0.4	0.1	2.9	94.3		0.66	16

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 7

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84	10.5	3.0	4.0	1.0	14.5	150.0		893.55	55
09-Jan-85	13.4	2.8	5.7	2.9	19.1	200.0	12.0	892.83	
07-Feb-85	11.4	2.5	2.8	0.3	14.2	210.0	14.0	892.22	
20-Mar-85	2.8	4.7	4.8	0.1	7.6	260.0	23.0	893.70	
25-Apr-85	13.3	15.0	16.0	1.0	29.3	200.0	18.0	893.60	38
16-May-85	15.0	15.0	15.0	0.0	30.0	200.0	11.0	893.45	37
25-Jun-85	9.4	9.0	9.1	0.1	18.5	170.0	9.0	892.75	
24-Jul-85	10.1	6.7	6.9	0.2	17.0	170.0	8.0	892.18	40
28-Aug-85	8.7	4.9	5.5	0.6	14.2	130.0	12.0	892.29	58
07-Oct-85	6.6	3.0	3.4	0.4	10.0	72.0	5.0	891.98	60
24-Nov-85	8.6	1.2	1.4	0.2	10.0	180.0		893.25	59
16-Jan-86	7.2	0.7	1.0	0.3	8.2	77.0	9.0	892.56	50
18-Feb-86	5.6	0.5	0.9	0.4	6.5	88.0	9.0	892.52	43
26-Mar-86	1.9	0.1	0.6	0.5	2.5	170.0	13.0	893.87	48
AVERAGE	8.9	4.9	5.5	0.6	14.4	162.6	11.9	892.91	
STD DEV	3.7	4.7	4.7	0.7	7.7	52.6	4.6	0.62	16

WELL 10

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	GW ELEV	TEMP F
11-Dec-84									
09-Jan-85									
07-Feb-85									
20-Mar-85									
25-Apr-85									
16-May-85	26.0	11.0			26.0	200.0			
25-Jun-85	26.0	7.3	7.6	0.3	33.6	220.0	8.0	892.76	
24-Jul-85	18.2	2.5	3.0	0.5	21.2	240.0	11.0	892.18	53
28-Aug-85	15.3	0.4	0.8	0.4	16.1	250.0		892.98	66
07-Oct-85	10.1	0.1	0.5	0.4	10.6	240.0		891.95	68
24-Nov-85	10.5	0.1	0.4	0.3	10.9	230.0	10.0	893.26	59
16-Jan-86	7.8	0.1	0.4	0.3	8.2	250.0		892.53	56
18-Feb-86	9.5	0.6	1.0	0.4	10.5	200.0		892.47	41
26-Mar-86	10.3	0.7	0.8	0.1	11.1	110.0	8.0	893.83	43
AVERAGE	14.9	2.5	1.8	0.3	16.5	215.6	9.3	892.75	
STD DEV	6.7	3.7	2.3	0.1	8.2	41.4	1.3	0.57	20

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 12

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84									
09-Jan-85									
07-Feb-85									
20-Mar-85									
25-Apr-85									
16-May-85									
25-Jun-85									
24-Jul-85									
28-Aug-85									
07-Oct-85									
24-Nov-85	10.2	1.4	1.6	0.2	11.8	200.0	15.0	891.73	58
16-Jan-86	9.6	0.1	0.4	0.3	10.0	210.0	9.0	891.06	58
18-Feb-86	7.0	0.1	0.4	0.3	7.4	220.0	10.0	891.06	52
26-Mar-86	4.4	0.1	0.3	0.2	4.7	230.0	12.0	892.01	50
AVERAGE	7.8	0.4	0.7	0.3	8.5	215.0	11.5	891.47	
STD DEV	2.3	0.6	0.5	0.1	2.7	11.2	2.3	0.42	4

WELL 13

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84									
09-Jan-85									
07-Feb-85									
20-Mar-85									
25-Apr-85									
16-May-85									
25-Jun-85									
24-Jul-85									
28-Aug-85									
07-Oct-85									
24-Nov-85	10.8	1.6	2.0	0.4	12.8	220.0	7.0	892.04	57
16-Jan-86	10.2	1.4	1.6	0.2	11.8	210.0	9.0	891.70	54
18-Feb-86	9.8	0.6	0.8	0.2	10.6	210.0		891.37	54
26-Mar-86	5.9	0.2	0.5	0.3	6.4	230.0	9.0	892.45	53
AVERAGE	9.2	1.0	1.2	0.3	10.4	217.5	8.3	891.89	
STD DEV	1.9	0.6	0.6	0.1	2.4	8.3	0.9	0.40	22

APPENDIX A.1
(Continued)
EVANSVILLE ABSORPTION POND STUDY DATA
(Concentrations in mg/l)

WELL 11

Date	NO3	NH3	TKN	ORG N	TOTAL N	CL	COD	ELEV	TEMP F
11-Dec-84									
09-Jan-85									
07-Feb-85									
20-Mar-85									
25-Apr-85									
16-May-85									
25-Jun-85	8.3	0.1	0.4	0.3	8.7	17.0	5.0	891.15	
24-Jul-85	7.4	0.1	0.2	0.1	7.6	14.0	5.0		63
28-Aug-85	8.0	0.1	0.2	0.1	8.2	14.0		891.20	60
07-Oct-85									
24-Nov-85						10.0		892.01	46
16-Jan-86						14.0		891.80	38
18-Feb-86						7.7		891.45	38
26-Mar-86						4.6		892.38	38
AVERAGE	9.2	1.0	1.2	0.3	10.4	217.5	8.3	891.89	
STD DEV	1.9	0.6	0.6	0.1	2.4	8.3	0.9	0.40	22

APPENDIX A.2

LYSIMETER DATA (1985) -- EVANSVILLE
(Concentrations in mg/l)

Lysimeter 1 (2.5 ft./SW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun	17.2	0.1	0.8	18.0	250	13	-	-
25-Jun	22.0	0.1	0.4	22.4	250	-	-	-
27-Jun	26.0	0.1	0.5	26.5	250	8	-	-
29-Aug			(I.S.)					
03-Nov			(I.S.)					
08-Nov			(I.S.)					

Lysimeter 2 (5.0 ft./SW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun			(I.S.)					
25-Jun	29.0	<0.1	0.6	29.6	240	-	-	-
27-Jun	28.0	<0.1	0.8	28.8	240	19	-	-
29-Aug	14.2	<0.1	0.8	15.0	240	22	-	-
03-Nov			(I.S.)					
08-Nov	6.8	<0.1	0.4	7.2	240	11		

Lysimeter 3 (8.0 ft./SW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun			(I.S.)					
25-Jun	18.1	0.4	1.5	19.6	100	-	-	-
27-Jun	30.0	0.8	2.2	32.2	160	41	-	-
29-Aug	13.6	0.1	1	14.6	240	26	-	-
03-Nov			(I.S.)					
08-Nov			(I.S.)					

Lysimeter 4 (Ceramic Tip) (3.0 ft./SW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
29-Aug	4.4	<0.1	1	5.4	250	26	-	-
03-Nov			(I.S.)					
08-Nov	10.2	<0.1	0.8	11.0	240	20	1.8	-

APPENDIX A.2
(Cont.)
LYSIMETER DATA (1985) -- EVANSVILLE
(Concentrations in mg/l)

Lysimeter 5 (2.5 ft./NW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun			(I.S.)					
19-Jun			(I.S.)					
20-Jun	18.7	<0.1	0.5	19.2	220	6	-	7.6
02-Oct	2.0	<0.1	0.6	2.6	260	16	3.3	-
22-Oct	5.0	<0.1	0.5	5.5	250	13	-	-
28-Oct	7.0	<0.1	0.5	7.5	250	13	-	-

Lysimeter 6 (5.0 ft./NW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun			(I.S.)					
19-Jun			(I.S.)					
20-Jun	18.5	<0.1	1.4	19.9	200	-	-	-
02-Oct	6.9	<0.1	1	7.9	240	30	-	-
22-Oct	4.8	<0.1	-	4.8	190	-	-	-
28-Oct	4.5	<0.1	0.5	5.0	160	15	-	-

Lysimeter 7 (7.5 ft/NW Cell)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
18-Jun	7.6	0.1	1.0	8.6	180	32	-	-
19-Jun			(I.S.)					
20-Jun	10.7	0.2	1.0	11.7	240	20	-	7.7
02-Oct			(I.S.)					
22-Oct			(I.S.)					
28-Oct			(I.S.)					

Lysimeter 8 (2.5 ft./SE Cell - background)

Date	NO3	NH3	TKN	Total N	Cl	COD	BOD	pH
19-Jun	14.1	0.1	0.6	14.7	18	12	-	-
24-Jun	17.1	0.1	0.5	17.6	17	14	-	-
28-Jun	20.4	0.1	0.6	21.0	15	19	-	-
02-Oct	15.0	0.1	1.2	16.2	4	37	-	-
22-Oct	9.7	0.1	0.6	10.3	2.7	17	-	-
28-Oct	9.0	0.1	0.7	9.7	2.6	20	-	-

APPENDIX A.3

EVANSVILLE SOILS DATA

LOCATION & DEPTH-FT	PH	OM %	P (ppm)	K (ppm)	Ca (ppm)	Mg (ppm)	Na (ppm)	CEC meq/100g	Nitrogen (%)	#10 GRAVEL (%)	#20 VC SAND (%)	#60 M SAND (%)	#100 F SAND (%)	(P200) VF SAND/ SILT/CLAY (%)
WELL 5D														
23-25	8.2	0.1	4	18	1140	118	26	3.9	0.04	30	10.1	46.2	9.4	4.3
33-35	-	-	-	-	-	-	-	-	-	34.3	7.2	44	10.5	4
43-45	8.9	0	2	10	-	-	-	-	0.02	28.3	7.8	49.5	11.2	3.2
53-55	-	-	-	-	-	-	-	-	-	22.8	8.7	50.2	13.9	4.4
63-65	-	-	-	-	-	-	-	-	-	23.2	9.6	52.7	11.1	3.4
73-75	8.9	0	2	15	684	88	24	2.4	0.01	15.7	9.1	49.2	16.3	9.7
WELL 8														
25	8.9	0	2	12	684	88	24	2.4	0.01	52.8	8.6	29.8	6.9	1.9
33	-	-	-	-	-	-	-	-	-	42.2	7.2	39.1	9.7	1.8
43	-	-	-	-	-	-	-	-	-	26.1	11.2	50.9	9.3	2.5
55	-	-	-	-	-	-	-	-	-	22.1	8.8	51.3	14.2	3.6
WELL 9														
35	8.9	0	2	11	685	77	8	2.4	0.01	25.7	10.8	52.4	9.4	1.7
45	-	-	-	-	-	-	-	-	-	31.3	12.5	44.2	8.8	3.2
73-75	-	-	-	-	-	-	-	-	-	36.2	10.4	50.6	8.4	2.8

SOIL SAMPLES WERE TAKEN OCT-DEC 1984 & ANALYZED BY THE UW EXTENSION SOIL/PLANT ANALYSIS LAB IN DEC, 1984 FOR WELL 5D, 8 AND 9

APPENDIX A.3
(Cont.)
EVANSVILLE SOILS DATA

LOCATION & DEPTH-FT	PH	OM %	P ppm	K ppm	Ca ppm	Mg ppm	Na ppm	CEC meq/100g	Nitrogen %	SAND %	SILT %	CLAY %	MOISTURE CONTENT % (dry wt)
WELL 11													
6	-	-	-	-	-	-	-	-	-	29	60	11	-
SW CELL													
(EAST EDGE)													
1	7.7	0.1	77	65	500	150	-	1	-	96	1	3	6.5
2	7.9	0.2	68	60	400	100	-	1	-	98	0	2	10.1
3.5	8.4	0.2	56	45	2250	320	-	4	-	93	3	4	6.5
5	8.2	0.2	75	55	950	290	-	2	-	94	2	4	8.2
(ALONG PIPE)													
1	8.5	0.2	46	50	1450	250	-	3	-	95	2	3	6.4
2	8	0.2	62	50	450	140	-	1	-	97	0	3	9.8
3.5	8.7	0.2	47	45	1250	390	-	3	-	93	3	4	6.6
5	8.9	0.2	34	65	2450	480	-	5	-	87	8	5	6.0
(MIDDLE)													
4.5	9.1	0.2	38	60	1950	620	-	4	-	89	6	5	5.4
8	9.1	0.2	23	45	2450	230	-	4	-	95	2	3	6.3
12	9.1	0.1	29	45	3150	300	-	5	-	96	1	3	5.5
16	8.9	0.1	30	30	2750	190	-	5	-	96	1	3	9.8
20	8.9	0.1	45	45	2950	260	-	5	-	97	0	3	8.3

SOIL SAMPLES WERE TAKEN 11/17/85 & ANALYZED BY THE UW EXTENSION
SOIL/PLANT ANALYSIS LAB IN DEC, 1985 FOR WELL 11 AND SAMPLES FROM THE SW CELL

APPENDIX A.3 (Cont.)

EVANSVILLE SOILS DATA -- BULK DENSITY DETERMINATIONS

SAMPLE NO.	(1) MOISTURE CONTENT (dry wt. %)	(2) BULK DENSITY g/cm3	(3) POROSITY	(4) MOISTURE CONTENT (volum. %)
C2	14.8	1.6	0.39	24
C3	8.7	1.8	0.32	16
C4	8.4	1.76	0.33	15
C5	7.8	1.7	0.36	13
C7	6	1.74	0.35	10
C8	9.4	1.67	0.37	16
C11	5.3	1.68	0.37	9
C12	6	1.73	0.35	10

1) Moisure Contents were determined following ASTM D2216 Procedure
2) Bulk Densities were determined following ASTM 2937 Procedure
3) Porosity calculated assuming a specific weight of 2.65 g/cm3
4) Samples were taken from 5" to 12" below the surface of the SE cell, on 11/17/85, analyzed on 11/18/85

APPENDIX B - BRUNKOW DATA

B.1 Wastewater and Groundwater Data

B.2 Soils Data

APPENDIX B.1

BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

RAW WASTEWATER

DATE	TKN	NH3	NO3	CL	BOD5	COD	TSS	PH		ORG N	TOTAL N
20-Nov-84											
20-Dec-84	220.0	15.0	1.0	60	5900	11000		4.0		205.0	221.0
16-Jan-85	300.0	28.0	1.0	1800	7800	19000		4.8		272.0	301.0
10-Feb-85	240.0	23.0	1.0	710	9100	11000	710	4.3		217.0	241.0
19-Mar-85	300.0	17.4	1.0	920	9700	18000	2020	4.6		282.6	301.0
21-Apr-85	150.0	8.4	1.0	540	3500	6600	930	5.3		141.6	151.0
20-May-85	220.0	20.0	1.0	520	7700	10000	900	4.9		200.0	221.0
27-Jun-85	230.0	12.4	1.0	920	6100	10000	870			217.6	231.0
17-Jul-85	590.0	35.0	0.2	980	15000	24000	1920	4.1		555.0	590.2
22-Aug-85	300.0	16.0	1.0	810	11000	12000	1220			284.0	301.0
22-Sep-85	250.0	21.0	1.6	320	5400	9400	970	4.3		229.0	251.6
30-Oct-85											
AVERAGES:	280.0	19.6	1.0	758	8120	13100	1193			260.4	281.0
STD DEV:	112.4	7.3	0.3	444	3116	5127	468	0.4	ERR	106.5	112.2

POND 1

DATE	TKN	NH3	NO3	CL	BOD5	COD	TSS	PH		ORG N	TOTAL N
20-Nov-84											
20-Dec-84	290.0	150.0	1.0	720	7400	10000		4.0		140.0	291.0
16-Jan-85	340.0	170.0	1.0	830	6200	7800		4.4		170.0	341.0
10-Feb-85	440.0	210.0	1.0	940	12000	13000	370	4.2		230.0	441.0
19-Mar-85	190.0	85.0	1.0	400	4000	7600	200	4.4		105.0	191.0
21-Apr-85	400.0	150.0	1.0	770	9600	13000	242	3.6		250.0	401.0
20-May-85	540.0	220.0	1.0	850	5000	12000	360	4.3		320.0	541.0
27-Jun-85	700.0	380.0	1.0	870	7700	13000	380			320.0	701.0
17-Jul-85	700.0	430.0	1.0	940	12000	14000	380	5.2		270.0	701.0
22-Aug-85	490.0	310.0	1.0	870	18000	11000	270			180.0	491.0
22-Sep-85	320.0	200.0	1.0	600	4600	6800	175	5.1		120.0	321.0
30-Oct-85	280.0	180.0	1.0	600	4600	7400	270	5.4		100.0	281.0
AVERAGES:	426.4	225.9	1.0	763	8282	10509	294			200.5	427.4
STD DEV:	160.2	100.0	0.0	161	4106	2572	76	0.6	ERR	78.3	160.2

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

POND 2

DATE	TKN	NH3	NO3	CL	BOD5	COD	TSS	PH	ORG N	TOTAL N
20-Nov-84										
20-Dec-84	240.0	180.0	1.0	580	3400	5600		4.7	60.0	241.0
16-Jan-85	200.0	180.0	1.0	510	3320	5600		5.3	20.0	201.0
10-Feb-85	200.0	190.0	1.0	620	4300	5100	68	5.5	10.0	201.0
19-Mar-85	140.0	130.0	1.0	430	2000	3900	108	5.5	10.0	141.0
21-Apr-85	160.0	150.0	1.0	480	2200	3400	62	5.6	10.0	161.0
20-May-85										
27-Jun-85	650.0	480.0	1.0	860	6200	10000	200		170.0	651.0
17-Jul-85	650.0	550.0	1.0	900	6700	8500	104	6.8	100.0	651.0
22-Aug-85	560.0	440.0	1.0	960	4600	5300	225		120.0	561.0
22-Sep-85	380.0	340.0	1.0	800	2800	3800	90	7.4	40.0	381.0
30-Oct-85										
AVERAGES:	353.3	293.3	1.0	682	3947	5689	122		60.0	354.3
STD DEV:	200.6	152.0	0.0	188	1565	2082	59	0.9	54.6	200.6

POND 3

DATE	TKN	NH3	NO3	CL	BOD5	COD	TSS	PH	ORG N	TOTAL N
20-Nov-84										
20-Dec-84	220.0	170.0	1.0	570	2600	4700		5.6	50.0	221.0
16-Jan-85	140.0	130.0	1.0	500	2200	3900		5.1	10.0	141.0
10-Feb-85	150.0		1.0	540	3100	3700	34	6.0	150.0	151.0
19-Mar-85	88.0	86.0	1.0	300	1200	2100	20	6.2	2.0	89.0
21-Apr-85	110.0	100.0	1.0	380	1200	2200	48	7.0	10.0	111.0
20-May-85	84.0	67.0	1.0	490	770	1100	176	7.8	17.0	85.0
27-Jun-85										
17-Jul-85										
22-Aug-85	350.0	290.0	1.0	860	4400	600	244		60.0	351.0
22-Sep-85	230.0	190.0	1.0	690	1700	2400	130	7.6	40.0	231.0
30-Oct-85										
AVERAGES:	171.5	147.6	1.0	541	2146	2588	109		42.4	172.5
STD DEV:	84.8	71.2	0.0	163	1123	1323	82	0.9	45.2	84.8

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

WELL 1

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84	0.6	0.1	12.9	50.0	7.1	13.0	626.0	7.7	1015.33	0.5	13.5
20-Dec-84	0.4	0.1	9.6	59.0	3.0	9.0	614.0	6.4	1016.16	0.3	10.0
16-Jan-85	0.5	0.1	12.8	54.0	3.0	9.0	602.0	7.0	1015.43	0.4	13.3
10-Feb-85	0.4	0.1	7.0	69.0	3.0	8.0	672.0	6.9	1014.54	0.3	7.4
19-Mar-85	0.4	0.1	11.8	40.0	3.0	12.0	494.0	6.9	1016.61	0.3	12.2
21-Apr-85	0.4	0.4	15.7	36.0	3.0	8.0	512.0	6.9	1015.94	0.0	16.1
20-May-85	0.4	0.1	13.4	47.0	3.7	9.0	612.0	6.9	1015.34	0.3	13.8
27-Jun-85	0.4	0.1	7.1	69.0	3.0	5.0			1014.47	0.3	7.5
17-Jul-85											
22-Aug-85											
22-Sep-85	0.4	0.1	4.6	63.0	3.0	10.0	602.0	6.9	1014.49	0.3	5.0
30-Oct-85	0.4	0.1	9.9	56.0	3.0	7.0	612.0	7.1	1015.64	0.3	10.3
AVERAGES:	0.4	0.1	9.5	49.4	3.2	7.5	534.6			0.3	10.9
STD DEV:	0.1	0.1	4.3	18.6	1.5	3.9	185.1	2.1	0.7	0.1	3.3

WELL 2

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84	1.6	0.7	0.1			45.0			995.77	0.9	1.7
20-Dec-84	1.0	0.5	0.1	280.0	4.0	20.0	60.0	7.2	996.16	0.5	1.1
16-Jan-85	0.8	0.4	0.1	260.0	3.7	18.0	720.0	8.0	995.72	0.4	0.9
10-Feb-85	0.8	0.3	0.1	210.0	7.4	11.0	640.0	7.7	995.16	0.5	0.9
19-Mar-85	0.9	0.5	0.2	280.0	3.0	18.0	890.0	7.5	996.52	0.4	1.1
21-Apr-85	1.4	0.8	0.1	320.0	3.7	19.0	946.0	7.4	995.77	0.6	1.5
20-May-85	1.4	0.8	0.1	270.0	3.4	16.0	862.0	7.5	995.35	0.6	1.5
27-Jun-85	1.2	0.7	0.1	160.0	4.6	16.0			994.70	0.5	1.3
17-Jul-85	1.4	1.1	0.3	99.0	4.3	10.0	572.0		994.86	0.3	1.7
22-Aug-85	0.6	0.4	2.4	52.0	3.0	5.0	536.0	7.2	994.27	0.2	3.0
22-Sep-85	0.6	0.3	2.3	48.0	3.0	11.0	508.0	7.4	994.74	0.3	2.9
30-Oct-85	0.3	0.1	1.1	64.0	3.0	8.0	540.0	7.4	995.80	0.2	1.4
AVERAGES:	1.0	0.6	0.6	185.7	3.9	16.4	627.4			0.5	1.6
STD DEV:	0.4	0.3	0.8	99.5	1.2	9.8	242.8	0.2	0.6	0.2	0.7

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

WELL 3

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84	1.2	0.8	2.3	87.0	6.8	15.0	596.0	7.9	998.00	0.4	3.5
20-Dec-84	0.8	0.5	7.5	60.0	3.0	6.0	602.0	7.0	998.53	0.3	8.3
16-Jan-85	0.4	0.2	8.3	50.0	3.0	9.0	532.0	7.5	998.15	0.2	8.7
10-Feb-85	0.8	0.3	4.8	88.0	3.4	7.0	618.0	7.2	997.92	0.5	5.6
19-Mar-85	0.3	0.1	9.5	41.0	3.0	5.0	514.0	7.3	998.20	0.2	9.8
21-Apr-85	0.4	0.1	10.0	34.0	3.0	5.0	502.0	7.3	998.04	0.3	10.4
20-May-85	0.4	0.1	9.3	38.0	4.0	5.0	532.0	7.3	998.11	0.3	9.7
27-Jun-85	0.5	0.1	9.9	38.0	3.0	5.0			997.55	0.4	10.4
17-Jul-85	0.2	0.1	9.7	39.0	3.0	5.0	496.0		997.32	0.1	9.9
22-Aug-85	0.4	0.1	8.6	41.0	3.0	5.0	552.0	7.2	997.24	0.3	9.0
22-Sep-85	0.4	0.1	8.3	46.0	3.1	12.0	550.0	7.4	997.71	0.3	8.7
30-Oct-85	1.2	0.8	2.6	100.0	4.6	11.0	614.0	7.2	998.25	0.4	3.8
AVERAGES:	0.6	0.3	7.6	55.2	3.6	7.5	509.0			0.3	8.2
STD DEV:	0.3	0.3	2.7	22.2	1.1	3.3	158.9	0.2	0.4	0.1	2.4

WELL 4

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84	2.7	2.1	0.1	190.0	28.0	51.0	798.0	8.3	998.15	0.6	2.8
20-Dec-84	2.8	2.5	0.1	190.0	6.1	16.0	702.0	6.9	998.51	0.3	2.9
16-Jan-85	2.6	2.2	0.5	220.0	4.0	19.0	758.0	7.4	998.08	0.4	3.1
10-Feb-85	3.0	2.5	0.4	240.0	10.0	14.0	852.0	7.3	997.82	0.5	3.4
19-Mar-85	3.6	3.2	0.3	290.0	6.8	24.0	992.0	7.3	998.12	0.4	3.9
21-Apr-85	3.8	3.2	0.1	270.0	19.0	23.0	938.0	7.3	997.92	0.6	3.9
20-May-85	3.7	3.4	0.1	180.0	3.0	15.0	756.0	7.4	997.78	0.3	3.8
27-Jun-85	3.2	2.9	0.1	81.0	4.3				997.55	0.3	3.3
17-Jul-85	3.0	3.0	0.1	66.0	3.0		524.0		997.22	0.0	3.1
22-Aug-85	3.5	3.1	0.1	48.0	3.0	5.0	484.0	7.3	997.12	0.4	3.6
22-Sep-85	2.7	2.1	0.1	50.0	3.0	12.0	464.0	7.3	997.58	0.6	2.8
30-Oct-85	3.7	3.1	0.1	190.0	9.2	11.0	768.0	7.2	998.16	0.6	3.8
AVERAGES:	3.2	2.8	0.2	167.9	8.3	19.0	669.7			0.4	3.4
STD DEV:	0.4	0.4	0.1	82.2	7.4	11.9	257.9	0.3	0.4	0.2	0.4

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

WELL 5

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84	5.7	4.8	0.1	370.0		35.0	1091.0	8.0	995.00	0.9	5.8
20-Dec-84	5.9	5.3	0.1	370.0	12.0	31.0	1100.0	6.8	995.14	0.6	6.0
16-Jan-85	6.5	6.2	0.1	410.0	3.0	31.0	1240.0	7.2	994.91	0.3	6.6
10-Feb-85	6.2	5.5	0.1	400.0	9.8	24.0	1210.0	7.2	994.45	0.7	6.3
19-Mar-85	6.2	5.8	0.1	400.0	5.2	29.0	1270.0	7.3	995.70	0.4	6.3
21-Apr-85	6.2	5.9	0.1	410.0	9.5	23.0	1310.0	7.2	995.10	0.3	6.3
20-May-85	6.8	6.3	0.1	410.0	7.0	26.0	1250.0	7.3	993.75	0.5	6.9
27-Jun-85	5.5	5.0	0.1	320.0	9.8	22.0	832.0		993.69	0.5	5.6
17-Jul-85	5.0	4.5	0.1	250.0	4.3	15.0	724.0		993.42	0.5	5.1
22-Aug-85	4.1	3.7	0.1	170.0	3.7	11.0	694.0	7.2	993.60	0.4	4.2
22-Sep-85	4.3	3.5	0.1	160.0	6.4	13.0	736.0	7.3	993.66	0.8	4.4
30-Oct-85	4.0	3.6	0.1	190.0	4.9	15.0	768.0	7.3	994.65	0.4	4.1
AVERAGES:	5.5	5.0	0.1	321.7	6.3	22.9	1018.8			0.5	5.6
STD DEV:	0.9	1.0		96.6	3.3	7.6	236.1	0.3	0.7	0.2	0.9

WELL 6

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84											
20-Dec-84											
16-Jan-85											
10-Feb-85											
19-Mar-85											
21-Apr-85											
20-May-85											
07-Jun-85	1.0	0.5	5.5	120.0	2.2	11.0	634.0	7.4		0.5	6.5
27-Jun-85	0.5	0.3	8.8	67.0	3.7	5.0			993.24	0.2	9.3
17-Jul-85	0.4	0.2	9.2	59.0	3.0	6.0	534.0		993.04	0.2	9.6
22-Aug-85	0.4	0.1	9.6	52.0	3.0	5.0	546.0	7.1	992.99	0.3	10.0
22-Sep-85	0.4	0.2	9.2	54.0	3.0	5.0	536.0	7.2	993.39	0.2	9.6
30-Oct-85	0.2	0.1	5.4	50.0	3.0	5.0	522.0	7.2	994.16	0.1	5.6
AVERAGES:	0.5	0.2	8.0	67.0	3.0	6.2	554.4	7.2	993.36	0.3	8.4
STD DEV:	0.2	0.1	1.8	24.3	0.4	2.2	40.5	0.1	0.4	0.1	1.7

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

WELL 7

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84											
20-Dec-84											
16-Jan-85											
10-Feb-85											
19-Mar-85											
21-Apr-85											
20-May-85											
07-Jun-85	1.4	1.0	0.1	260.0	6.8	14.0	792.0	7.5		0.4	1.5
27-Jun-85	1.5	0.9	0.1	280.0	4.9	16.0			992.93	0.6	1.6
17-Jul-85	2.0	1.2	0.1	340.0	6.1	20.0	950.0		992.66	0.8	2.1
22-Aug-85	2.0	1.4	0.1	340.0	3.0	18.0	1060.0	7.2	992.62	0.6	2.1
22-Sep-85	2.2	1.6	0.1	290.0	3.1	15.0	594.0	7.2	992.98	0.6	2.3
30-Oct-85	2.4	1.9	0.1	280.0	4.3	20.0	940.0	7.1	993.73	0.5	2.5
AVERAGES:	1.9	1.3	0.1	298.3	4.7	17.2	867.2		992.98	0.6	2.0
STD DEV:	0.4	0.3		30.8	1.4	2.3	161.0	0.2	0.4	0.1	0.4

WELL 8

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84											
20-Dec-84											
16-Jan-85											
10-Feb-85											
19-Mar-85											
21-Apr-85											
20-May-85											
07-Jun-85	0.4	0.1	9.1	240.0	3.7	14.0	506.0	7.6		0.3	9.5
27-Jun-85	0.2	0.1	10.4	21.0	3.7	5.0			992.73	0.1	10.6
17-Jul-85	0.2	0.1	9.7	24.0	3.0	5.0	482.0		992.59	0.1	9.9
22-Aug-85	0.2	0.1	8.1	30.0	3.0	5.0	532.0	7.0	992.57	0.1	8.3
22-Sep-85	0.2	0.1	7.8	31.0	3.0	5.0	508.0	7.1	992.88	0.1	8.0
30-Oct-85	0.2	0.1	7.7	26.0	3.1	5.0	496.0	7.1	993.25	0.1	7.9
AVERAGES:	0.2	0.1	8.8	62.0	3.3	6.5	504.8		992.80	0.1	9.0
STD DEV:	0.1		1.0	79.7	0.3	3.4	16.4	0.2	0.2	0.1	1.0

APPENDIX B.1
(Continued)
BRUNKOW ABSORPTION POND STUDY
(Concentrations in mg/l)

WELL 9

DATE	TKN	NH3	NO3	CL	BOD5	COD	TDS	PH	ELEV	ORG N	TOTAL N
20-Nov-84											
20-Dec-84											
16-Jan-85											
10-Feb-85											
19-Mar-85											
21-Apr-85											
20-May-85											
07-Jun-85	9.6	9.2	0.1	430.0	4.6	29.0	1270.0	7.1	994.14	0.4	9.7
27-Jun-85	9.5	8.4	0.1	420.0	16.0	38.0				1.1	9.6
17-Jul-85	22.0	18.0	0.1	460.0	15.0	54.0	1330.0		993.92	4.0	22.1
22-Aug-85	14.0	13.0	1.0	440.0	8.2	30.0	1230.0	7.0	993.73	1.0	15.0
22-Sep-85	15.0	13.0	1.0	430.0	15.0	38.0	1210.0	7.1	994.26	2.0	16.0
30-Oct-85	13.0	11.0	16.4	400.0	18.0	44.0	1238.0	6.8	995.63	2.0	29.4
AVERAGES:	13.9	12.1	3.1	430.0	12.8	38.8	1255.6		994.34	1.8	17.0
STD DEV:	4.2	3.2	6.0	18.3	4.7	8.5	41.9	0.1	0.7	1.2	7.0

APPENDIX B.2

BRUNKOW SOILS DATA

LOCATION & DEPTH-FT	PH	OM %	P	K	Ca	Mg	Na	CEC	Nitrogen	SAND	SILT	CLAY
			ppm					meq/100g		%		
BORING #1												
TOPSOIL	6.8	3.5	14	88	2180	695	21	8.9	0.2	10	64	26
3	5.9	0.7	43	156	1940	728	33	8.3	0.04	32	40	28
5	7.4	0.5	12	131	1935	820	28	8.6	0.03	16	54	30
5.5-6.5	8.3	0.3	9	91	1310	582	24	5.9	0.01	34	50	16
7-8	8.6	0.2	10	56	980	440	15	4.4	0.01	56	36	8
11.5-12	8.8	0.3	7	47	1400	680	43	6.5	0.03	40	56	4
WELL 2												
1-2	7.3	4.4	24	74	2670	844	35	10.9	0.18	16	62	24
4.5	7.5	2.9	21	128	2850	1100	135	12.3	0.13	14	62	24
5.5	8.4	0.4	19	75	1065	455	51	4.7	0.01	40	52	8
8.5-9	8.9	0.4	12	107	1425	715	35	6.6	0.01	54	30	6

SOIL SAMPLES WERE ANALYZED BY THE UW EXTENSION
SOIL/PLANT ANALYSIS LAB IN DEC, 1984

APPENDIX C - EVANSVILLE INSTRUMENT INSTALLATION DETAILS AND CALCULATIONS

APPENDIX C.1

GROUNDWATER MONITORING WELL INSTALLATION SPECIFICATIONS
EVANSVILLE RAPID INFILTRATION SYSTEM

WELL	DATE INSTALLED	TOP OF CASING MSL-ft	APPROX SCREEN. DEPTH-ft	SCREEN LENGTH-ft	HYDRAULIC COND. ft/day
9	12/5/84	925.97	46.5	2.5	194
101*	11/78	920.87	29	5.0	-
4	10/15/84	912.28	41.1	2.5	85
5S	10/15/84	914.38	23.3	5.0	53
5M	10/15/84	915.00	44.6	2.5	92
5D	10/15/84	914.66	70.1	2.5	124
6	10/16/84	914.85	40.4	2.5	77
102*	11/82	910.15	22	5.0	-
10	5/21/85	911.76	24.5	5.0	90
7	10/16/84	911.10	38.9	2.5	131
8	12/4/84	905.47	19.5	2.5	46
11	5/21/85	895.05	7.5	5.0	-
12	10/17/85	904.06	20.0	5.0	-
13	10/17/85	904.60	17.0	5.0	-

*City monitoring wells installed by Donohue and Associates

APPENDIX C.1 (continued)
EVANSVILLE RI SYSTEM - SLUG TEST RESULTS

Slug tests at the Evansville site were performed on July 2, 1985. The slug test method used was to drop a solid cylindrical slug four feet in length and 1 1/4" in diameter into the well. As the slug was dropped, the rise and corresponding decay in the water level was recorded with a pressure transducer and chart readout until an equilibrium point was reached. The slug was then removed quickly and the drop and subsequent rise in the water table was also measured. (This is similar to a "baildown" test.) Theoretically the two falling and rising head tests should result in the same calculated hydraulic conductivity. The pressure transducer and chart recorder was used because well recovery was on the order of 2 to 3 seconds and manual measurement of recovery levels was not possible.

The method that was used to calculate hydraulic conductivity is based on potential flow theory as developed by Hvorslev (ref. Freeze & Cherry, p. 341).

The basic equation developed by Hvorslev:

$$K= \frac{r^2 \ln\ (L/R)}{2L\ T_o} \quad \text{for} \quad \frac{L}{R} > 8$$

Where L = Screen Length
R = Radius of Screen
r = Radius of Well
T_o = Basic time lag; i.e., time that would be required for complete equilization of the head difference if the original rate of inflow was maintained., i.e., To = V/q.

This method assumes unconfined conditions where there are no boundary effects. This includes: infinite areal extent of the aquifer, no effect from bedrock, and that the screen is located below the water table surface (see Freeze & Cherry for further discussion). The limitations to slug tests are that they only apply to the region near the well and that results may vary by a factor of 10 at a given location.

Results for Wells 5S, 5M, 5D, 4, 6, 7, 10, 8 and 9 are listed on the following page. Wells 102 and 101 were not tested since these wells were not installed by DNR and installation techniques and well development significantly affect slug test results. Well 11 is screened in silt material in the low area south of the site and was not tested.

APPENDIX C.1 (continued)
EVANSVILLE RI SYSTEM - SLUG TEST RESULTS

		Falling Head		Rising Head			
Well	Screen Length (ft)	To (Sec)	K ft/sec (cm/sec)	To (Sec)	K ft/sec (cm/sec)	Ave K ft/sec (cm/sec)	ft/day
5S	5.0	4.65	6.11x10^{-4}ft/s (0.019 cm/sec)	---	---	6.11x10^{-4}ft/s (.0186 cm/sec)	53
5M	2.5	4.02	1.175x10^{-3}ft/s (.0358 cm/sec)	4.96	9.517x10^{-4}ft/s (.029 cm/sec)	1.06x10^{-3}ft/s (.032 cm/sec)	92
5D	2.5	3.19	1.482x10^{-3}ft/s (.045 cm/sec)	3.36	1.405x10_{-3}ft/s (.043 cm/sec)	1.44x10^{-3}ft/s (.044 cm/sec)	124
10	5.0	2.72	1.045x10^{-3}ft/s (.0318 cm/sec)	---	---	1.045x10^{-3}ft/s (.032 cm/sec)	90
7	2.5	3.12	1.513x10^{-3}ft/s (.046 cm/sec)	---	---	1.513x10^{-3}ft/s (.046 cm/sec)	131
4	2.5	4.78	9.876x10^{-4}ft/s .0301 cm/sec)	4.77	9.897x10^{-4}ft/s (.0297 cm/sec)	9.886x10^{-4}ft/s (.03 cm/sec)	85
6	2.5	5.9	8x10^{-4}ft/s (.0244 cm/sec)	4.84	9.75x10^{-4}ft/s (.0297 cm/sec)	8.88x10^{-4}ft/s (.027 cm/sec)	77
8	2.5	8.87	5.32x10^{-4}ft/s (0.016 cm/sec)	---	---	5.32x10^{-4}ft/s (.016 cm/sec)	46
9	2.5	2.01	2.35x10^{-3} (.0716 cm/sec)	2.21	2.14x10^{-3}ft/s (.065 cm/sec)	2.25x10^{-3}ft/s (.068 cm/sec)	194

A computer program was used to calculate the hydraulic conductivity for each well. The program requires the time/drawdown data obtained in the field. 10 to 20 points were selected from the curves. The program then plots time vs. h/h_o (in semilog space) and fits a line through the data using linear regression. T_o is determined where h/h_o = .37. K is then calculated from the equation on the previous page. (This .37 value was determined empirically by Hvorslev). For shallow wells 5S, 10 and 8 distilled water was poured into the well as a slug so that the transducer would not be damaged by dropping the slug and the water level would not drop below the top of the screen.

APPENDIX C.2

WATER TABLE MOUND HEIGHT ANALYSIS
EVANSVILLE RAPID INFILTRATION SYSTEM

The method developed by Hantush (1967) was used to predict the rise and decay of a water table mound at the Evansville site which resulted from a typical wastewater loading to the infiltration basins. Steps for this procedure are outlined below.

General Equation (developed by Hantush, 1967 and presented by K. Bradbury, Small Scale Waste Management Short Course, February, 1984).

$$
\begin{aligned}
h_{x,y,t} - H = \frac{\nu t}{4f} \{ & F\,[(W/2 + X)\,n,\ (L/2 + y)\,n] \\
+ & F\,[(W/2 + X)\,n,\ (L/2 - y)\,n] \\
+ & F\,[(W/2 - X)\,n,\ (L/2 + y)\,n] \\
+ & F\,[(W/2 - X)\,n,\ (L/2 - y)\,n]\}
\end{aligned}
$$

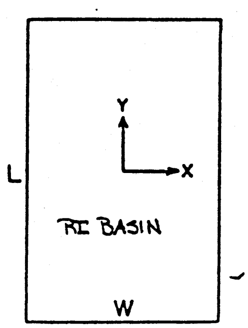
Where (for $h_{x,y,t} < .5H$),

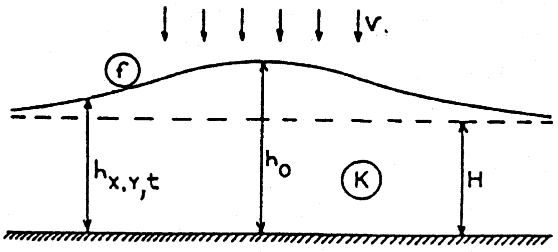
$h_{x,y,t}$ = height of water table above impermeable layer [L]

H = original water table height [L]
ν = arrival rate of water from the infiltration [L/T] basin (at the water table)
t = time since start of recharge [T]
tp = time to peak of water table rise [T]
f = fillable porosity (specific yield)
L = length of basin (in y directon) [L]
W = width of basin (in x direction) [L]
$n = (4tT/f)^{-1/2}$
$T = (K h_{x,y,t})$ $[L^2/T]$
$F(\alpha,\beta) = \int' \mathrm{erf}\,(\alpha\tau^{-1/2})\ \mathrm{erf}\,(\beta\tau^{-1/2})\, d\tau$

and $\alpha = (W/2 \pm X)\,n$
$\beta = (L/2 \pm y)\,n$

(tabulated by Hantush, 1967, Table 8)





Definition of Variables:

L = 320 ft. f = .25
W = 215 ft. K = 100 ft/day
H = 100 ft.

Assume ν (infiltration rate at the water table):

$\nu = (V/t_i) * 1/\text{Area}$ (L/T)

Where V = Volume of Wastewater applied [L^3] for a given loading

t_i = Infiltration time at the water table for volume V, of recharge [(T)]

area = L x W = 68,800 FT^2

Calculation of water table mound height at the location of Well 5S, (X=120') for conditions on August 19, 1985 (selected as a typical loading).

<u>Procedure</u>

1. Determine ν,

   V = 660,000 gallons/7.48 gal/ft^3 = 88,235 ft

   t_i = time over which infiltration is distributed at the water table.

2. Determine n(t)

   $n = (4t\ T/f)^{-1/2}$

   Where $T = KH_{x,y,t} = KH$ initially

   Since H chances with mound height, solve iteratively substituting,

$$T = \frac{K(H+h_{xyt})}{2}$$

$$n = \left[\frac{4t(100\ ft/day)(100\ ft)}{.25}\right]^{-1/2}$$

3. Determine $\Sigma F(\alpha,\beta)$ @ X = 120', y = 0' (Well 5S location)

$$\Sigma F(\alpha,\beta) = 2 * F[(215/2 + 120)n,\ (320/2 + 0)n]$$
$$+2 * F[(215/2 - 120)n,\ (320/2 + 0)n$$

4. Water table rise. $(t < t_p)$

Determine $h_{x,y,t} - H$

$$h_{x,y,t} - H = \frac{\nu t}{4f}\, \Sigma F(\alpha,\beta)$$

$$h_{x,y,t} - H = \frac{\nu\ (t)}{4\ (.25)}\ \Sigma\ F\ (\alpha,\beta)$$

5. Water table decay. $(t > t_p)$

Use principal of super position in time, to predict the decay of the mound.

Let $t' = t - t_p$

Determine $(h_{x,y,t} - H) - (h_{x,y,t'} - H)$

<u>Results</u>

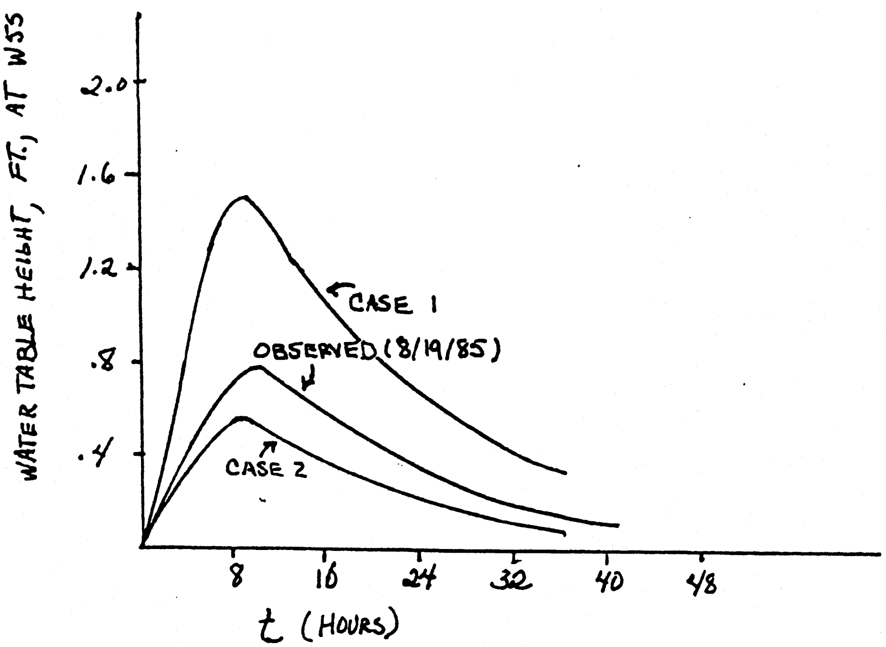
Assume t_p = 8 hours (based on observed data)

<u>Case 1</u>:

For t_i = 9 hours
ν = 3.4 ft/day

<u>Case 2</u>:

For t_i = 24 hours
ν = 1.28 ft/day



APPENDIX C.2 (continued)

		Case 1		Case 2	
t	$\sigma F(\alpha,\beta)$	Rise	Decay	Rise	Decay
1	1.4036	.19	--	0.075	--
2	1.5644	.44	--	0.17	--
3	1.5588	.66	--	0.25	--
4	1.5518	.90	--	0.33	--
5	1.5262	1.08	--	0.41	--
6	1.4818	1.26	--	0.47	--
7	1.4260	1.41	--	0.53	--
8	1.3692	1.55	--	0.58	--
9	1.330	1.70	1.51	0.63	0.56
10	1.314	1.86	1.42	0.70	0.53
12	1.246	2.11	1.21	0.80	0.47
16	1.110	2.52	0.97	0.95	0.37
20	1.0315	2.92	0.81	1.10	0.30
24	0.926	3.15	0.63	1.18	0.23
28	0.8582	3.40	0.48	1.28	0.18
32	0.804	3.64	--	--	--
36	0.748	3.81	0.41	1.44	0.16

Note: Rise = $H_{x,y,t} - H$

Decay = $(h_{x,y,t} - H) = H_{x,y,t'} - H)$

APPENDIX C.3
EVANSVILLE - UNSATURATED FLOW CALCULATIONS

Governing Equation: (for unit area finite depth of soil)

$$\frac{dV}{dt} = q_i - q_o$$

Where q_i = infiltration at the surface = 4.4 ft day (measured)
(assumed constant) (.00306 ft/min)

q_o = outflow from volume of soil

$$= - K(\Theta) \frac{d(h+Z)}{dz} = - K(\Theta) \left(\frac{dh}{dz} + 1\right)$$

h = tension in ft.
z = depth of soil (ft.), positive downward
$K(\Theta)$ = hydraulic conductivity as a function of moisture content, Θ

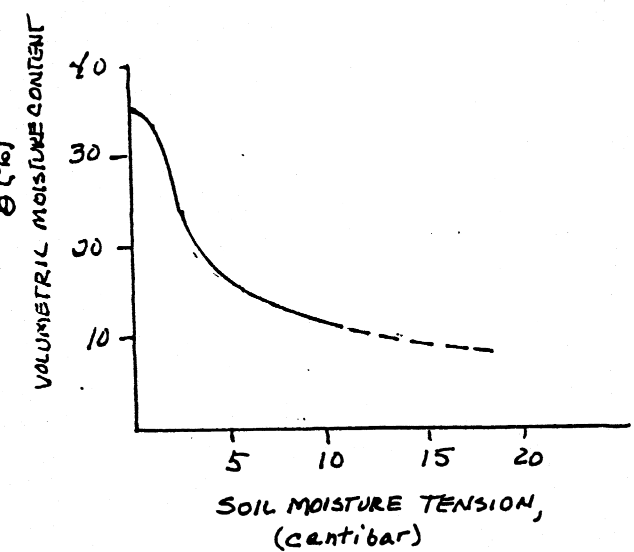
Field measurements and related data: (from 7/19/85)
At Z=2 ft., the following results apply.

Time[1] (min)	Soil Moisture Tension h (ft)	Θ	$\frac{dh}{dz}$	$K(\Theta)$ (ft/day)	$\frac{dV}{dt}$ (ft^3/min)
15	2.04	.15	1.13	---	---
19.5	1.7	.155	.41	13	.016
20.5	1.36	.165	.54	22	.025
21.5	1.02	.18	.54	25	.03
23.5	0.34	.20	.54	---	---

1 Time from start of infiltration

Procedure:

1) Determine Θ for measured tensions from previous developed curve:



APPENDIX C.3 (continued)
EVANSVILLE - UNSATURATED FLOW CALCULATIONS

2) Determine dV/dt from $V = \Theta L$, where L is the length of unit area volume of soil.

for this case L = z = 2 ft.

$\frac{dV}{dt}$ was determined from the slope at the curve ΘL vs t for the data listed previously

3) Determine dh/dz from measured tension gradients in the field. At a depth of 2 ft. it was assumed that the tension gradient was the differential between corresponding measurements at 1 ft and 3.5 ft. (over the distance), Δz=2.5 ft.

$$\left(\frac{dh}{dz}\right)_{z=2} = \frac{h_{z=3.5} - h_{z=1}}{\Delta z} \quad \text{(at any time, t)}$$

4) Calculate K(Θ) at each time, t for measured values of Θ

Example calculation:

@ t = 19.5 min, $\theta = 0.155$, $\frac{dV}{dt} = 0.016$

$$\frac{dh}{dz} = .41$$

$$q_o = -K(\Theta)\,(0.41 + 1) = -K(\Theta)\,(1.41)$$

$$\frac{dV}{dt} = q_i + K(\Theta)\,(1.41)$$

for q_i = 4.4 ft/day = .00306 ft/min

$$\frac{dV}{dt} - q_i = .016 - .00306 = K(\Theta)\ 1.41$$

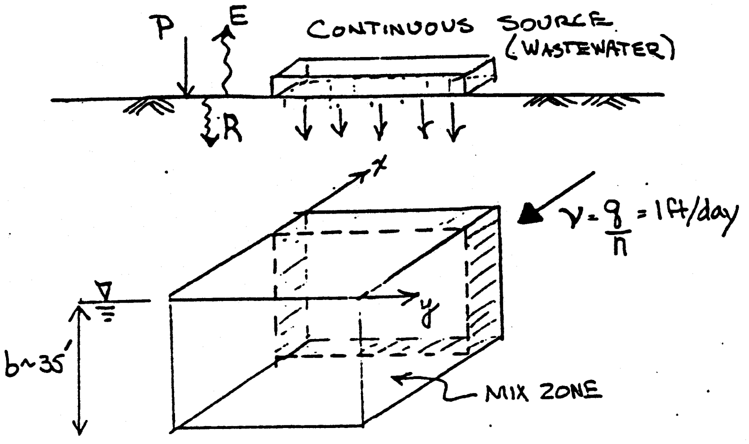
$$K(\Theta) = 9.2 \times 10^{-3} \text{ ft/min}$$
$$= 13 \text{ ft/day for } \Theta = 15.5\%$$

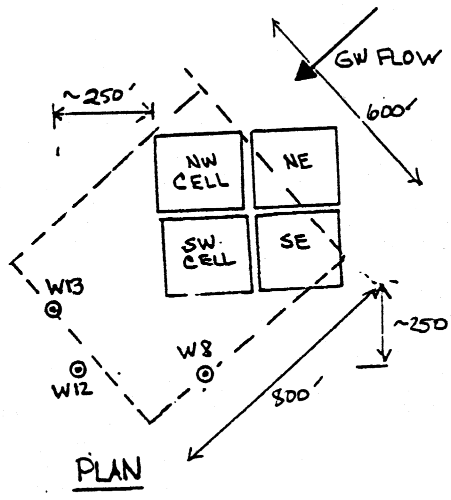
A similar development for calculating K(Θ) was found in the following reference:

Olson, Roy E. and David E. Daniel, "Measurement of the Permeability of Fine Grained Soils", paper for the ASTM Symposium on Permeability and Groundwater Containment Transport. June 21, 1979, Philadelphia, Pennsylvania.

APPENDIX C.4

EVANSVILLE RI SYSTEM - MIXED CELL APPROACH FOR ESTIMATING CHLORIDE CONCENTRATIONS IN THE GROUNDWATER





Conservation. of Mass: Mass into Volume = Mass out

$Vw(Cn)$	+	$F(C_r)$	+	$V_{gw}(C_{gw})$	+	$\Delta V_{gw}(C_{\Delta vgw})$	=
↓		↓		↓		↓	
mass waste in		mass recharge in		mass in represented G.W. volume		mass in from background	

$V\ mix\ (C\ mix)$	+	$\Delta\ V\ mix\ (C\ mix)$
↓		↓
mass out to mix zone		incremental increase to mix zone

<u>Where</u>:

Vw = Volume Wastewater = 300,000 gpd. = 109 million gallons/year
Cw = 245 mg/L Chloride concentration in the wastewater
R = Recharge = 10"/year or 3 million gal/year
C_r ~ 1 mg/L Chloride concentration in recharge due to precipitation
V_{gw} = X(y)(b)(n) = (800 ft)(600 ft)(35 ft)(.3)(7.5 gal/ft^3) = 37.8 million gal
ν = Interstitial velocity = $\frac{q}{n} = \frac{K\Delta h}{h\Delta x}$ = 1 ft/day
(Defined groundwater volume based on boundaries of interest and width of waste disposal system.)

C_{gwo} = Background chloride conc. ~30 mg/L (varies from 14 to 45)

APPENDIX C.4 (continued)

$$\Delta Vgw = \nu\underbrace{[(y)(b)(n)]}_{\text{pore area of volume}}\ (7.5)$$

$$= 1\ \text{ft/day}\ (365\ \text{days/year})\ (600\ \text{ft})(35\ \text{ft})(.3)(7.5\ \text{gal/ft}^3)$$
$$= 17.25\ \text{million gal}$$

$Vmix = Vgw = 37.8$ million gal.
$\Delta Vmix = \Delta Vgw + R + V_W = 17.25 + 3 + 109 = 129.25$ million gal.

Determine Cmix, (concentration in represented volume after n years)

$$Cmix = \frac{Vw(Cw) + R(Cr) + Vgw(Cgw) + \Delta Vgw(Cgwo)}{Vmix + \Delta Vmix} \quad \text{(per year basis)}$$

Example Calculation:

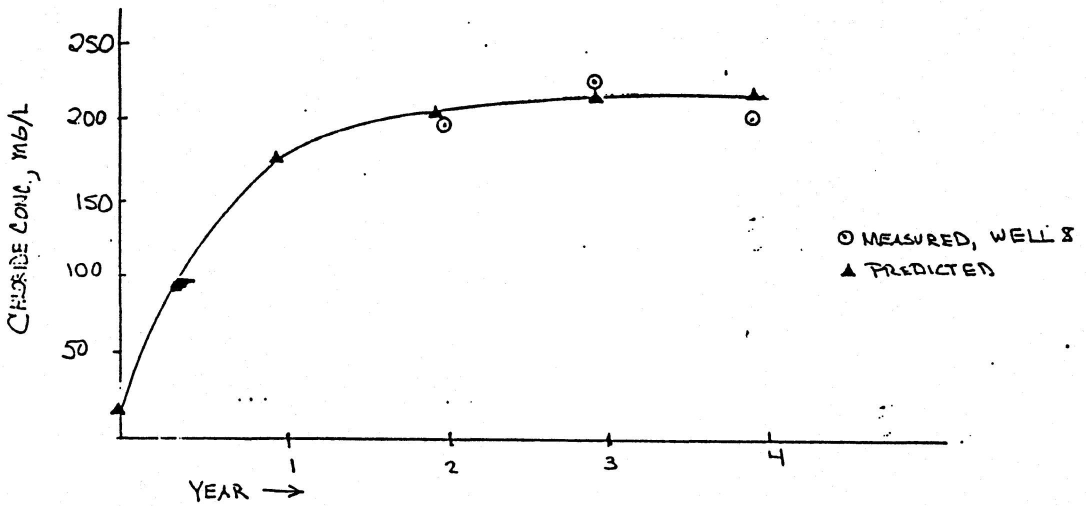
Year 1:

$$Cmix = \frac{109.5(245) + 3.(1) + 37.8\ (30) + 17.25(30)}{37.8 + 129.25} = \underline{\underline{172\ mg/L}}$$

for year 2, 3, etc., use Cgw from previous year.

Comparison with measured chloride concentrations:

	Year	Predicted Cl Conc. in Mix Zone	Measured Cl Conc. (mg/L)			
			W102	W8	W12	W13
1983 (system on line)	1	172 mg/L	208 (7/83)	---	----	---
1984	2	204 mg/L	236/180 (4/84)(12/84)	190 (12/84)	---	---
1985	3	211 mg/L	230 (mo. ave.)	220 (mo.ave.)	200 (11/84)	230 (11/84)
1986	4	213 mg/L	280 (1/86)	200 (1/86)	211 (1/86)	210 (1/86)



APPENDIX C.5

EVANSVILLE NITROGEN BALANCE ESTIMATION

MONTH	FLOW AVE MGD	EFFLUENT TOTAL NITROGEN CONC. MG/L	EFFLUENT TOTAL NITROGEN LOADING LB/DAY	EFFLUENT TOTAL NITROGEN LOADING LB/MO	WELL 5S TOTAL NITROGEN CONC. MG/L	WELL 5S TOTAL NITROGEN LOADING LB/DAY	WELL 5S TOTAL NITROGEN LOADING LB/MO
12/84	.284	27.8	65.8	2040	14.3	33.5	1040
1/85	.274	29.0	66.3	2055	17.8	40.7	1260
2/85	.379	29.8	94.2	2640	23.1	73.1	2056
3/85	.364	23.0	69.8	2165	24.6	74.7	2315
4/85	.343	20.6	58.9	1765	24.5	70.1	2100
5/85	.300	17.0	42.5	1320	23.7	59.3	1840
6/85	.246	14.0	28.7	860	15.8	32.4	970
7/85	.283	13.8	32.2	1000	14.6	34.5	1070
8/85	.242	7.6	15.3	475	13.7	27.7	860
9/85	.259	11.4	24.6	740	10*	21.6	650
10/85	.278	10.7	24.8	770	9.6	22.3	690
11/85	.371	12.5	38.7	1160	11.5	35.6	1068

*Estimated value

TOTAL FOR YEAR ----- EFFLUENT = 16990 LB WELL 5S = 15919 LB

NET LOSS FOR YEAR -------- 1070 LBS ====> 6% REMOVAL

APPENDIX D - BRUNKOW INSTRUMENT INSTALLATION SPECIFICATIONS AND CALCULATIONS

APPENDIX D.1

GROUNDWATER MONITORING WELL INSTALLATION SPECIFICATIONS BRUNKOW ABSORPTION POND SITE

WELL	INSTALLED	TOP OF CASING MSL-FT	APPROX SCREEN DEPTH-FT	SCREEN LENGTH	LOCATION & SOIL TYPE	K FT/DAY
1	11/20/84	1018.99	4.25	5 ft	background in silt loam	-
2	11/14/84	1001.62	9	5 ft	E of Pond 3 in silt layer	.605
3	11/14/84	1003.78	14	2.5 ft	E of Pond 2 in bedrock	.864
4	11/14/84	1003.94	9	5 ft	E of Pond 2 in silt layer	.415
5	11/14/84	999.85	9	5 ft	SE of Pond 3 in silt layer	.975
6	5/22/85	999.32	15	2.5 ft	SE of Pond 3 in bedrock	.860
7	5/22/85	1000.83	7	5 ft	S of Pond 3 in silt layer	.226
8	6/5/85	998.38	6	5 ft	dwngrdt across stream in loam	-
9	5/22/85	1002.61	10	5 ft	S berm of Pond 3 in silt layer	1.5

[1] Hydraulic conductivities were calculated by the method developed by Hvorslev. See Appendix C.1 for a summary and reference to this method. Slug tests were used for wells with screens fully below the water table. Baildown tests were used for wells with screens partially above the water table.

APPENDIX D.2

NITROGEN LOADINGS TO THE SOIL
BASED ON ESTIMATED SEEPAGE RATES IN THE ABSORPTION PONDS

POND 1

Area : 5658 ft^2 Approximate Volume: 127,000 gal (@ ave depth ~3')

Estimated Seepage Rate: 0.15"/day - 0.10"/day evap. - 0" precip. = <u>0.05"/day</u>
(based on measurements 9/11-9/16/85)

Loading Rate: $.05''/\text{day} \times \frac{1\ \text{ft}}{12\ \text{inches}} \times 5658\ \text{ft}^2 \times 7.48\ \frac{\text{gal}}{\text{ft}^3} = \underline{176\ \text{gal/day}}$

$$176\ \text{gal/day} \times \frac{1}{.13\ \text{acres}} = 1356\ \text{gal/acre/day}$$

Ave. TKN (Conc.) mg/L = 430, Dissolved ~ .9 (TKN) = .9(430) = 387 mg/L

TKN Loading: $176\ \text{gal/day} \times .9(430) \times \frac{8.34}{1\text{x}10^6} = \underline{\underline{0.57\ \text{lb/day}}} = 4.4\ \text{lb/acre/day}$

POND 2

Area : 4278 ft^2 Approximate Volume: 192,500 gal (@ ave depth ~6')

Estimated Seepage Rate: 0.35"/day - 0.10"/day evap. - 0" precip. = <u>0.25"/day</u>
(based on measurements 9/11-9/16/85)

Loading Rate: $0.05''/\text{day} \times \frac{1\ \text{ft}}{12\ \text{inches}} \times 4278\ \text{ft}^2 \times 7.48\ \frac{\text{gal}}{\text{ft}^3} = \underline{667\ \text{gal/day}}$

$$667\ \text{gal/day} \times \frac{1}{.098\ \text{acres}} = 6800\ \text{gal/acre/day}$$

Ave. TKN (Conc.) = 353 mg/L

TKN Loading: $667\ \text{gal/day} \times .9(353) \times \frac{8.34}{1\text{x}10^6} = \underline{\underline{1.77\ \text{lb/day}}} = 18\ \text{lb/acre/day}$

POND 3

Area: 4896 ft^2 Approximate Volume: 220,000 gal (@ ave depth ~6')

Estimated Seepage Rate: 0.2"/day - 0.1"/day evap. - 0" precip. = <u>0.1"/day</u>
(based on measurements 9/11-9/16/85)

Loading Rate: $0.1''/\text{day} \times \frac{1\ \text{ft}}{12\ \text{inches}} \times 4896\ \text{ft}^2 \times 7.48\ \frac{\text{gal}}{\text{ft}^3} = \underline{305\ \text{gal/day}}$

$$305\ \text{gal/day} \times \frac{1}{.112\ \text{acres}} = 2714\ \text{gal/acre/day}$$

Ave. TKN (Conc.) = 177 mg/L

TKN Loading: $305\ \text{gal/day} \times .9(177) \times \frac{8.34}{1\text{x}10^6} = \underline{\underline{.41\ \text{lb/day}}} = 3.7\ \text{lb/acre/day}$

APPENDIX D.2 (continued)

CALCULATIONS FOR POND EVAPORATION - BRUNKOW
September 11-16, 1985

Day	Air Temp. °C	$\frac{S}{S+\gamma}$	σT^4 * mm/day	Rg mm/day (langley/ day)	RTN mm/day	Rn mm/day	E mm/day
9/11 Clear	14.4	.622	13.7	8.35 (491)	3.05	3.63	2.89
9/12 am Clear pm Cloudy	12.8	.611	13.58	6.44 (379)	3.12	2.03	1.6
9/13 Clear	11.7	.586	13.17	7.35 (403)	3.09	2.8	2.1
9/14 am Clear pm Cloudy	12.2	.593	13.26	5.67 (334)	3.08	1.46	1.1
9/15 Clear	12.2	.593	13.26	8.1 (477)	3.08	3.4	2.58
9/16 Cloudy Windy	17.2	.656	13.30	7.09 (416)	2.76	2.91	2.55
							12.7 mm

*Ref: Tanner & Bouma paper on "Influence of Climate on Subsurface disposal of Sewage Effluent"

The method for estimating evaporation is outlined on the following page.

$ET = 1.28\ [S/(S+\gamma)]Rn$ (no advection)

Where: s = slope of saturation vapor presure curve corresponding to ambient air temp.
γ = psychrometer constant
Rn = net radiation in evap. units (mm/day)

$[s/s+\gamma]$ was determined for different air temps from tables listed in Reference, (Tanner & Bouma paper) for different air temps.

$Rn = (1-r)R_G - R_{TN}$

R_G = solar radiation
r = albedo (~.2)
R_{TN} = net long wave thermal radiation loss

R_{TN} (clear) $= (\Sigma T^4)\ [0.26 \exp(-7.77 \times 10^{-4}\ T_c{}^2)]$

ΣT^4 = black body radiation corresponding to absolute temp, T (table values taken from reference cited above)

T_c = mean air temp in °C

Rg = solar radiation in Langley/day (cal/cm^2/day)

To convert to mm/day, if above freezing, 1 mm = $(59.5-0.05\ T_c)$ cal/cm^2

$$\text{Then } Rg \times \frac{1 \text{ mm}}{59.5-0.05\ T_c \text{ cal/cm}^2} = Rg \text{ in mm/day}$$

Example calculation:
on 9/11:

$$Rg = \frac{491 \text{ langley/day}}{(59.5-0.05(14.4))} = 8.35 \text{ mm/day}$$

$$R_{TN} = (13.7)\ [.26 \exp\ (-7.77 \times 10^{-4}\ (14.4)^2)] = 3.05 \text{ mm/day}$$

$$R_N = (1-.2)8.35 - 3.05 = 3.63 \text{ mm/day}$$

$$E = 1.28\ [.622]\ 3.63 = 2.89 \text{ mm/day} = .11 \text{ inches on 9/11}$$

The total estimated depth of evaporation from the ponds is 12.7 mm for the period 9/11-9/16 (.5 inches).

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Groundwater Impacts of
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