AN INVESTIGATION OF THE SURFACE WATER CONTAMINATION POTENTIAL FROM ON-SITE SEWAGE DISPOSAL SYSTEMS (OSDS) IN THE TURKEY CREEK SUB-BASIN OF THE INDIAN RIVER LAGOON

Project Funded by:

St. Johns River Water Management District SWIM Project IR-1-110.1-D

Under Florida Department of Health and Rehabilitative Services (HRS)

Contracts No. LP114 and LP596

Final Report

February 1993

AYRES ASSOCIATES

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EXECUTIVE SUMMARY

This report presents the methods and results of a study conducted to determine the potential impact from OSDS (onsite sewage disposal systems) on water quality in the Turkey Creek Sub-Basin of the Indian River Lagoon. Groundwater and surface water samples were collected in a residential subdivision in Palm Bay, Florida in a site-specific study of individual OSDS. This subdivision was typical of many OSDS subdivisions in the area in that surface water and groundwater drainage from the site flows via canals to the Indian River Lagoon.

The primary objective of the Indian River Lagoon OSDS Study was to assess the impact of several existing OSDS on water quality, particularly nutrient and bacteriological quality, in adjacent canals. A secondary objective of the study was to add to the database on migration of pollutants from individual OSDS and to evaluate pollutant attenuation in the subsurface environment below and downgradient from such systems.

To accomplish these objectives, two different residential OSDS, and an undeveloped control site were investigated over a two year period to determine impacts on a drainage canal which empties into the Indian River Lagoon.

Septic tank effluent samples were collected and analyzed to characterize the quality of wastewater discharged to the OSDS drainfields. Water meter readings were collected to estimate the average wastewater flow to the OSDS drainfields. Twenty five (25) monitoring wells and twelve (12) piezometers were clustered at the two specific homes and a control site in the study area. Surface and groundwater samples were collected on fourteen (14) different sampling dates over a study period from February 1990, through March, 1992 and analyzed for key water quality parameters indicative of OSDS impacts.

Seepage meters and canal piezometers were installed in the canal bottom to determine seepage rates, in an attempt to estimate nutrient loading to the canal. Canal surface water and seepage water samples were collected and analyzed for the same parameters as the groundwater collected from the monitoring wells.

Depth to the groundwater water table was measured during each sampling event and used in conjunction with survey data to determine groundwater flow direction. Aquifer testing and a bromide tracer test were conducted to determine travel time through the unsaturated zone and groundwater seepage velocity.

The residences studied were typical of those in the Port Malabar subdivision and utilized separate blackwater and graywater septic tanks and drainfields. Water use monitoring indicated wastewater loading rates to the drainfields were below design loading rates per Chapter 10D-6, F.A.C.

The soils of the study area were typical of the South Florida Flatwoods land resource area and consisted of Myakka sand at the Jones site, Oldsmar sand at the Groseclose site, and Eau Gallie sand at the control site. A sandy clay loam layer was encountered at depths of five to seven feet at the Groseclose site.

Based on the data collected in the study the following conclusions regarding groundwater quality and the potential impact to surface water quality from OSDS were drawn:

- Groundwater flow direction at both residences and the control site was in the general direction of canal 66, to the north. Groundwater elevation monitoring indicated an unsaturated soil thickness below the drainfields which varied over the study from 1.2 to 2.9 ft. at the Groseclose site and from 3.3 to 5.2 at the Jones site.
- Bromide tracer testing at the Groseclose site indicated an unsaturated zone travel time of 5 days below the drainfield and an average groundwater seepage velocity of 0.24 ft./day towards the canal.

- Analysis of groundwater and surface water samples from wells located at different distances from the OSDS drainfields indicated that the concentration of nitrate, nitrite-nitrogen (NO_3,NO_2-N), total kjeldahl nitrogen (TKN), total phosphorus (TP), and conductivity were generally significantly higher in the vicinity of the drainfield when compared to the background wells. However, contaminant concentrations were at or below background concentrations in wells located twenty (20) to forty (40) feet from the drainfield.

- Fecal coliform counts in the monitoring wells were generally below 10 cols./100 ml on two-thirds of the sampling dates. Fecal streptococcus levels were high in all wells, generally ranging from 100 to 2000 col./100 ml (geometric mean). Fecal streptococcus and fecal coliform bacterial data did not statistically ($p \leq 0.05$) indicate significant reductions in number with increasing distance from the drainfield. The high levels of fecal streptococcus encountered in the groundwater at the Groseclose site were thought to be attributable to the utilization of canal water for lawn irrigation at this site and the presence of ducks, geese, and chickens nearby. (See following conclusion).

- Comparison of fecal coliform (FC) bacterial data to fecal streptococcus (FS) data indicated a wildlife rather than human source of contamination. The FC/FS ratios in monitoring wells were very low. The average monitoring well FC/FS ratio was 0.04 which is indicative of a non-human source of fecal contamination.

- Bacterial counts were high and variable in the surface water obtained from canal 66. Fecal coliform and fecal streptococcus levels peaked at canal station C-3 which is located near the Groseclose site. The peak levels of bacteria appeared to be related to the presence of numerous ducks, geese, and chickens located near this sampling station.

- This was supported by FC/FS ratios at the receiving canal stations (C-0 through C-5) which averaged 0.17:1, also indicating the likelihood of non-human sources of pollution. The FC/FS ratio also suggested that stormwater run-off may be a source of bacterial loading to canals.

- Based on the canal water sampling, OSDS impacts on the receiving canal water quality were not evident. There were no statistically significant ($p \leq 0.05$) relationships between nutrient concentrations in the canal surface water and sampling locations in the canal relative to OSDS.

- Considerable increases in concentrations of several parameters were measured towards the end of the study period. Nitrate-nitrogen concentrations increased in groundwater obtained from monitoring wells located within twenty-five (25) feet of the blackwater drainfields. Phosphorous and TKN concentrations also increased in some of the wells. At the Jones site, peak nitrate-nitrogen concentrations exceeded 50 mg/L at several wells located within twenty (20) feet of the blackwater drainfield. Total phosphorous and TKN concentrations were also elevated. Fecal coliform also increased during the August 1991 and September 1991 sampling events. It was speculated that these increases were due to higher water table elevations or a shift in groundwater flow direction, but further monitoring would be required to determine the specific cause.

- Based on data collected after rainfall events (five {5} occasions), no conclusive cause-effect relationships on either groundwater or surface water quality were found.

- Seepage fluid water quality data generally indicated that seepage meter water quality may not be directly comparable to monitoring well or even canal piezometer data and, in turn, may not be useful for the determination of nutrient loading to the canal. Based on parameter concentrations encountered in the seepage water, seepage meter water quality was probably effected by conditions within the seepage meter itself.

- Data collected from bromide tracer tests at the Groseclose site indicated that conservative parameters such as nitrate and chloride should reach the canal from the drainfield in approximately two hundred seventy (270) days, yet concentrations of these compounds were measured at background levels within twenty (20) to forty (40) feet of the drainfields. Calculations of "dilution factors" indicated that, although some dilution may be responsible for these results, it also appeared that phosphorous was significantly attenuated by onsite soils and that denitrification was contributing to substantial nitrogen removal. Additional monitoring of the bromide tracer should be conducted to more accurately estimate dilution.

- The results of the study indicated that while OSDS were impacting groundwater in their immediate vicinity, they were not impacting canal water quality significantly at the time of this study. This may not continue indefinitely however, and it was estimated that total phosphorous loading to the canal may eventually reach a maximum of 1 to 2 kg/home/year for homes bordering the canal. Although nitrogen was significantly reduced at the study sites (especially Groseclose), it was estimated that under unfavorable conditions, total nitrogen loading from homes bordering the canal could be as high as 4 to 7 kg/home/year. Fecal bacterial impacts to the canal could not be assessed from the variability of the data collected. A better indication of fecal bacterial impacts than fecal coliform is needed.

Based on the results of this study and the conclusions listed above, Ayres Associates recommends that the Water Management District complete a preliminary nutrient budget for the Indian River Lagoon from all sources, utilizing the estimated loadings above for OSDS inputs. If the OSDS nutrient loading is a significant part of the overall nutrient budget to the lagoon, additional study to refine the nutrient loading estimates above would be recommended. If these estimates proved accurate, an investigation of nutrient reduction techniques for OSDS should be initiated.

I. INTRODUCTION

Florida is experiencing a rapid rate of growth, with a significant portion occurring in new developments located outside a sewer service area. Homes and business establishments in unsewered areas must rely on on-site sewage disposal systems (OSDS) for wastewater treatment. Conventional OSDS typically consist of a septic tank followed by a subsurface infiltration system that utilizes the natural soil's capacity to treat wastewater before ultimate recharge to the groundwater. Currently, over 1.5 million households in Florida utilize OSDS (Ayres Associates, 1987). According to unpublished data of the Florida Department of Health and Rehabilitative Services (HRS), Florida has issued an average of 40,000 to 60,000 new OSDS permits annually since 1983.

It was estimated in a recent report on OSDS use in Florida that over 75,000 OSDS existed in the three counties that border the Indian River Estuary as of 1985 (Ayres Associates, 1987). In the three years 1984 to 1986, Brevard, St. Lucie, and Indian River Counties ranked 1st, 18th, and 22nd respectively, in the number of OSDS permits issued relative to Florida's 67 counties. Together, these counties accounted for over 22,000 new OSDS installations during those three years.

The Indian River Lagoon (IRL) is a vital biological and economic waterway to eastern Florida, and has been designated as a priority water body under the Surface Water Improvement and Management Act (SWIM) by two water management districts. Recently, the lagoon was included in the National Estuary Program by the EPA. The IRL system is a biogeographic transition zone rich in habitat and species, exhibiting the highest species diversity of any estuary in North America (SJRWMD, 1989). There have been 2,200 species identified, thirty-six of which are listed as endangered or threatened (SJRWMD, 1989).

With the tremendous population growth in the IRL region in the early 1900's extensive drainage projects were undertaken to render the land more suitable for agricultural and urban uses. These drainage projects resulted in an extensive canal network that crisscrosses central and south Florida. Many of these canals eventually drain into the IRL and introduce large amounts of freshwater into the lagoon.

With rapid population growth, many developments were located outside an existing municipal sewer system and were developed utilizing on-site sewage disposal systems (OSDS) for their wastewater treatment and disposal. These OSDS (septic tank) systems utilize the soils ability to treat the wastewater before being allowed to enter the groundwater. The soil is capable of treating organic materials, inorganic substances, and pathogens in wastewater by acting as a filter, exchanger, absorber, and a surface on which many other physical, chemical and biological processes occur (Clements and Otis, 1980). When site and soil conditions are favorable, these systems generally produce water of acceptable quality for discharge into the groundwater. Under saturated soil conditions however, the wastewater moves faster through the soil and may exceed the soil's capacity to properly treat the effluent, thus allowing water which may be high in nutrients and other contaminants to enter the groundwater. In addition, OSDS transform the nitrogen species in wastewater to nitrate-nitrogen, which moves readily with soil water and groundwater. Thus, there is concern that high densities of septic systems in a given area and the resulting large volumes of wastewater may lead to groundwater and surface water contamination if housing density is increased or if suitable unsaturated soil thicknesses are not present.

This rapid development and the decreasing water quality in the Indian River Lagoon Estuary has caused concern that developments utilizing OSDS may be directly or indirectly contributing to the estuary's pollutant load. This concern needs to be investigated because of the estuary's importance as a fishery and shellfish harvesting area. Water quality violations have caused shellfish harvesting bans in the estuary several times in recent years. This study was initiated to examine the potential for surface water contamination from subdivisions served by OSDS in an area which drains to the Indian River Lagoon.

Objective and Scope

This study of OSDS impacts to surface water in the Indian River Lagoon was conducted as part of the State of Florida's Surface Water Improvement and Management (SWIM) Program. The Indian River Lagoon SWIM Plan was developed by the St. Johns River Water Management District (SJRWMD) and the South Florida Water Management District (SFWMD), and its goals are to monitor, improve, and protect the water quality of the Indian River Lagoon. The study was administered by the Florida Department of Health and Rehabilitative Services (HRS) under contract to SJRWMD and is funded by the SJRWMD SWIM Program. The study itself was conducted by a team of engineers, hydrogeologists, and environmental scientists from Ayres Associates in Tampa and the Florida Institute of Technology (FIT) in Melbourne, Florida.

The primary objective of the Indian River Lagoon OSDS Study was to assess the impact of several existing OSDS on water quality in adjacent canals which eventually drain to the Indian River Lagoon. A secondary objective of the study was to add to the database on migration of pollutants from individual OSDS and to evaluate pollutant attenuation in the subsurface environment below and downgradient from such systems.

The scope of the study was to investigate groundwater and surface water quality around two existing OSDS and at one control area which is presumably unaffected by OSDS inputs. In order to determine the potential impacts, the study included estimation of pollutant loading to the groundwater and adjacent drainage canals from two OSDS's and a control area located within the Turkey Creek Sub-basin of the Indian River Lagoon basin. To evaluate these impacts, the following general tasks were included in this study:

1. Selection and characterization of the study sites.
2. Determination of groundwater flow characteristics.
3. Monitoring of water quality in wells located upgradient and downgradient of each study area.
4. Determination of seepage rates of groundwater into the canal.
5. Monitoring quality of water seeping into the canal.
6. Monitoring water quality of the canal water itself.

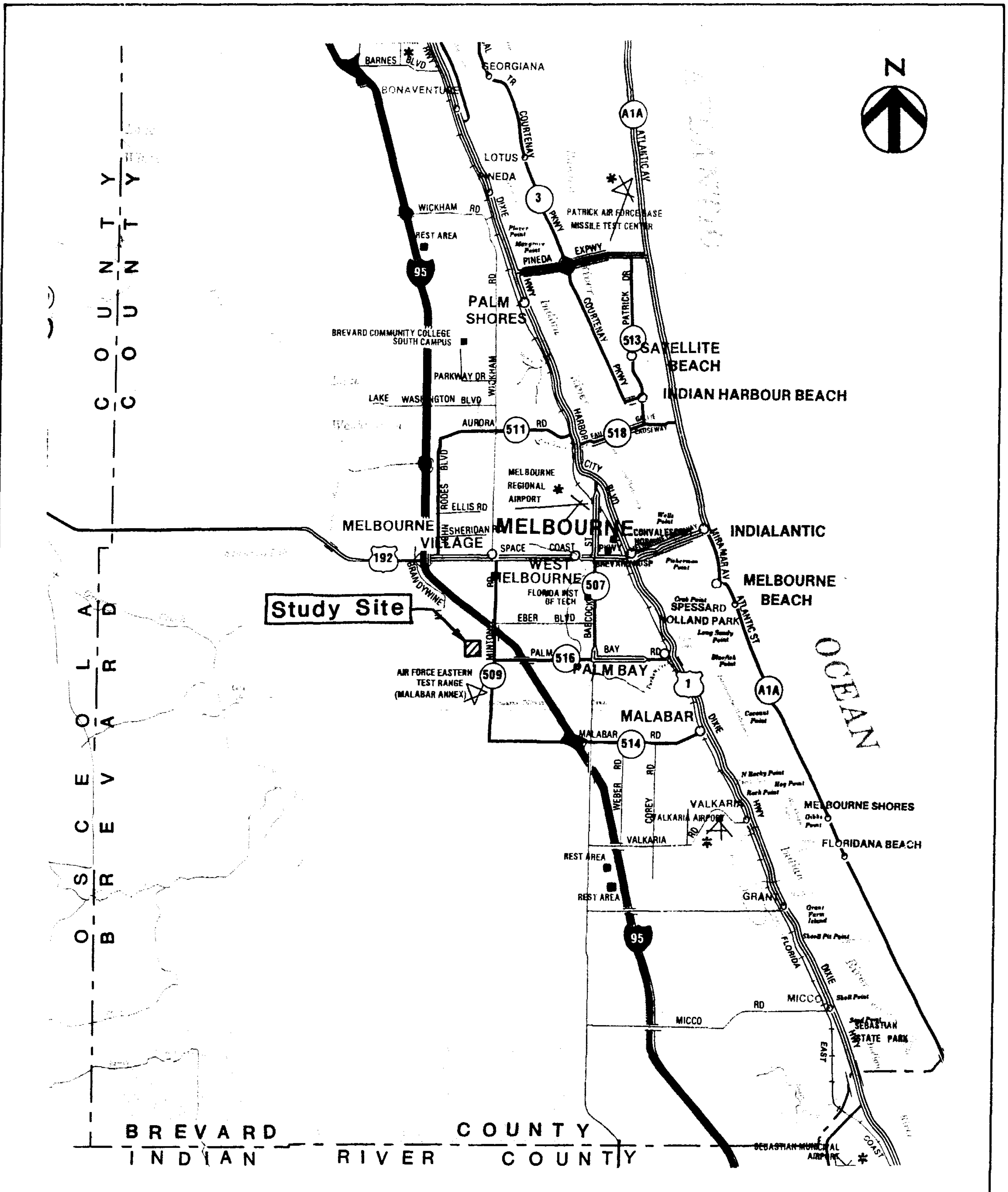
This report describes the sites selected for study, the methods used in the study, and the results of monitoring and data analyses at the study sites. Finally, a discussion of results and conclusions, and recommendations from the study are presented.

II. SITE SELECTION AND DESCRIPTION

The site selected for the Indian River Lagoon project was the Port Malabar Subdivision located in the City of Palm Bay in Brevard County, Florida. Figure 1 shows the general location of the study area. The subdivision was previously one of the areas monitored as part of the Department of Health and Rehabilitative Services On-site Sewage Disposal System Research Project and was selected partly because of the existing data available as part of the HRS research (Ayres Associates, 1989). Port Malabar is a relatively new subdivision of immense size (over 70,000 platted lots) and the majority of home owners depend on OSDS's for treatment and disposal of their wastewater. Port Malabar is drained by a series of man-made drainage canals which discharge into the Indian River Lagoon via Turkey Creek.

The soils in the study area are typical Florida flatwoods soils. In its original state, the subdivision area was made up of nearly level pine and palmetto flatwoods interspersed with small to large grassy ponds and sloughs. The seasonal high water table in the flatwood areas was at a depth of less than ten inches for one to four months out of the year and between ten and forty inches for more than six months. The shallow ponds and sloughs received surface runoff as well as subsurface water moving laterally from the surrounding soils and were typically ponded for more than six months in most years (Unit 45 Task Force, 1986).

The Soil Survey of Brevard County (USDA Soil Conservation Service, 1974) identifies the soil series in the area as Eau Gallie Sand; Eau Gallie, Winder, and Felda Soils, ponded; Malabar sand; and Malabar, Holopaw and Pineda soils. These soils are all nearly level poorly drained sandy soils. The upper horizons consist of fine sands and may exhibit a hardpan or spodic layer at depths from 24 to 40 inches below the surface.



SCALE NTS	DRAWN BY: NA	DATE: —	LOCATION OF STUDY AREA IN FLORIDA	FIGURE: 1
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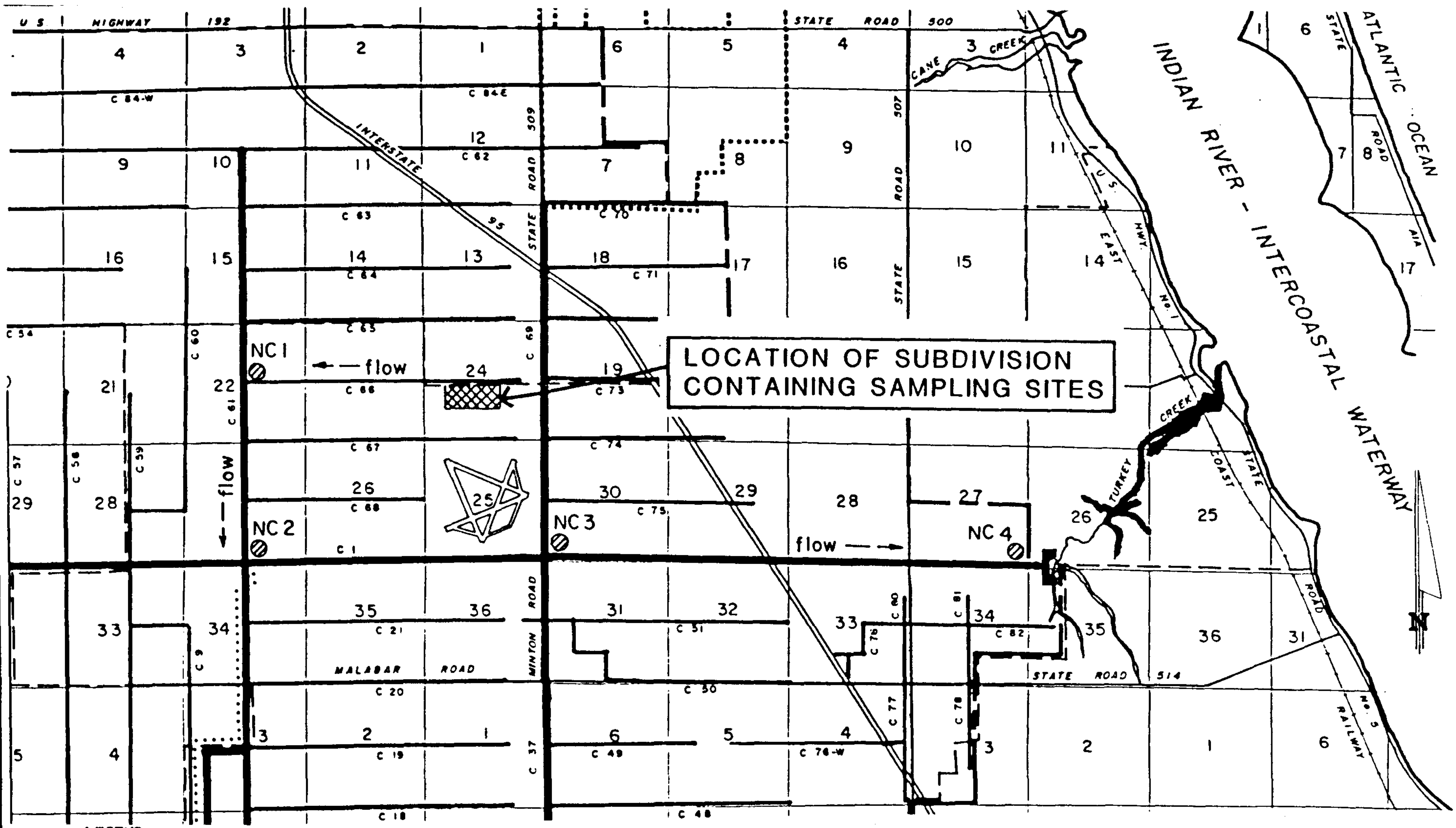
These layers have lower permeabilities than the sandy soils and in some cases may perch water after wet periods. Below the hardpan, or at depths of 40 to 60 inches, these soils typically exhibit a finer textured soil layer, usually sandy loam or sandy clay loam. These layers have a slow or moderately slow permeability rate and also may develop perched water tables.

In the 1970s a system of drainage ways were constructed to develop the Port Malabar area. Roadside swales direct excess surface water runoff from lots and roadways to drainage ditches which, in turn, discharge into a larger system of ditches and canals. The primary drainage system in the area consists of canals C-1, C-69, C-74 and C-75, which are deeply cut drainage canals that penetrate the shallow groundwater table and increase the discharge of groundwater from the area (Unit 45 Task Force, 1986). All canals in the area discharge via canal C-1 into Turkey Creek and ultimately into the Indian River Lagoon System. (See Figure 2).

Initially, fill dredged from the ditches and canals was added to the subdivision area. As lots were developed, additional fill was brought in to raise the building foundations and in some cases to meet the site requirements for on-site sewage disposal systems outlined in Chapter 10D-6 of the Florida Administrative Code. These requirements have varied over the years but currently require 54 inches of suitable soil below the OSDS drainfield infiltrative surface of which 24 inches must remain unsaturated at all times of the year. To meet this requirement, the hardpan and/or clayey layers are typically excavated in the immediate area of the drainfield and replaced with sandy soil, and additional fill material is placed over the remaining lot area as needed.

Two individual home OSDS were selected for study as part of this project. The homes were selected after mailing questionnaires to many homes in the area which are located along the drainage canal, and then screening the home owners by personal interviews and inspections of their OSDS. The final determination of the two sites was influenced



LEGEND

C - CANAL

NC1-4 - DOWNSTREAM CANAL SAMPLING STATIONS

SCALE NTS		LOCATION OF STUDY SITE AND DRAINAGE PATHWAY TO THE INDIAN RIVER LAGOON	FIGURE: 2
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by the characteristics of the household and accessibility to the area for drilling and sampling. The location of the two homes selected are labelled and shaded on Figure 3. Both homes are adjacent to Canal 66.

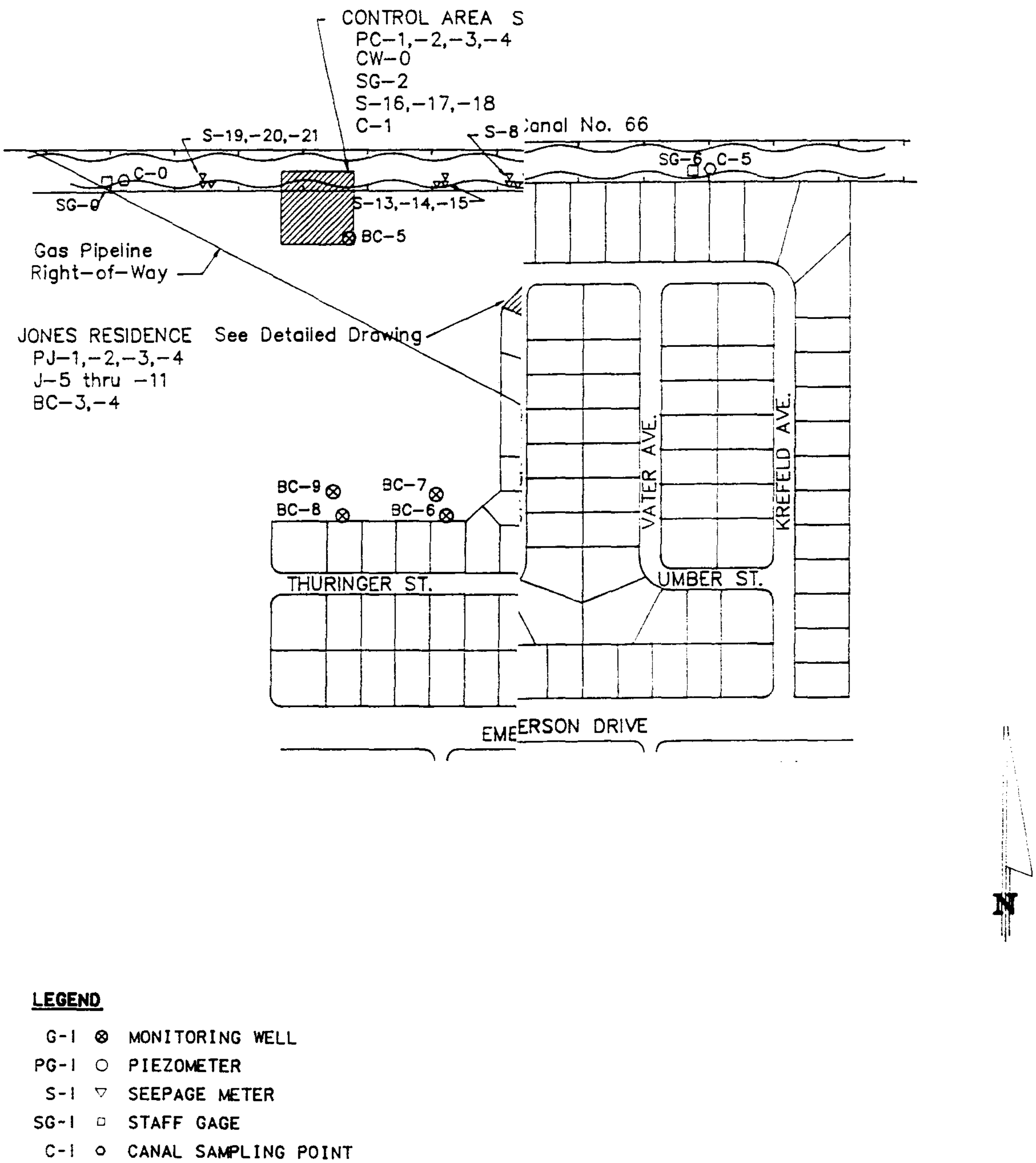
In addition to the two home sites, a control area was selected in a nearby undeveloped area along the same drainage canal. The purpose of the control area was to monitor groundwater quality in an area unaffected by OSDS. The site of the control area is also shaded and labelled in Figure 3.

Groseclose Site Description

The Groseclose site was a raised lot directly adjacent to the canal (Figure 4). The fill material used to raise the lot elevation was soil which had been excavated when the canals were constructed then mixed with fill brought in from other areas. The house was constructed in 1983 and had separate graywater/blackwater septic tanks located behind the house. There were five people, two adults and three teenagers (16 to 18 years old), living in the house at the time of the study. The house had two bathrooms. The owner had never had a problem with his septic tank or had it cleaned at the initiation of the study. The yard was watered with canal water and lightly fertilized every two months.

Jones Site Description

The Jones residence (Figure 5) is a slightly raised lot in a wooded area a short distance further from the canal than the Groseclose residence. Because the Jones residence is not directly adjacent to the canal, it is assumed that fill which was used to raise the lot was probably brought in from another area. The house was also constructed in 1983 and



SCALE 1" = 400'	DRAWN BY:	DATE: 11/91	FIGURE: 3
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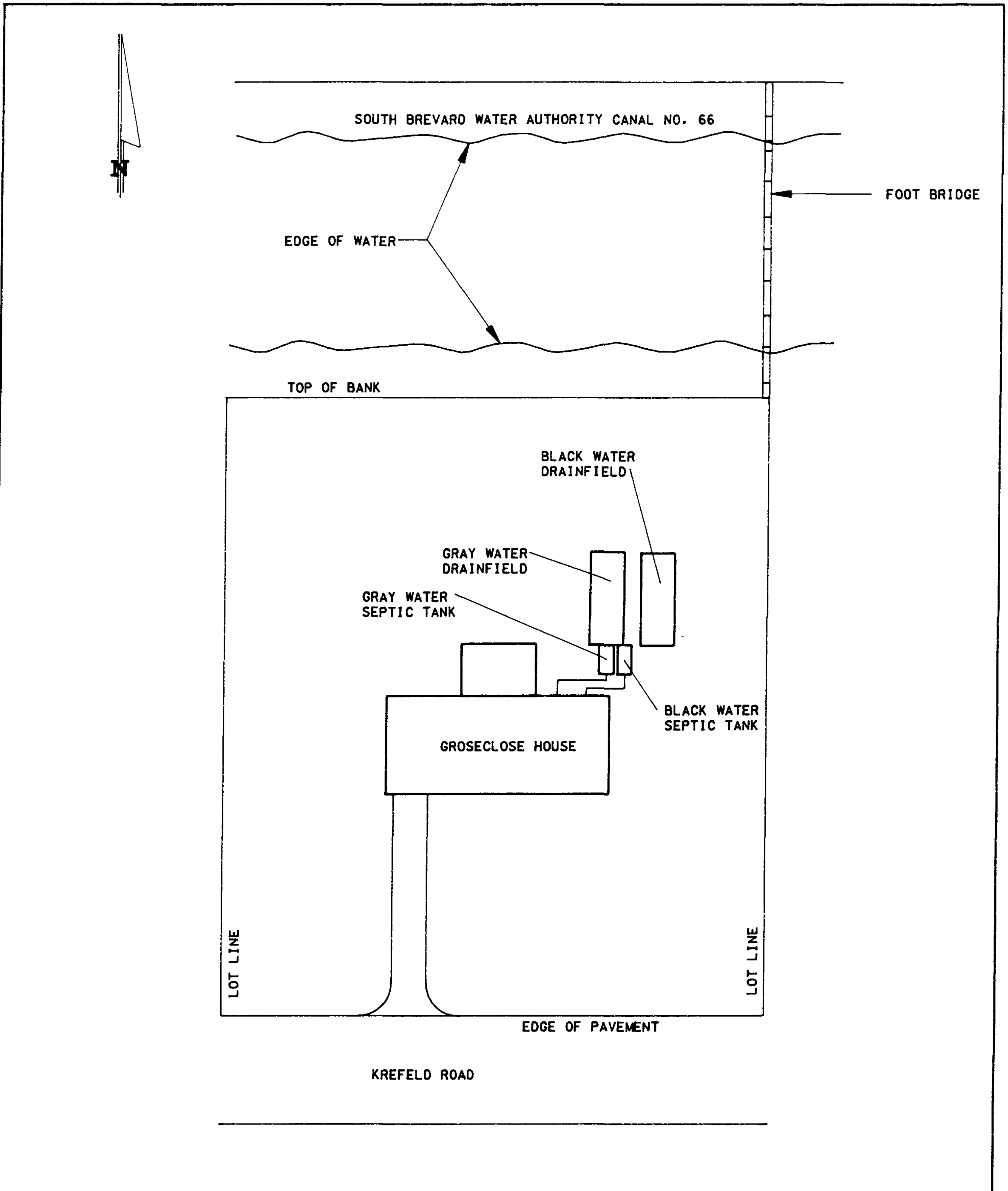

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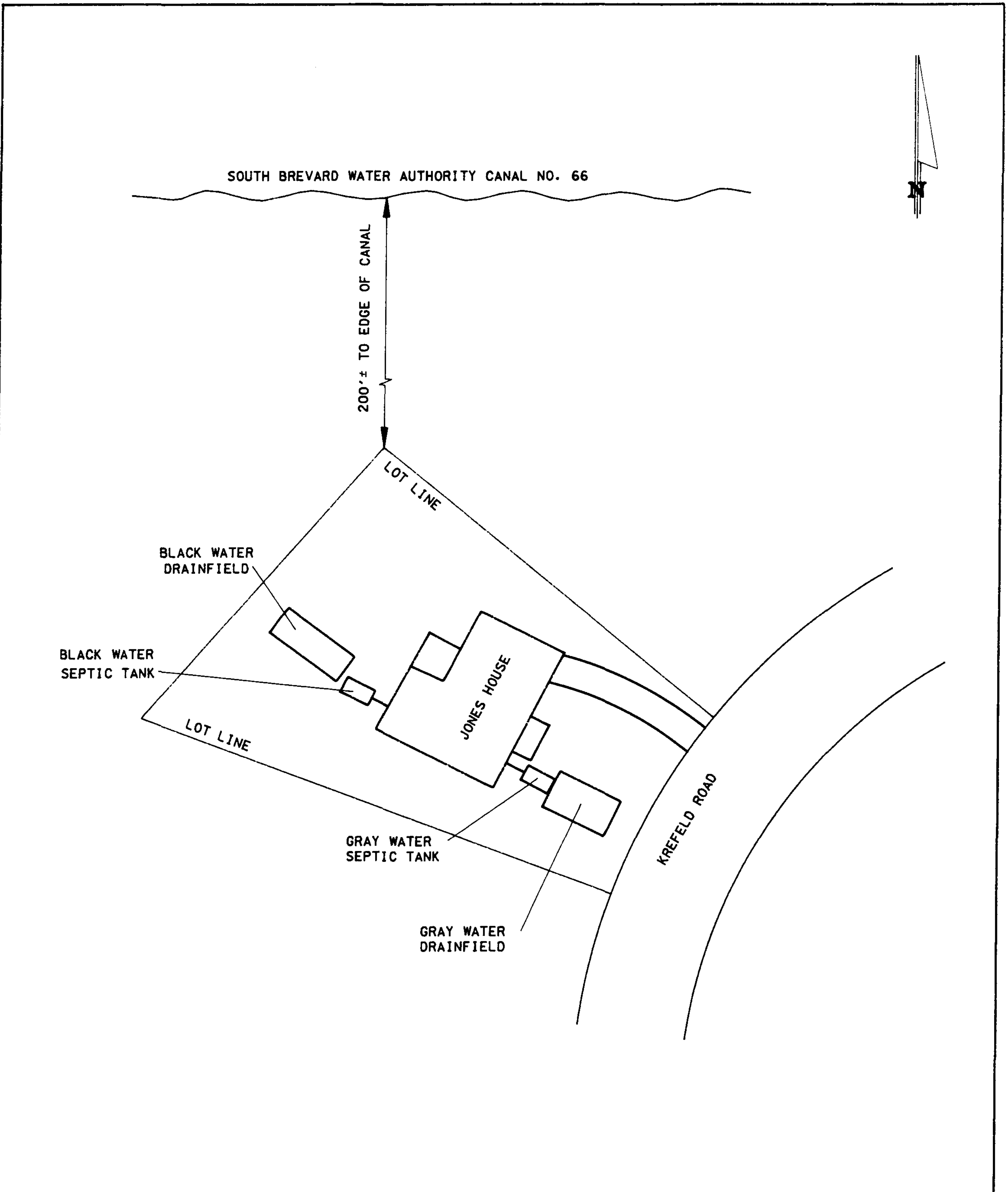
GROSECLOSE RESIDENCE
SITE PLAN

FIGURE: 4



SCALE SCALE: 1"=40'	DRAWN BY:	DATE: 12/92	JONES RESIDENCE SITE PLAN	FIGURE: 5
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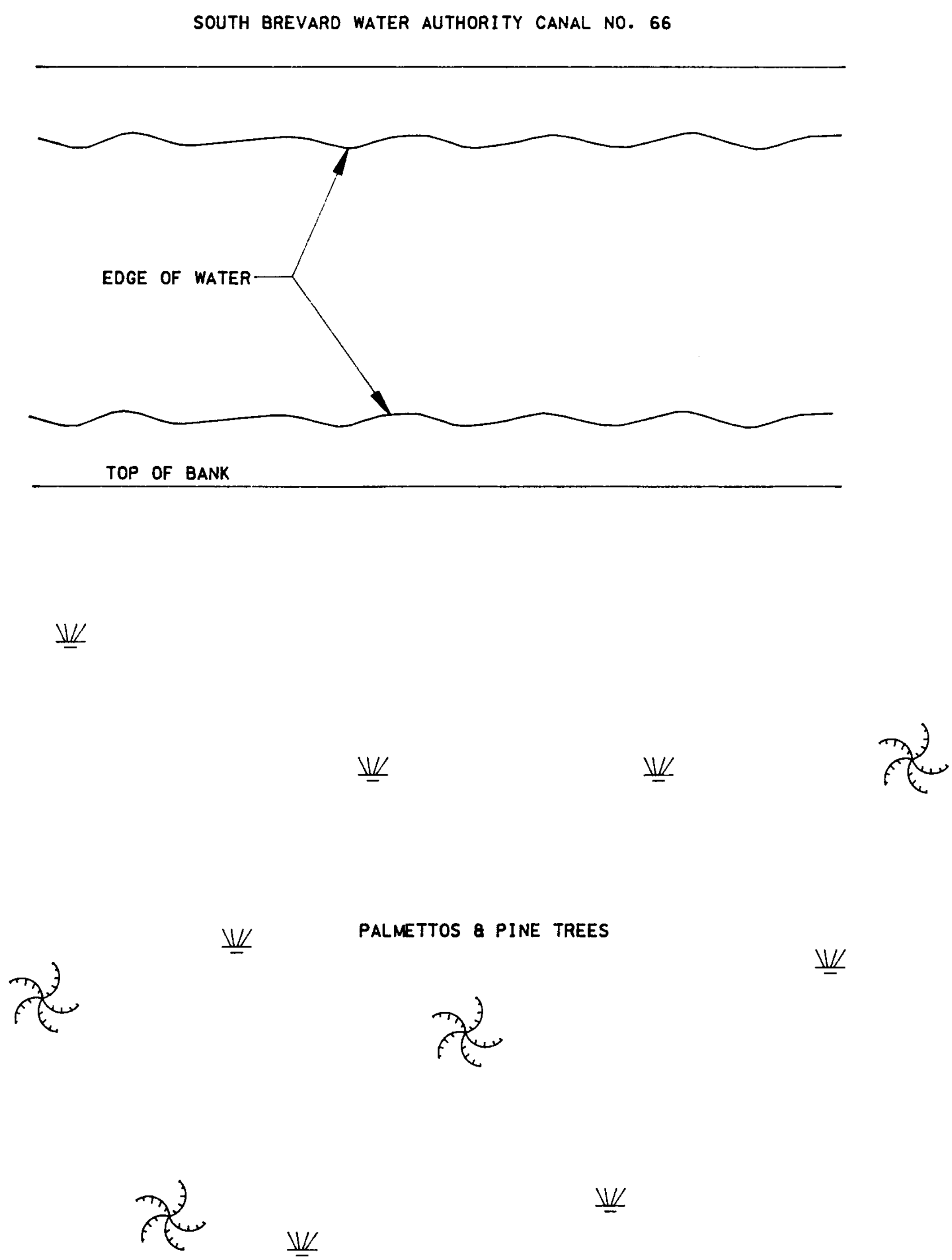
has separate graywater/blackwater septic tanks located in front of and behind the house, respectively. The tanks have never been pumped. There were two adults living in the house at the time of the study. The home has two bathrooms and a garbage disposal system. The home owner has many decorative plants in his back yard and fertilizes this area approximately once every two months. The yard was watered frequently with municipal water.

Control Site Description

The control area (Figure 6) was chosen in a natural area which appeared to be unaffected by development. Older pine trees and palmetto palms are mixed throughout the area and the general elevation of the control site is lower than the developed lots. Groundwater monitoring was conducted in this area to determine if differences existed between groundwater quality compared to the developed subdivision.



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CONTROL AREA
SITE PLAN

FIGURE: 6

III. METHODS AND MATERIALS

Groundwater Elevations and Flow Direction

Groundwater Flow Direction: Groundwater movement in shallow aquifers is generally governed by forces of gravity and, therefore, moves from areas of higher water table elevations to areas of lower water table elevations. Water table elevations can be contoured to distinguish areas of higher or lower water table elevation. The groundwater flow direction is perpendicular, or normal, to these water table elevation contour lines. Water table elevation contour lines are determined by obtaining the depth to groundwater at various locations and referencing that depth to a known elevation at the site.

An initial direction of groundwater flow was determined by the installation of three to four piezometers at each site. Piezometers PJ-1 through PJ-4 were installed at the Jones site, PG-1 through PG-4 were installed at the Groseclose site, and PC-1 through PC-4 were installed at the control site (see Figures 7,8, and 9 in the following section). The elevations of the tops of the piezometer casings were initially surveyed by Ayres Associates on February 7, 1990 and referenced to Mean Sea Level (MSL) based on National Geodetic Vertical Datum (NGVD) of 1929 using City of Palm Bay benchmarks. Depth to groundwater measurements were obtained by measuring from the top of the casing to the water table surface with a chalked steel tape. Subsequent depths to water table measurements were obtained on at periodic intervals in 1990 through 1992 using either a chalked tape or a Keck KIR-89 electronic water level indicator.

Thickness of Unsaturated Soil: Water table elevation data was also obtained to determine the thickness of the unsaturated soil at the site. The thickness of unsaturated soil, described in this study as the thickness of the unsaturated soil layer between the bottom of the drainfield and the groundwater surface, is an important component in a study of OSDS impact to groundwater quality. Theoretically, the greater the thickness of unsaturated soil beneath the drainfield the greater the degree of treatment of septic tank effluent before it reaches to the groundwater.

Construction details of the septic systems were not available, therefore, the depth to the bottom of the drainfield was determined by the installation of observation ports in the drainfields. OPW (Observation Port West) and OPE (Observation Port East) were installed in the drainfields at the Groseclose residence and OPF (Observation Port Front) and OPB (Observation Port Back) were installed at the Jones residence drainfields. The observation ports were installed to the infiltrative surface which is the base of the drainfield and depth to that surface was measured. Depth to groundwater measurements were obtained to determine the range in thickness of the unsaturated soil layer over time.

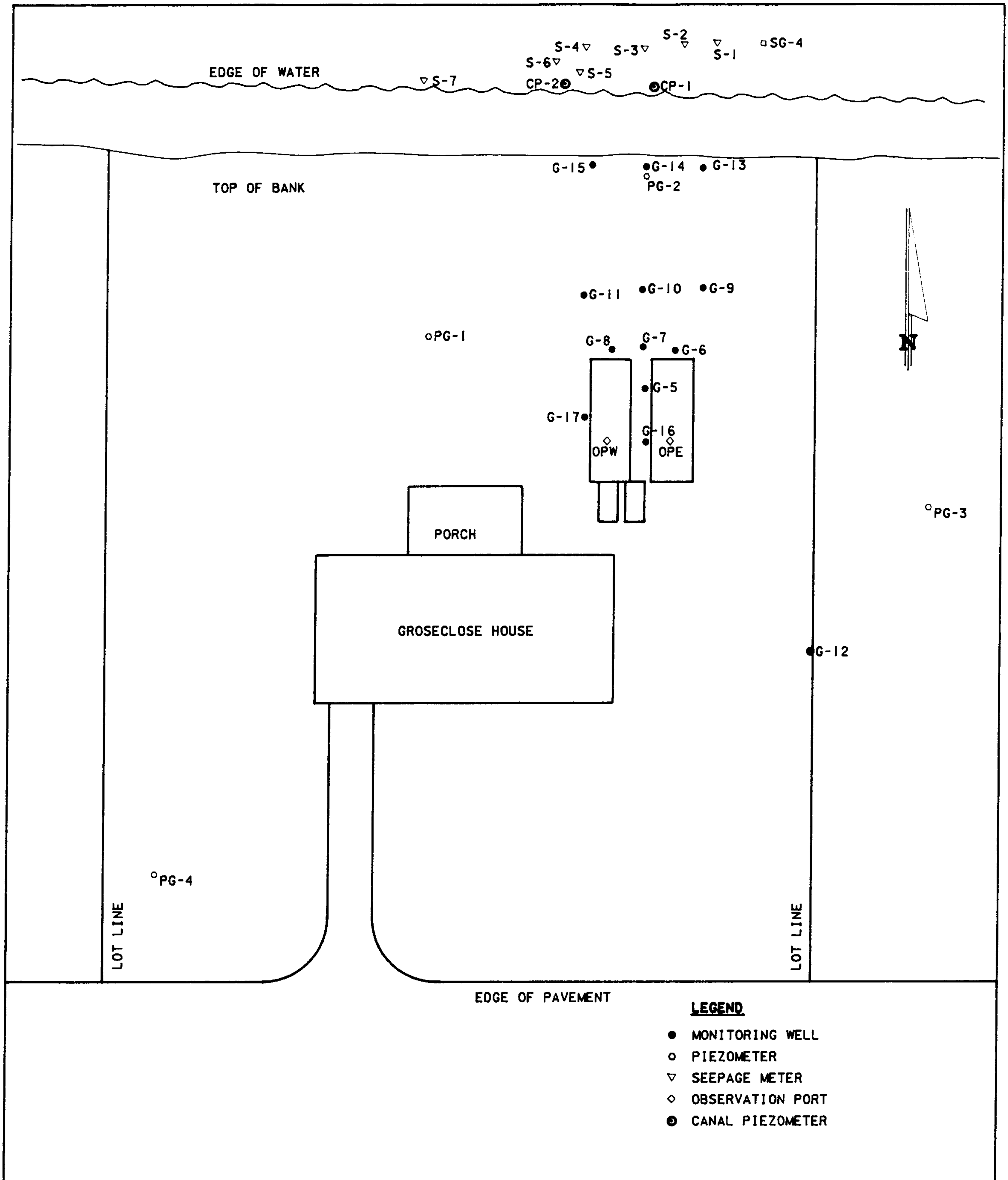
Water level measurements were collected on various dates to determine the seasonal change in water table elevation. Rainfall data was also collected to determine the influence of rainfall on water table elevations. The estimated rainfall was determined by averaging rainfall data collected from the Wilbro Dairy, located approximately 2 miles south of the sampling area, and the West Melbourne Wastewater Treatment Plant which is located 1.5 miles northeast of the sampling site.

Monitoring Well Installation

Piezometer wells and monitoring wells were constructed of 2-inch diameter Schedule 40 PVC riser coupled to 2-inch diameter Schedule 40, 0.010-inch slotted well screen. The screened interval varied from three to eight feet depending on depth to water. Piezometer wells were installed first to determine the general direction of groundwater flow by measuring the change in water table elevation between piezometers. The piezometer wells were installed manually using a four inch stainless steel bucket auger, and were placed so that the screened interval intercepted the water table. The borehole around the well casing was backfilled with native soil from the boring. The well was capped and a well cover was placed flush with the ground surface.

The groundwater monitoring wells were installed using a drilling rig provided by the Brevard County Water Resources Department. Wells were installed downgradient of the OSDS drainfields as indicated by the initial piezometer data. The wells were drilled to a depth of about five (5) feet below the water table or to the top of a sandy clay loam layer encountered at the Groseclose site using a six inch diameter hollow stem auger. The well was sand packed to approximately one foot above the screened interval. The top one foot of the casing was cemented to prevent vertical movement of contaminants. A well cover was installed flush to the ground surface and the well was capped. Soil samples were collected from the various depths while the well was being installed for particle size and organic matter analysis.

Groseclose Site Wells: Once the direction of groundwater flow was determined, monitoring wells were placed from the edge of the drainfield towards the canal in the direction of groundwater flow. Monitoring wells G-5 and G-16 were placed between the two septic tank drainfields. Wells G-6, G-7, and G-8 were placed three (3) feet from the edge of the drainfields. Wells G-9, G-10, and G-11 were placed fifteen (15) feet from the edge of the drainfield. Monitoring well G-12 was placed in an area of the site upgradient of the OSDS and assumed to be unaffected by septic tank effluent. This well was used to determine the background characteristics of the site. Preliminary data indicated the need for additional wells and as a result wells G-13, G-14, and G-15 were subsequently installed at a distance of fifty (50) feet from the edge of the drainfield. Monitoring wells G-16 and G-17 were installed manually using a stainless steel bucket auger immediately east and west of the graywater drainfield, respectively. These monitoring wells were installed to monitor water levels during a tracer test. Monitoring well locations at the Groseclose site are shown on Figure 7.



SCALE: 1"=30'	DRAWN BY: / DATE: 12/92	MONITORING WELL AND SEEPAGE METER LOCATION MAP	FIGURE:
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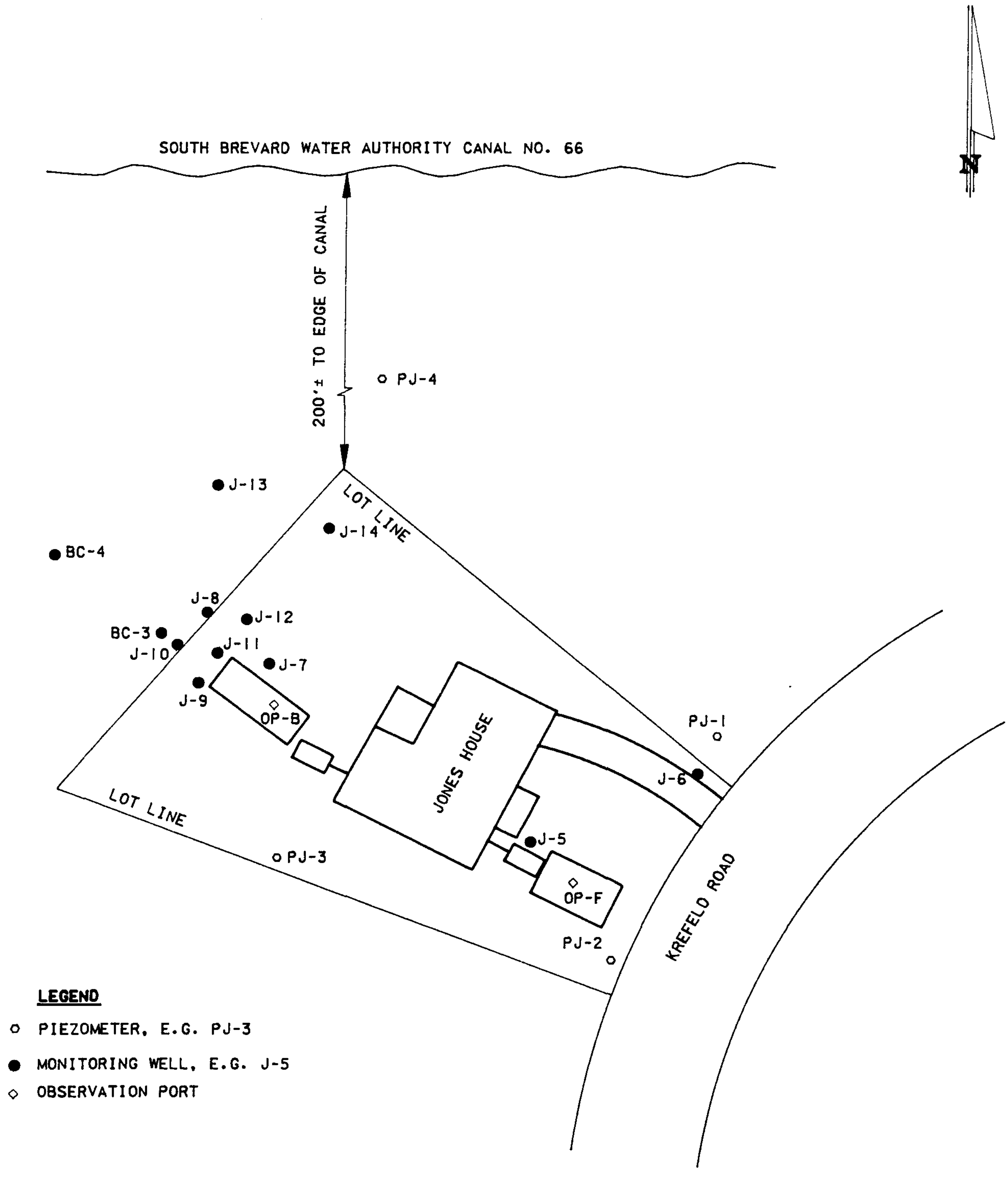
Jones Site Wells: Monitoring well J-6 was placed in the front yard of the house and served as a background indicator of the water quality for the site. Monitoring wells J-7, J-9, and J-11 were placed four (4) feet from the edge of the blackwater drainfield. Monitoring wells J-8 and J-10 were placed fifteen (15) feet from the edge of the blackwater drainfield. Preliminary data collected during the study suggested that additional monitoring wells should be installed. Monitoring well J-12 was added fifteen (15) feet from the drainfield in a slightly different direction than monitoring wells J-8 and J-10. Monitoring well J-13 was added fifty (50) feet from the edge of the drainfield. Monitoring well J-14 was placed forty-five (45) feet north of the drainfield along the row of decorative plants to determine if there was a possible impact from fertilizer use. Monitoring well J-5 was placed downgradient of the graywater drainfield in the front yard. Further monitoring of the impacts of this drainfield was not conducted because of the location of the house downgradient from the drainfield. The locations of the Jones site monitoring wells are shown on Figure 8.

Control Site Wells: Several wells were installed at the control site area to determine groundwater quality characteristics of a relatively undisturbed area of the subdivision. Monitoring well BC-5 was installed in the older pine trees and palmetto palms characteristic of the area while monitoring well CW-0 was installed on the bank of the canal just to the north. Figure 9 shows the location of the control site monitoring wells.

After well installation, the top of casing elevations of each well and piezometer was surveyed by Ayres Associates and referenced to MSL. Monitoring well construction and elevation details are presented in Table 1.



SCALE: 1"=40'	DRAWN BY: DATE: 12/92	MONITORING WELL LOCATION MAP	FIGURE:
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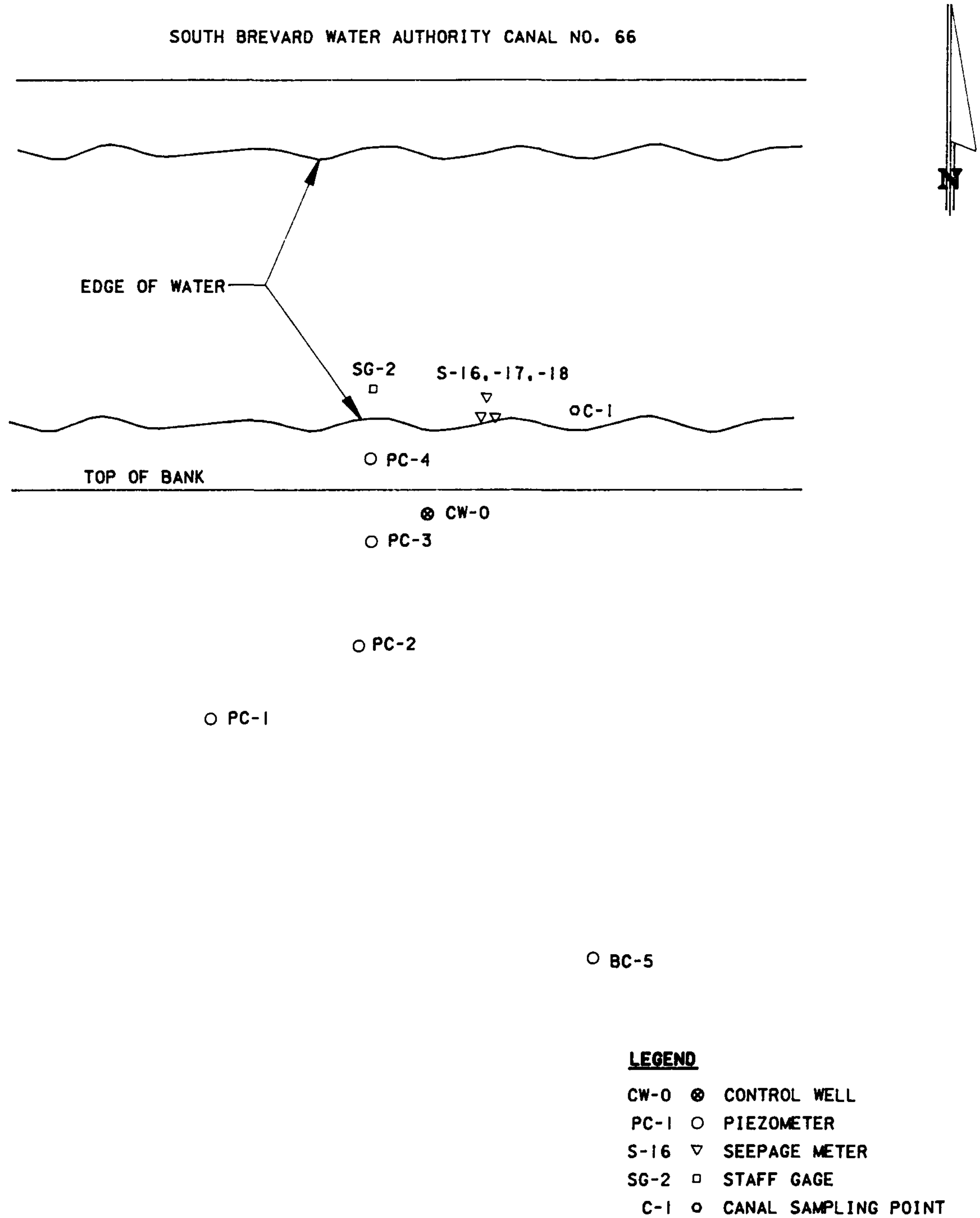


SCALE			
SCALE: 1"=40'	DRAWN BY: / DATE: 12/92	MONITORING WELL LOCATION MAP	FIGURE: 9
AYRES ASSOCIATES	CHECKED BY: / DATE: 2/93 APPROVED BY: / DATE:	CONTROL AREA	

Table 1. Monitoring Well and Piezometer Construction Details

STATION ID	TOC ELEVATION NGVD (ft)	TOTAL DEPTH (ft)	SCREEN INTERVAL (ft)	GROUND ELEVATION NGVD (ft)
---JONES RESIDENCE---				
Piezometers				
PJ-1	26.04	5.69	2.58	26.1
PJ-2	25.97	5.02	1.95	26.2
PJ-3	25.86	6.09	3.04	26.0
PJ-4	27.79	8.35	5.25	25.9
Observation Wells				
J-5	28.07	9.27	8.10	28.2
J-6	26.11	9.13	6.75	26.4
J-7	26.96	9.18	7.00	27.2
J-8	26.08	9.61	7.57	26.3
J-9	26.53	10.12	8.12	27.1
J-10	26.23	8.86	8.15	26.2
J-11	26.59	9.45	7.37	26.9
J-12	26.46	7.83	5.08	26.7
J-13	28.69	10.75	5.04	25.9
J-14	25.81	8.25	5.00	26.0
---GROSECLOSE RESIDENCE---				
Piezometers				
PG-1	27.4	8.55	4.88	27.2
PG-2	26.87	8.50	5.08	26.9
PG-3	27.19	9.70	4.95	27.3
PG-4	27.01	7.85	4.93	27.2
Observation Wells				
G-5	27.46	9.95	7.16	27.5
G-6	27.02	8.25	6.29	27.5
G-7	27.15	7.55	5.05	27.6
G-8	27.10	7.95	5.57	27.6
G-9	26.99	8.90	6.15	27.6
G-10	27.42	9.95	7.20	27.6
G-11	27.45	9.95	7.32	27.5
G-12	27.08	8.90	7.27	27.4
G-13	26.45	7.70	5.00	26.7
G-14	26.10	7.94	4.70	26.5
G-15	26.05	7.70	5.00	26.3
---CONTROL WELLS---				
CW-0	26.81	11.95	9.78	27.6
BC-5	25.36	11.87	9.79	25.8
---STAFF GAGES---				
SG-0	20.68			
SG-2	21.11			
SG-3	20.95			
SG-4	22.88			
SG-5	21.95			
SG-6	22.98			

NGVD - National Geodetic Vertical Datum of 1929
TOC - Top of Casing

Subsurface Characterization Methods

The subsurface characteristics of the study sites were determined during well installation by describing the soil characteristics encountered and by taking representative soil samples for particle size and organic carbon analyses. Also, slug tests were performed at selected monitoring wells to determine the hydraulic conductivity of the water table aquifer at the sites.

Soil Descriptions: The soils at the three (3) study sites were described using standard USDA Soil Conservation Service (SCS) soil classification methods. Mr. Chris Noble, Resource Soil Scientist from the SCS office in Vero Beach, visited the sites with Ayres Associates staff and prepared soil descriptions based on hand augered test holes, landscape position, and other site characteristics. The subsurface lithology at the sites was also recorded during the installation of piezometers and monitoring wells.

Particle Size Analysis: Particle size analysis was conducted on samples obtained from the bucket auger or hollow stem auger flights using a standard sieving technique (U.S. standard sieve series) according to the method outlined by Hakanson and Jansson (1983). With this method, the percentages of the various sand fractions and the silt-clay fraction of the soil samples were determined.

Percent Organic Matter: Organic matter content (loss on ignition) was determined on the soil samples by combusting a weighed amount of dry sediment for two (2) hours at 550°C in a muffle furnace and weighing the remaining ash (APHA, 1989).

Hydraulic Conductivity: Slug tests were performed at selected wells by lowering a 1.5 meter long, 3 cm diameter PVC slug into the well so that it was completely submerged below the water level. Changes in water levels were measured by a pressure transducer connected to a Hermit SE-1000B electronic data logger (In-situ, Inc.). The data logger recorded the changes at log phase time intervals. When the water level in the well returned to its static level, the slug was quickly removed from the well and the water level was then allowed to return to its original position. Changes in water levels during recharge were then recorded by the data logger. The slug test data were analyzed by the method described by Bouwer (1989).

Tracer Tests

Two multi-well, natural gradient, ground-water tracer tests were conducted during this study. In order to conduct these tests, a network of sampling tubes and miniature wellpoints were installed at the Groseclose site. Teflon™ tubing (3/8-inch diameter) was attached to miniature, stainless-steel wellpoints and installed using the "direct push" method as follows. The Teflon™ tubing was inserted in sections of ½-inch diameter, stainless-steel tubing with the miniature wellpoint extending from one end. The stainless-steel tubing was driven into the soil using a hand held, electric reciprocating hammer. After the wellpoint was driven to the desired depth, the stainless-steel tubing was withdrawn leaving the stainless-steel wellpoint/screen assembly and Teflon™ tubing in place. After installation each wellpoint was developed using an electric, peristaltic pump. A total of 24 sampling tubes were installed in the vicinity of the Groseclose residence drainfields. The locations of these sampling tubes (T-0 through T-21) are shown on Figure 19 in the Results section. These wellpoints were installed to a depth of 5.6 feet below ground surface (bgs) or approximately 1 foot beneath the groundwater surface. Wellpoints T-5i (Intermediate) and T-12i were installed to the top of a sandy clay loam layer encountered at a depth of 7.6 feet or approximately 3 feet beneath the groundwater surface.

Bromide (Br^-) was used as the tracer material for the two tests conducted at the Groseclose residence. Bromide was selected for use as the tracer because of its low background concentration (less than 1 mg/L in most aquifers containing potable water [Davis et al. 1980]), its relative stability in the subsurface environment (Schmotzer et al.

1973) and its ease of detection (Davis et al. 1980). The tracer slugs were prepared by mixing granular sodium bromide (NaBr) with distilled water.

Wellpoint samples were analyzed for Br^- using a specific ion electrode (Orion Model 290A equipped with Model 94-35 Bromide Electrode). The Br^- detection limit for this instrument using the analytical procedures employed during this study was 0.2 parts per million (ppm). The meter was calibrated prior to use each day with 10 ppm, 100 ppm, and 1,000 ppm Br^- standards prepared by Southern Analytical Laboratories Inc., Oldsmar, Florida.

Test #1 was begun on March 4, 1992. A two-gallon Br^- tracer slug was poured into IP-1 (Injection Port #1) at 9:30 a.m. The initial Br^- tracer concentration was approximately 102,500 mg/L. Injection Port IP-1 was located adjacent to the graywater septic tank within the boundaries of the graywater septic tank drainfield (Figure 19). The bottom of IP-1 was flush with the bottom of the gravel drainfield at a depth of 2.8 feet bgs. After injecting the slug, Wellpoint T-0 was sampled approximately every hour for the first 12 hours and every 3 hours thereafter. This wellpoint was located adjacent to IP-1 and was monitored in order to measure the slug travel time through the unsaturated zone from the bottom of the drainfield to the groundwater.

After the first 24 hours, samples were also collected from the other wellpoints approximately every 3 hours in order to track the movement of the Br^- tracer slug.

Test #2 was begun on April 8, 1992. A one-gallon Br^- tracer slug was poured into IP-2 (Injection Port #2) at 11:24 a.m. The initial Br^- tracer concentration was again approximately 102,500 mg/L. Injection Port IP-2 was located 5.3 feet south of the northernmost row of wellpoints (Figure 19). The bottom of IP-2 was also flush with the bottom of the graywater drainfield. After injecting the slug, Wellpoint T-16 was monitored in order to measure travel time of the slug through the unsaturated zone below the drainfield. The northern row of wellpoints were sampled to measure Br^- tracer travel time in the groundwater. All samples were collected using an electric, peristaltic pump.

Septic Tank Effluent Characterization

STE Quality: Septic tank effluent from the graywater tank at the Groseclose site was sampled directly on November 20, 1992 when the tank was pumped clean. The septic tank effluent from the black water tank was sampled only for oil and grease on that date because of the presence of a significant layer of scum.

Further sampling of septic tank effluent was accomplished by inserting sampling tubes in the effluent lines of the black water and graywater septic tanks at the Groseclose site. The septic tank effluent from these tanks was sampled on February 12, 1992 and April 8, 1992 to assess the quality of the wastewater discharged to the drainfields.

STE Quantity: The wastewater flow at the residences was estimated by reading water meters over several intervals during the study. The water usage at the Groseclose residence should be relatively representative of wastewater flows because canal water was used for irrigation. The water usage at the Jones site was probably higher than the wastewater flow because water meter readings included water used for lawn irrigation.

Groundwater Sampling

Groundwater samples were obtained from monitoring wells in two separate monitoring phases. Phase I sampling occurred February, 1990 through August, 1990, and Phase II sampling occurred from August, 1991, to March, 1992. Prior to obtaining a groundwater sample for laboratory analysis, each well was pumped using a submersible pump. The pump was cleaned with a dilute chlorine solution and then rinsed three times with distilled water after each sample point. Approximately three (3) to five (5) well volumes of water were pumped from the well prior to sampling to obtain a representative sample of groundwater. After the well was pumped, dissolved oxygen and temperature were measured using a Leeds and Northup Model 7932 dissolved oxygen meter. The meter was standardized in the laboratory and the calibration was field checked before each measurement. Conductivity was measured using a YSI Model 33 SCT meter. These probes were rinsed with the chlorine solution and distilled water before being placed in each well. After these measurements were taken, groundwater samples were obtained using pre-cleaned and disinfected PVC bailers. A separate bailer was used for every well. The bailers were washed with a dilute chlorine solution, rinsed with distilled water and wrapped with aluminum foil at the laboratory. In the field, care was taken to keep used bailers separate from the clean bailers. Samples were collected at the control areas first, and then at the OSDS sites. At each residence (OSDS site), the background sample was collected first and subsequent samples were progressively collected from wells furthermost from the drainfield to the wells next to the drainfield. This was done as a further step to prevent contamination of wells. After the samples were collected, they were placed on ice in coolers and taken back to the laboratory for analysis. At each well, the first sample was poured into a sterile whirlpak bag to be analyzed for fecal bacteria. Water samples were then taken and poured into laboratory prepared containers. Duplicate water samples were collected from each well.

Seepage Meter Installation and Sampling

Twenty-one (21) seepage meters were installed in the canal bottom along the study area to measure groundwater seepage into Canal 66. Seven (7) meters were installed in the canal behind the Groseclose residence; five (5) meters were installed in the canal north of the Jones residence; and nine (9) control meters were installed. These meters were placed at varying distances from the shore. Meters were also placed next to each other to provide replicate seepage measurements.

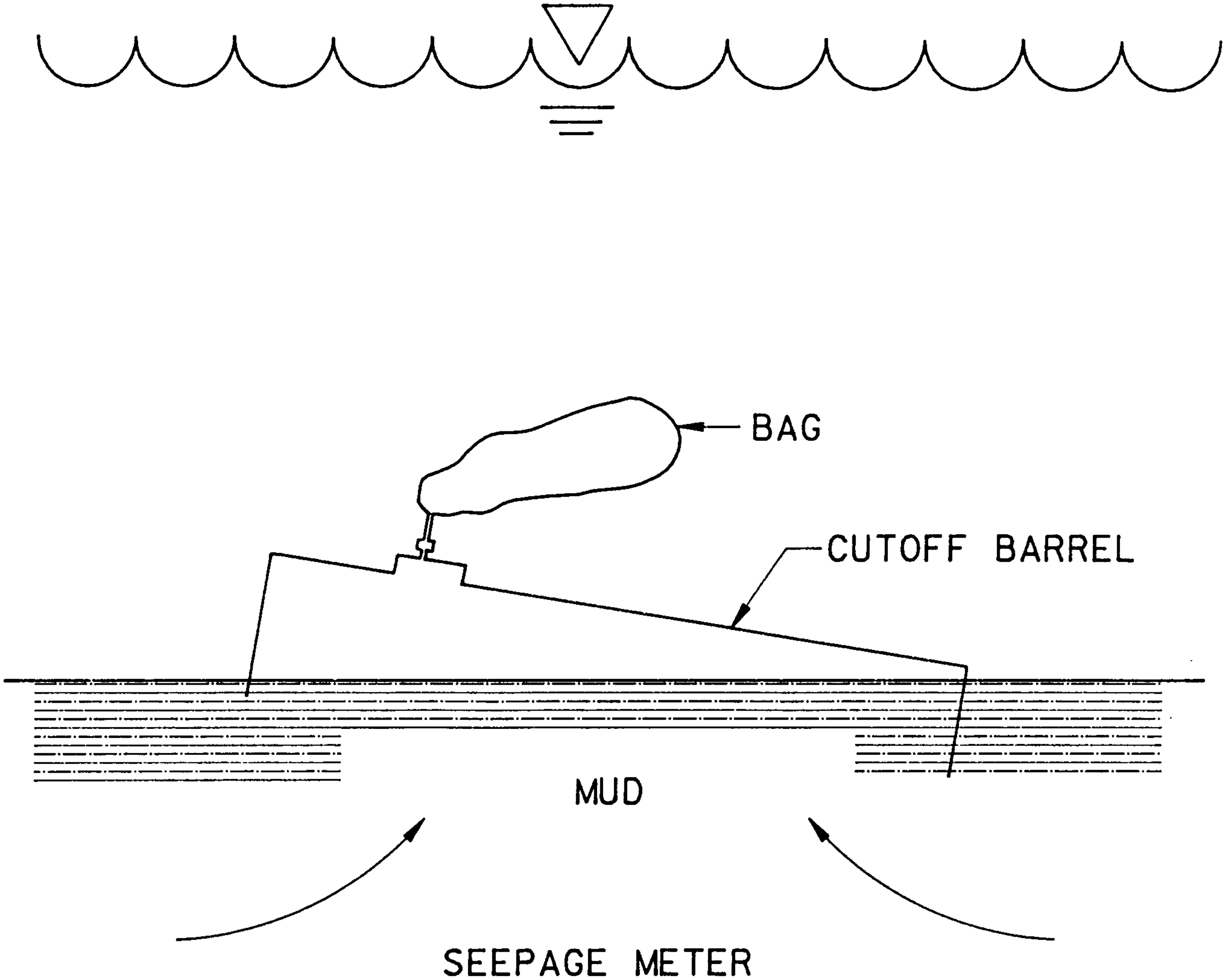
The meters were constructed of steel 55-gallon drums that were cut and inserted into canal sediments (Figure 10). The design of these meters is similar to those described by Lee (1977), Belanger and Conner (1979), and Fellows and Brezonik (1980) for measurement of groundwater seepage into water bodies. A plastic bag and tubing were attached to each meter through a rubber stopper inserted into the bung of the drum. The rate of seepage was calculated by measuring the change in volume of water in the bag over time. The change in water volume was converted to units of liters per m^2-day. Details of meter construction and proper techniques for meter installation and sampling are discussed by Belanger and Mikutel (1985).

The water quality of the seepage water was measured on five (5) occasions during the study. Before these sampling events, the seepage bags were rinsed with acid wash and distilled water to remove any contaminants. This seepage water was poured into acid washed 500 ml bottles and placed on ice for transport back to the laboratory for analysis.

Canal Sampling Point Installation and Sampling

Six canal surface water sampling locations, C-0 through C-5, were selected along a 1.5 mile length of the canal adjacent to the subdivision area, Canal 66 (Figure 3). Along this stretch of the canal there were many different land uses. The south side was dominated by homes along the entire length except for the wooded control area. The north side of the canal has wooded areas, an orange grove, and a horse stable along its banks. In addition, four other downstream canal sites, NC-1 through NC-4, in the drainage system between Canal 66 and the Turkey Creek discharge to the Indian River Lagoon were sampled quarterly for bacterial analysis. Figure 2 shows the downstream canal sampling points.

Piezometers and littoral interstitial porewater (LIP) samplers were also placed in the canal bottom near seepage meters at several locations. These monitoring devices were used for a comparison of water quality data obtained from the seepage meters. The LIP samplers are essentially mini-wells used to collect interstitial pore water for chemical analyses. The LIP samplers were constructed following a general design suggested by



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the U.S. Geological Survey (Winter et al. 1988). The samplers consisted of a 1.0 meter-long, pointed stainless steel tube (9.53 mm diameter fitted with a protective outer tube (12.7 mm diameter). The unit was pushed into the sediments to a depth of approximately 0.5 m, and then the outer tube pulled up to reveal the sampling ports in the interior sampling tube. A 250 μm brass screen covering the sampling ports protected these ports from clogging by sand or grit. Once the sampler was in place, a pore water sample was obtained by gentle suction on the sampling tube, drawing the liquid into a glass flask.

Analytical Procedures

After arrival at the laboratory, the water samples were placed in a refrigerator at 4°C. The samples were analyzed for total kjeldahl nitrogen (TKN), nitrate-nitrite nitrogen (NO_3, NO_2-N), total phosphorus (TP), chemical oxygen demand (COD), pH, color, and turbidity within 5 days. Nutrient parameters were analyzed first. All colorimetric analysis was performed using a Shimadzu 160A spectrophotometer.

Total Kjeldahl Nitrogen (TKN): TKN was determined using acid digestion followed by the indophenol method for ammonia determination, as described in the 17th Edition of Standard Methods (APHA, 1989). A standard curve, EPA reference sample, spike, and triplicate analyses were run with each set of samples.

Nitrate-Nitrite: Nitrate-nitrite was determined following the cadmium reduction method outlined by Jones (1984). Nitrate is reduced to nitrite in the presence of cadmium. The nitrite produced is determined by diazotizing with sulfanilamide and coupling with N-(1 naphthyl)-ethylenediamine dihydrochloride to form a highly colored azo dye which can be measured colorimetrically. This procedure allows for processing of large numbers of samples in a short time period. A standard curve, EPA reference sample, spike, and triplicate analyses were run with each set of samples. Interference by color was removed by precipitation with aluminum hydroxide.

Total Phosphorus: Total phosphorus was measured using persulfate digestion followed by the ascorbic acid method for determination of reactive phosphorus. The method is described in Standard Methods (APHA, 1989). A standard curve, EPA reference sample, spike and triplicate analyses were run with every set of samples.

Chemical Oxygen Demand (COD): COD was determined using the EPA-approved HACH semi-micro COD monitoring system (HACH, Loveland, Co.). This test measures the oxygen equivalent of the materials present in the sample subject to a strong chemical oxidant. This test uses dichromate as the oxidant. The HACH system allows for a fast determination with small amounts of sample. Mercuric sulfate is present to reduce interference from the oxidation of chloride ions. A reference sample was run with each set of samples. The color change was measured using a HACH DR 100 colorimeter.

pH: The pH of the water samples was measured using an Orion Research Model 601A pH meter. The meter was calibrated before each use with pH 4 and pH 7 buffers.

Color: Color was determined by filtering the samples through 0.45μm filters (Gellman GN-6), followed by spectrophotometric analysis at 540 nm. A potassium chloroplatinate standard curve was prepared once and used for all samples.

Turbidity: Turbidity was measured nephelometrically using a HACH 2100A turbidimeter (HACH company). The turbidimeter was calibrated frequently during each sampling run with prepared standards.

Bacteria: Groundwater and canal bacterial samples were collected with sterile Whirl Pak bags. Once in the laboratory, samples were kept chilled until ready to be filtered. All fecal coliform samples were filtered and incubated within six hours after collection. Fecal streptococcus samples were filtered and incubated within ten hours after collection. Millipore ampouled M-FC media containing 1% rosolic acid was used to culture fecal coliforms, with *Escherichia coli* being the major species in this group. This media is appropriate for recovering organisms from non-chlorinated effluent, or where interference from background growth is possible. The filtering procedure followed is described in the 17th edition of *Standard Methods for the Examination of Water and Wastewater* (APHA, 1989). Sample dilutions of 1 and 10 ml per 100 ml filtered volume were used to achieve the desired 20 to 60 colonies per filter. Sample volumes of greater than 10 ml were not filtered due to turbidity of some well samples. Incubation occurred in a Precision^R circulating water bath incubator which held a temperature of 44.5±0.2°C. After 24 hours, samples were removed and colonies were counted and converted to numbers per 100 ml of sample.

KF-Streptococcus Agar was used to recover fecal streptococcus organisms. This agar has been reported to be highly efficient in the recovery of fecal streptococci, specifically *S. faecalis, S. faecalis* subsp. *liquefaciens, S. faecalis* subsp. *zymogenes, S. faecium, S. bovis,* and *S. equinus*. The filtering procedure described in Standard Methods was used, with dilutions of 0.1, 1, and 10 ml of sample per 100 ml resulting in the desired 20 to 100 colonies per filter. Culture plates were incubated in an air incubator at 35 ± 0.5 °C for 48 hours. Colonies were counted and converted to numbers per 100 ml of sample. A dissecting type microscope (Olympus Stereoscopic) with magnification of 10-15X was used to count colony-forming units (usually referred to simply as "colonies"). Bacterial densities were reported as organisms per 100 ml.

If the filters contained no colonies, values were reported as less than the calculated value per 100 ml, based upon the largest single volume filtered. For example, a filter containing no colonies on a sample dilution of 10:100 would be reported as <10 colonies/100 ml. Conversely, for filters containing colonies too numerous to count, values were reported as greater than the upper recommended counting limit for that dilution. For serial dilution filters that yielded densities outside the desired range, an average of the high and low counts was taken to report bacterial densities for that particular sample.

Quality Control Testing--Bacteria

Same day duplicate analyses of some samples were performed by personnel at the Department of HRS Indian River County Public Health Unit laboratory several times during the course of the study for comparison of results. In general, the comparison between the two analyses were reasonable, especially with respect to fecal coliform values. Unfortunately, fecal streptococcus data from July 23, 1990 could not be compared due to failure of the media to support satisfactory growth in the Department of HRS laboratory analysis. Data from the quality control testing performed at the Department of HRS lab are presented in Appendix C. Bacteria were analyzed by F.I.T. in 1990, Pembroke Lab in August and September, 1991, and Aquatic Labs, Inc. in December 1991, and January and March, 1992. Duplicate analysis of 5% of the samples was completed. Fecal coliform and fecal streptococcus colonies were verified on selected samples by picking ten isolated colonies from membranes and transferring to appropriate media, according to verification procedures detailed in Standard Methods.

Statistical Analysis of Data

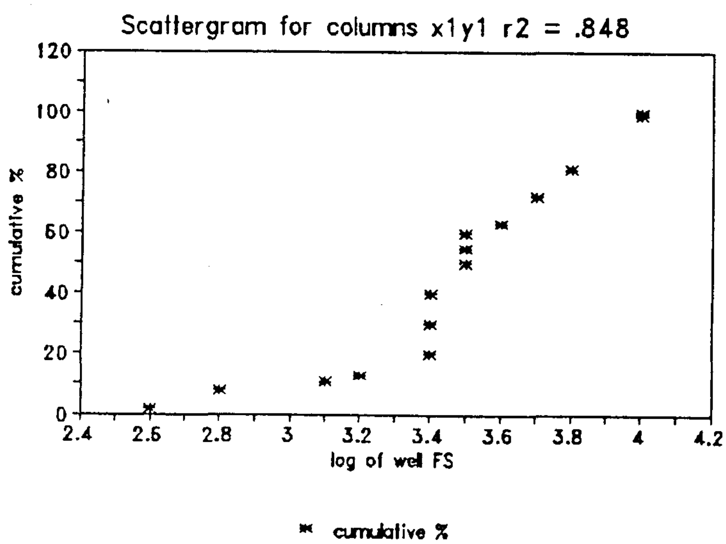
ANOVA Testing: ANOVA (analysis of variance) tests were run for various parameters between individual wells and various groups of wells to determine if there were significant differences in the means. All statistical significance in this report was based on a 95% confidence level, or probability level of 0.05. Comparisons of wells and well groups located varying distances from the drainfield were completed for each residence, in addition to comparisons between residences and between test wells and control wells.

The Fisher protected least significant difference (PLSD) test (Fisher, 1949) was used as one of the analysis tools. This test is based on the outcome of the omnibus F test; the significance or nonsignificance of this test determines whether additional statistical analysis is necessary. The Fisher PLSD test is most appropriate in situations in which initially, at least, all treatments are given equal consideration (i.e. there are no favored treatments or anticipated outcomes). The Fisher test has been studied by statisticians and shown to offer an excellent balancing of Type I and Type II errors (see Vieppel & Zedeck, 1989).

It is generally assumed that bacteriological data sets are log-normal in their distribution, and consequently must undergo log transformations prior to the application of parametric statistics. In order to verify that this study's data were distributed log-normally, several tests were performed. After bacteriological data were log-transformed, normal probability plots for fecal coliform and fecal streptococcus were developed. A probability plot could not be generated for fecal coliform data in the monitoring wells, since wells often contained <1 or <10 colonies/100 ml of sample. Probability plots for Phase I data are shown in Figures 11 and 12.

The data indicate a reasonable approximation of a straight line, indicating that the distribution is log-normal. Likewise, the logarithms of the bacterial data are normally distributed, indicating the data are positively skewed and have a log-normal distribution. The best estimate of central tendency of log-normal data is the geometric mean, and this statistic was computed and used in statistical tests. The geometric mean is equal to the antilog of the arithmetic mean of the logarithms.

Linear and Multiple Regression Analysis: Linear and multiple regression analysis was used in this study for statistical treatment of water quality data. Regression analyses are used to analyze data whenever a quantitative variable (the dependent variable) is to be studied as a function of, or in relationship to, any factor of interest (expressed as the independent variable). Thus, regression analysis allows an unknown variable to be predicted from a known variable, and allows the accuracy of this prediction to be determined (Cohen and Cohen, 1983). Regression analyses were performed in this



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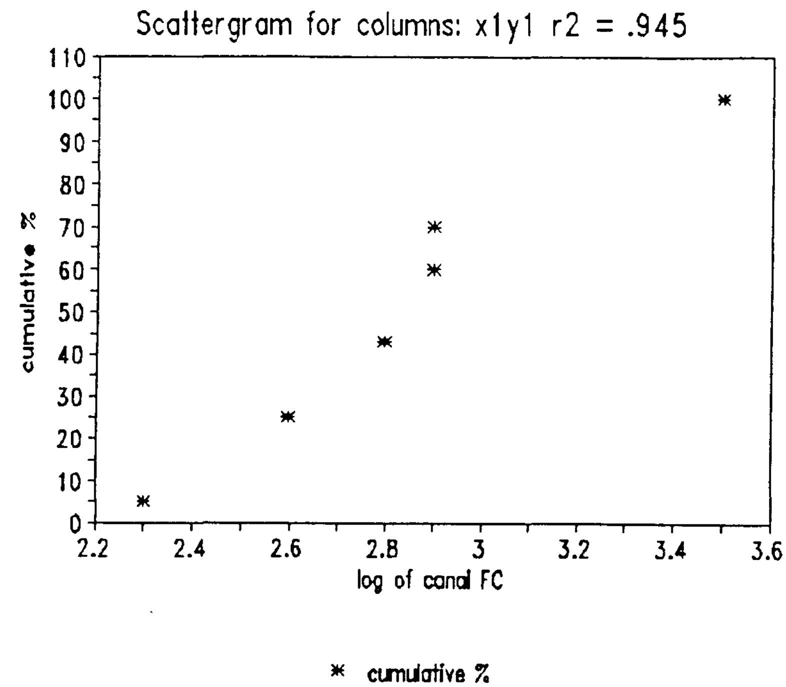
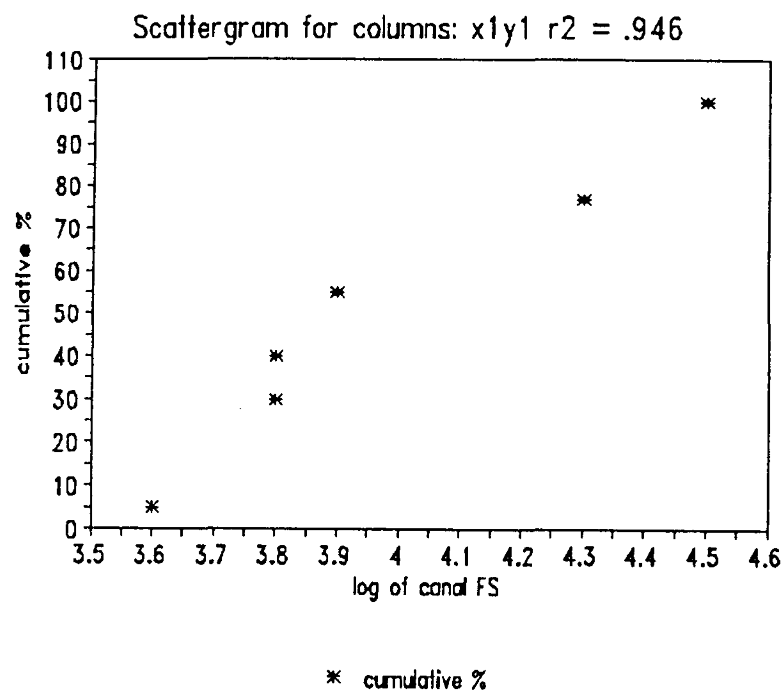
NORMAL PROBABILITY PLOT OF FECAL STREPTOCOCCUS IN MONITORING WELLS

FIGURE: 11





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NORMAL PROBABILITY PLOTS OF FECAL COLIFORM AND FECAL STREPTOCOCCUS IN CANALS

FIGURE: 12

research to determine the relationship between canal levels of fecal coliform, fecal streptococcus, nitrate, total phosphorus, TKN, and conductivity with the following factors:

1) Stage height (NGVD)

2) One day rain total prior to sampling (inches)

3) Seven day rain total prior to sampling (inches)

4) Temperature (°C)

Quality Control Testing -- Chemical Parameters

The quality of data reported is an important aspect which must be taken into consideration when analyzing data. In analyzing data for this study, several procedures were performed to ensure the quality of the data. An EPA reference sample was used for accuracy. Duplicate field samples and triplicate lab analysis were used for precision.

Finally, the method of known additions was used for percent recovery determination in in Phase I. The following subsections describe the quality control procedures used for the nutrient analyses in this study.

Total Phosphorus (TP): On every sampling date a new standard curve was prepared with concentrations of 0.01, 0.05, 0.10, 0.50, and 1.00 mg/l. A reagent blank was also analyzed each time to zero the spectrophotometer. Reagent blanks were also periodically run to check the spectrophotometer for drift. A duplicate field sample was collected from every well in a separate container. Analysis of duplicates produced results which deviated by less than 10%. When a deviation of greater than 10% was present, the sample was analyzed again until the deviation fell within the 10% limit. A spike of known concentration was added to at least one sample during every sampling set. Recovery of spike concentrations ranged from 77.8 to 116.1% (Table 2).

Table 2. Total Phosphorus (TP) Quality Assurance Data

Sample Date	EPA Reference Concentration	FIT Concentration (mg/L)	Percent Error (%)	Spike Recovery (% of known addition)
07-Feb-90	0.30	0.315	5.03	85.3
28-Feb-90	0.30	0.289	3.77	89.8
15-Mar-90	0.30	0.320	6.63	101.2
04-Apr-90	0.30	0.313	4.37	77.8
25-Apr-90	0.30	0.303	1.13	116.1
09-May-90	0.30	0.309	2.90	87.8
31-May-90	0.30	0.310	3.20	114.2
27-Jun-90	0.30	0.325	8.26	116.1
23-Jul-90	0.30	0.307	2.27	102.4
07-Aug-90	0.30	0.307	2.23	86.5
20-Aug-91	0.15	0.153	2.00	-----
20-Sep-91	0.10	0.103	3.00	-----
19-Dec-91	0.10	0.105	5.00	-----
08-Jan-92	0.10	0.095	5.00	-----
03-Mar-92	0.93	0.923	0.75	-----
26-Mar-92	0.76	0.800	5.20	-----

This is considered to be acceptable by most laboratories. Except for two sampling times (April 4 and May 31, 1990), triplicate analysis of samples yielded standard deviations which were less than 10% of concentration value. The percent error of EPA reference samples ranged from 0.75% to 8.26%, but was greater than 5% on only four occasions.

Nitrate, Nitrite-Nitrogen (NO_3,NO_2-N): The same procedures for precision and accuracy as described for TP were used for NO_3,NO_2-N. The standard curve was constructed from concentrations of 0.01, 0.05, 0.10, 0.50, and 1.00 mg/l. Spike recovery ranged from 81.7 to 100.9% of a known addition in Phase I (Table 3). The standard deviation of triplicate analyses were less than 10% of the concentration value. Percent error of EPA reference samples ranged from 0.00 to 9.65%. This compares favorably to the precision and accuracy data in Standard Methods (APHA, 1989) for the cadmium reduction method.

Table 3. Nitrate, Nitrite-Nitrogen (NO_3,NO_2-N) Quality Assurance Data

Sample Date	EPA Reference Concentration	FIT Concentration (mg/L)	Percent Error (%)	Spike Recovery (% of known addition)
07-Feb-90	0.200	0.216	7.75	91.5
28-Feb-90	0.200	0.197	1.40	92.9
15-Mar-90	0.200	0.184	7.85	81.7
04-Apr-90	0.200	0.219	9.65	91.7
25-Apr-90	0.200	0.183	8.55	100.9
09-May-90	0.200	0.186	7.05	90.1
31-May-90	0.200	0.203	1.70	89.6
27-Jun-90	0.200	0.196	2.00	90.4
23-Jul-90	0.200	0.210	5.15	100.5
07-Aug-90	0.200	0.211	5.40	96.0
20-Sep-91	0.400	0.420	5.00	-----
19-Dec-91	0.744	0.738	0.81	-----
08-Jan-92	0.744	0.726	2.42	-----
03-Mar-92	0.550	0.549	0.00	-----
26-Mar-92	0.550	0.543	1.27	-----

Total Kjeldahl Nitrogen (TKN): The same procedures for precision and accuracy as described previously were also used for TKN analysis. The standard curve was constructed from analyzing concentrations of 0.05, 0.10, 0.50, 1.00, 2.00, 3.00, and 5.00 mg/l. Spike recovery ranged from 79.6 to 109.2% of a known addition (Table 4). The standard deviation of triplicate analyses were less than 107% of the concentration value. Percent error of EPA reference samples ranged from 0.16 to 8.20%.

Table 4. Total Kjeldahl Nitrogen (TKN) Quality Assurance Data

Sample Date	EPA Reference Concentration	FIT Concentration (mg/L)	Percent Error (%)	Spike Recovery (% of known addition)
07-Feb-90	1.000	0.995	0.52	95.4
28-Feb-90	1.000	0.998	0.18	92.0
15-Mar-90	1.000	1.082	8.20	109.2
04-Apr-90	1.000	1.049	4.88	80.2
25-Apr-90	1.000	1.009	0.88	103.5
09-May-90	1.000	1.081	8.08	79.6
31-May-90	1.000	0.989	1.07	86.6
27-Jun-90	1.000	0.953	4.72	84.9
23-Jul-90	1.000	0.998	0.16	82.9
07-Aug-90	1.000	1.064	6.44	84.2
20-Aug-91	0.500	0.515	3.00	-----
20-Sep-91	0.250	0.270	8.00	-----
19-Dec-91	0.616	0.641	4.06	-----
08-Jan-92	0.616	0.625	1.46	-----
03-Mar-92	0.834	0.798	4.32	-----
31-Mar-92	0.834	0.847	1.56	-----

IV. RESULTS

Subsurface Characterization

Soil Descriptions: The soil profile at the Groseclose site was described by Chris Noble and David Prewitt of the USDA Soil Conservation Service (SCS) based on soil borings installed at the site (Noble, 1991). The soils were characterized as Oldsmar sand, a nearly level, poorly drained sandy soil that is found in the South Florida Flatwoods and on low knolls on the floodplains. The typical soil profile at the site was described as follows:

> A horizon--0 to 3 inches; very dark brown sand; weak, fine granular structure; very friable; common very fine and fine roots throughout; mixture of organic matter and uncoated sand grains has a salt-and-pepper appearance; abrupt smooth boundary.
>
> E1 horizon--3 to 16 inches; gray (10YR 5/1) sand; single grain; loose; common very fine and fine roots throughout; common dark streaks along root channels; gradual wavy boundary.
>
> E2 horizon--16 to 33 inches; light gray (10YR 7/1) sand; single grain; loose; few fine roots throughout; common dark streaks along root channels; abrupt wavy boundary.
>
> Bh1 horizon--33 to 42 inches; black (7.5YR 2/1) sand; weak fine subangular blocky structure; friable, uncemented; sands are well coated with organic material; gradual wavy boundary.

Bh2 horizon--42 to 48 inches; dark reddish brown (5YR 2/2) sand; weak fine subangular blocky structure; friable, uncemented; sand grains are well coated with organic matter; abrupt wavy boundary.

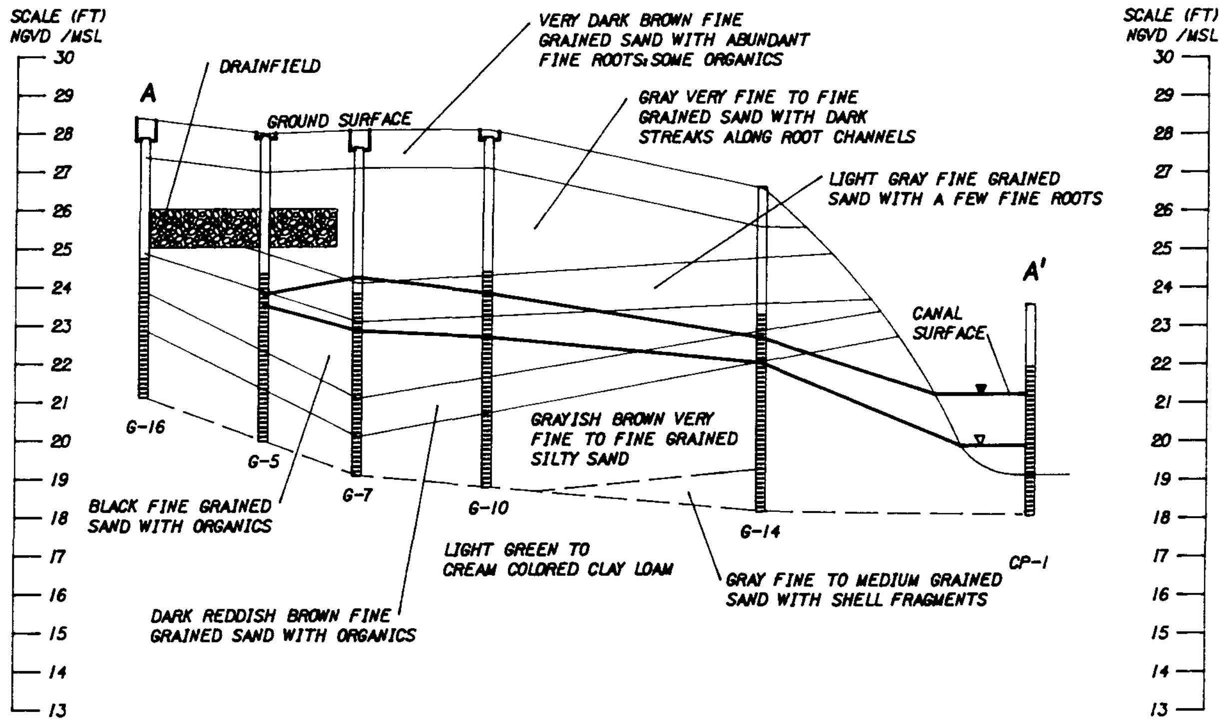
Btg horizon--48 to 63 inches; grayish brown (2.5Y 5/2) sandy loam; weak medium subangular blocky structure; friable; abrupt irregular boundary.

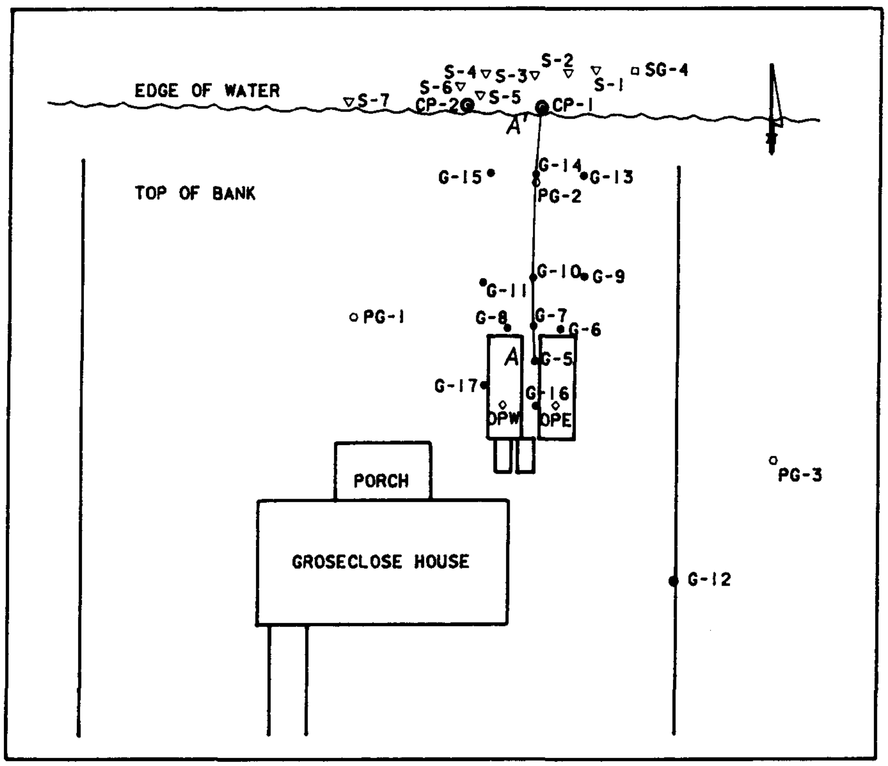
Cg horizon--63 to 80 inches; gray (5Y 5/1) sand; single grain; loose; 60% shell fragments 1 to 5 mm in size.

The subsurface lithology of the site was also evaluated during the installation of piezometers and groundwater wells at the site. Although the depths of the various soil layers varied slightly across the site, this evaluation generally agreed with the profile described above down to approximately 66 to 72 inches (5.5 - 6.0 feet) below ground surface (bgs). Below 66 to 72 inches, a light green to cream-colored sandy clay loam was encountered during the installation of some of the soil borings and miniature wellpoints. Small, whole shells and shell fragments were observed as inclusions in the sandy clay loam which extended to at least 9 feet bgs. Figure 13 is a cross section of the site depicting the soils and lithology described.

Chris Noble and David Prewitt of the SCS characterized the soils at the Jones site as Myakka sand based on their soil borings. Myakka sand is a nearly level, poorly drained sandy soil typically encountered in the South Florida flatwoods. The typical soil profile at the site was described as follows:

A horizon--0 to 3 inches; black (10YR 2/1) sand; weak fine granular structure; very friable; common very fine and fine roots throughout; mixture of organic matter and uncoated sand grains has a salt-and-pepper appearance; abrupt smooth boundary.





LEGEND

- – – – TOTAL DEPTH OF BORING
- ▼ MAXIMUM OBSERVED WATER TABLE ELEVATION
- ▽ MINIMUM OBSERVED WATER TABLE ELEVATION

TRANSECT A - A'

HORIZONTAL SCALE = 1"=20'
VERTICAL SCALE = 1"=4'

AYRES ASSOCIATES	DRAWN BY: LWF / DATE: DKD; CHECKED BY: Mew / DATE: 2/93; APPROVED BY: / DATE:	GROSECLOSE RESIDENCE GEOLOGIC CROSS-SECTION	FIGURE: 13

FILE: d:\dan\2987\figure16.dgn 25-F 14:5

E1 horizon--3 to 20 inches; gray (10YR 5/1) sand; single grain; loose; common medium roots throughout; gradual wavy boundary.

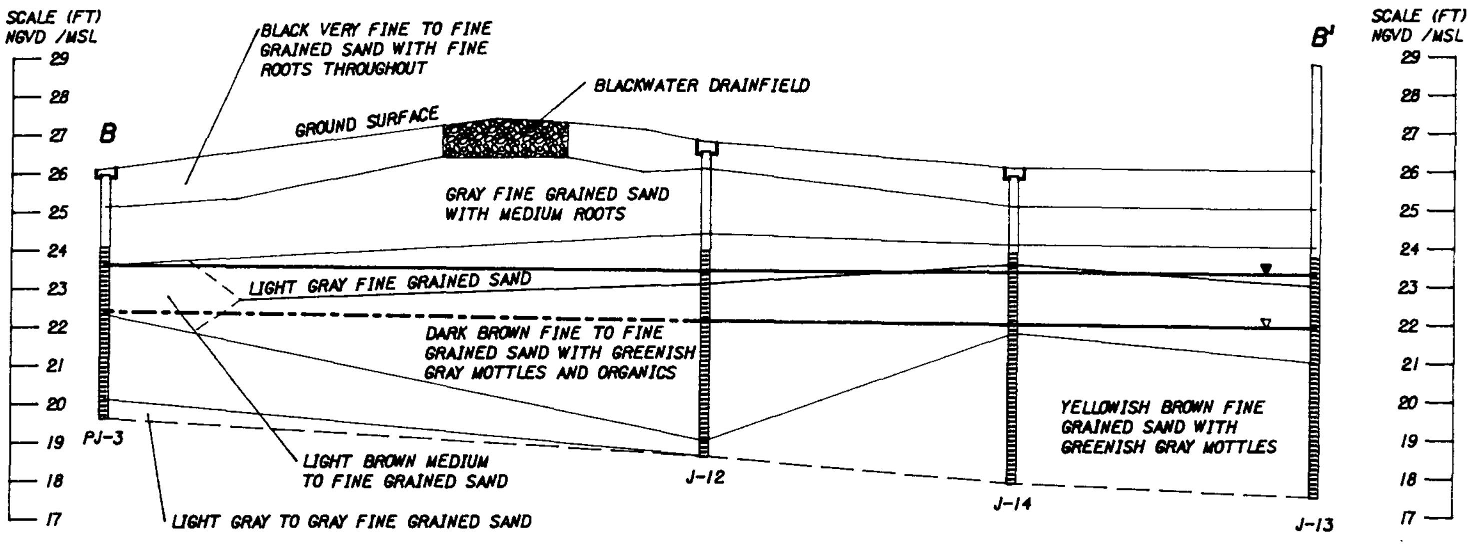
E2 horizon--20 to 29 inches; light gray (10YR 7/1) sand; single grain; loose; few fine roots throughout; sand stripping; abrupt wavy boundary.

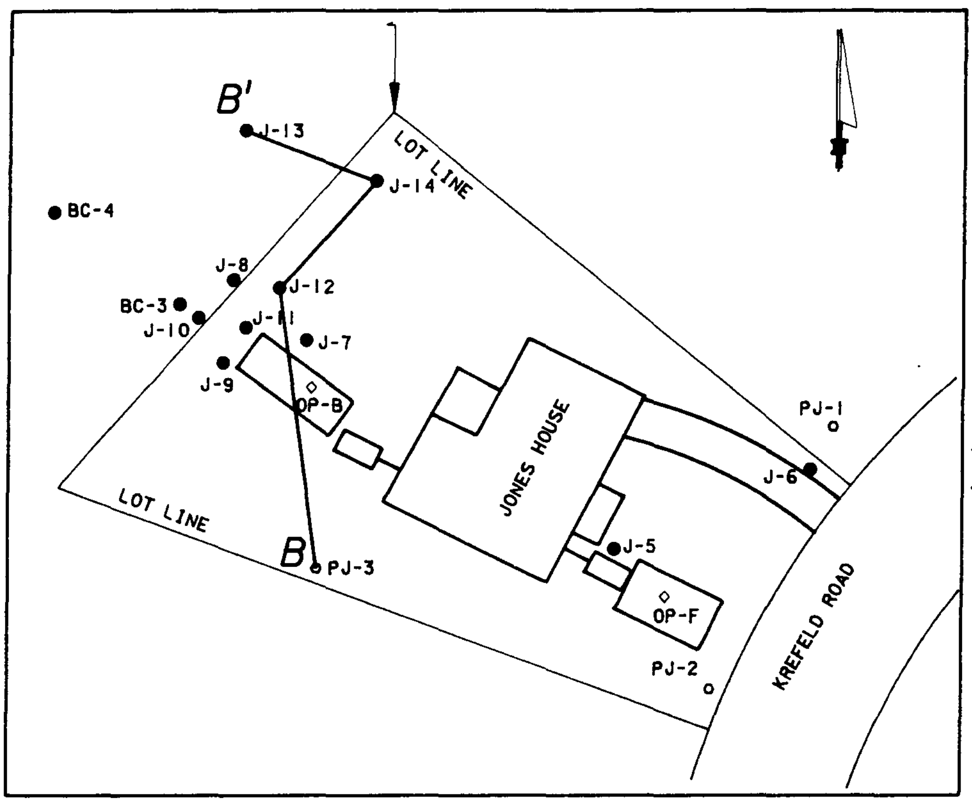
Bh horizon--29 to 35 inches; dark brown (7.5YR 3/2) sand; common medium prominent greenish gray (5G 5/1) mottles; weak fine subangular blocky structure; very friable, uncemented; sand grains are well coated with organic matter; gradual wavy boundary.

BC horizon--35 to 60 inches; yellowish brown (10YR 5/4) sand; single grain; loose; common medium prominent greenish gray (5G 5/1) mottles below 40 inches; clear wavy boundary.

Cg horizon--60 to 80 inches; light gray to gray (5Y 6/1) sand; single grain; loose, non sticky, non plastic.

The lithology of the site from ground surface to approximately 9.0 to 10.0 feet below ground surface was also evaluated during the installation of piezometers and monitoring wells at the site. This evaluation generally agreed with the profile description above although depths of the various soil layers varied slightly across the site. The light gray sand continued to a depth of at least 9.0 feet below ground surface at the site. Figure 14 depicts the soils and lithology encountered at the Jones site.





LEGEND

— — TOTAL DEPTH OF BORING
-·-·- INFERRED WATER TABLE ELEVATION
▼ MAXIMUM OBSERVED WATER TABLE ELEVATION
▽ MINIMUM OBSERVED WATER TABLE ELEVATION

TRANSECT B - B'

HORIZONTAL SCALE = 1"=20'
VERTICAL SCALE = 1"=4'

AYRES ASSOCIATES	DRAWN BY:	DATE:	JONES RESIDENCE GEOLOGIC CROSS-SECTION	FIGURE: 14
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	APPROVED BY:	DATE:		

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Soils at the control site were described by SCS staff as EauGallie sand. EauGallie sand is a nearly level, poorly drained soil found on broad, low ridges in the flatwoods. The description of the typical soil profile was as follows:

A horizon--0 to 3 inches; black (10YR 2/1) sand; weak fine granular structure; very friable; common fine and medium roots throughout; less than 10% organic matter accumulation; abrupt smooth boundary.

E1 horizon--3 to 20 inches; gray (10YR 5/1) sand; single grain; common medium roots throughout; common dark streaks along root channels; gradual wavy boundary.

E2 horizon--20 to 29 inches; light gray (10YR 7/1) sand; single grain; loose; few fine roots throughout; common dark streaks along root channels; abrupt wavy boundary.

Bh horizon--29 to 43 inches; very dark gray (10YR 3/1) sand; weak fine subangular blocky structure; very friable, uncemented; sand grains are well coated with organic matter; clear wavy boundary.

Bw horizon--43 to 52 inches; brown (10YR 5/3) sand; single grain; loose; abrupt wavy boundary.

Btg horizon--52 to 75 inches; light gray to gray (5Y 6/1) loamy sand; weak fine subangular blocky structure; friable; abrupt wavy boundary.

C horizon--75 to 80 inches; gray (5Y 5/1) sand; single grain; loose; 60% shell fragments 1 to 5 mm in size.

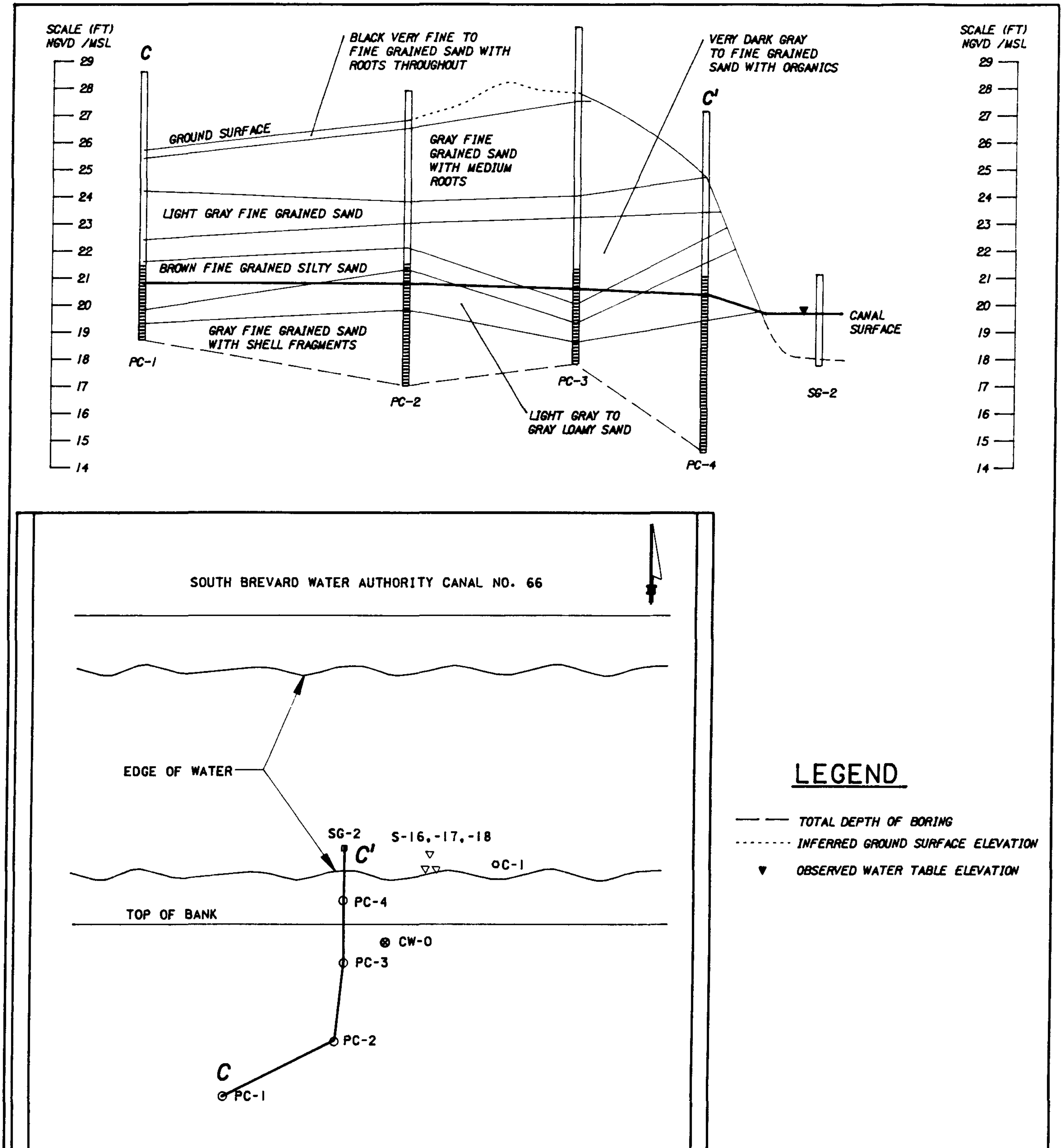
The lithology of the control site was also evaluated during the installation of piezometers and monitoring wells. This evaluation generally agreed with the above profile, and Figure 15 shows a geologic cross section of the control site.

Particle Size Analysis: A particle size analysis of the soil at the Groseclose site showed the soil consisted mainly of medium to fine sand (Appendix A). However, an increasing percentage of fines (silts and clays) were encountered in soil samples obtained from the soil borings installed closer to the canal. Fine to very fine sand percentages ranged from approximately 40% to 80% in the sediments obtained from soil borings G-5 through G-9. Fine to very fine sand percentages ranged from approximately 40% to 70% in the soil samples obtained from soil borings G-13 through G-15, which were closest to the canal. The amount of fines (silts and clays) in these samples ranged from approximately 15% to 45%. Percentages of medium sand were generally less than 10%.

Particle size analysis of soil samples at the Jones site showed the soils there were also mainly fine to very fine sand (Appendix A). With the exception of sediments encountered from approximately 5 to 8.5 feet bgs in soil boring J-11, 80 to 90% of the sediments encountered at the Jones site were fine to very fine-grained sands. The sediments encountered at 5 to 8.5 feet bgs in soil boring J-11 were 50% to 60% fine to very fine-grained sand and approximately 40% medium-grained sand. Silt and clay percentages of the sediments at the Jones site were generally less than 10%.

Soil samples for particle size were not obtained for the control site.

Organic Carbon Analysis: Organic matter content of the soil samples obtained at the Groseclose residence generally ranged from 1% to 2.48%. The highest organic matter content (2.48%) was noted in the soil sample obtained at 4.0 feet in soil boring G-5. The



TRANSECT C - C'

HORIZONTAL SCALE = 1"=20'
VERTICAL SCALE = 1"=4'

AYRES ASSOCIATES	DRAWN BY:	DATE:	CONTROL CROSS-SECTION	FIGURE: 15
	CHECKED BY:	DATE:		
	APPROVED BY:	DATE:		

highest percentages of organic content appeared to be associated with the dark reddish brown to black layers encountered at approximately 3.0 to 4.0 feet bgs. These sandy soils were noted to be well coated with organic material during the subsurface investigation.

Organic matter content of the soil samples obtained at the Jones residence was generally less than 1%. The highest organic content (8.70%) was encountered in the soil sample obtained from 2.0 - 3.0 feet in the soil boring J-11. These sands were also noted to be well coated with organic matter during the site investigation.

Soil samples were not obtained for organic content analysis at the control site.

Hydraulic Conductivity: Hydraulic conductivities were determined for each site using slug test and water table elevation data. Hydraulic conductivities ranged from 0.47 ft/day to 4.74 ft/day at the Groseclose site, with an average hydraulic conductivity of 1.40 ft/day. The highest hydraulic conductivities were calculated from slug tests conducted at monitoring wells G-14 and G-15. These monitoring wells are located adjacent to the canal. See Table 5 for summary of hydraulic conductivities calculated at the sites.

Table 5. Hydraulic Conductivities of Selected Monitoring Wells, K (ft./day)

Control Area		Jones Residence		Groseclose Residence	
Site	ft/day	Site	ft/day	Site	ft/day
CWO	0.95	J7	4.18	G6	1.14
		J9	3.42	G8A	0.82
		J10	1.22	G8B	0.51
		J11	1.47	G9	1.19
		J12	3.94	G10	0.67
		J13	1.55	G11	0.47
		J14	1.46	G12	0.55
				G13	1.11
				G14	4.74
				G15	2.84
Average	0.95		2.46		1.40

Hydraulic conductivities ranged from 1.22 to 4.18 ft/day at the Jones residence with an average hydraulic conductivity of 2.46 ft/day.

Hydraulic conductivity at the control site was determined from a single slug test conducted at monitoring well CW0. The hydraulic conductivity calculated from this test was 0.95 ft/day.

These results indicated that the average hydraulic conductivity calculated at the Jones site were substantially higher than at the Groseclose or control sites.

In summary, subsurface characterization studies at the site indicate that:

1) The percentage of fines (silts and clays) was higher at the Groseclose site than the Jones site. In addition, the percentage of fines at the Groseclose site was greater in soil samples obtained closer to the canal.

2) A light green to cream sandy clay loam was encountered at depths ranging from approximately 5 to 7 feet below ground surface at the Groseclose site, and probably limited vertical groundwater movement.

3) The percentage of organic matter was slightly higher in soil samples obtained at the Groseclose site.

4) Hydraulic conductivities calculated for the Jones soils were generally higher than those calculated at the Groseclose site.

5) Hydraulic conductivities were higher in soils closer to the canal at the Groseclose site.

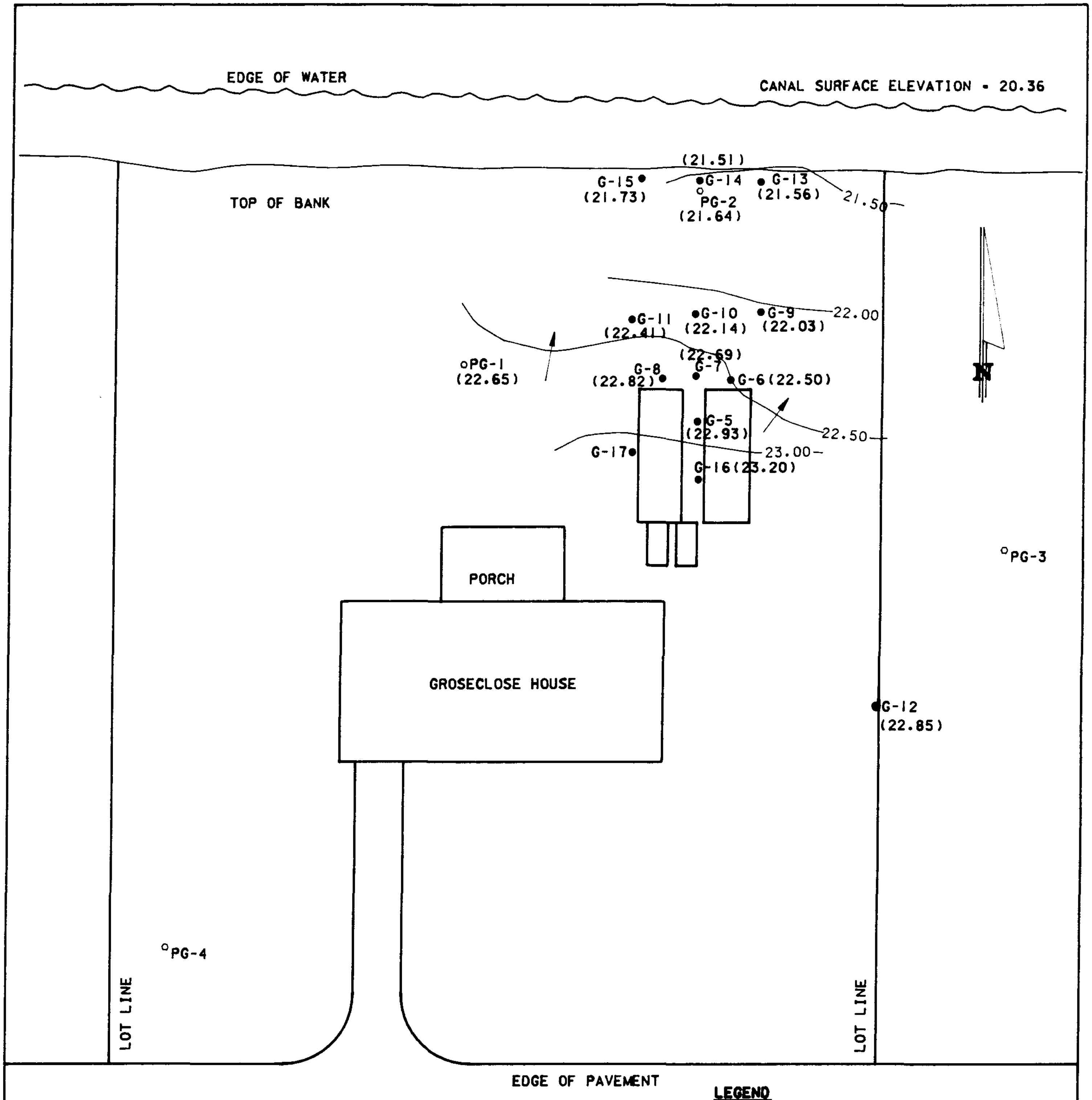
Groundwater Elevations and Flow Direction

Groundwater Flow Direction: Relative elevations of the monitoring well casings and depth to groundwater measurements were obtained to calculate relative groundwater elevations and direction of groundwater flow.

Depth to groundwater at the Groseclose site ranged from approximately 4.0 to 6.0 feet bgs. The lowest relative groundwater elevations were recorded at monitoring well G-14 located in the northern portion of the site adjacent to the canal. The highest relative groundwater elevations were calculated to be at monitoring well G-16 which is located between the gray and blackwater drainfields. The higher water table elevations in this area may be attributed to groundwater mounding from constant loading of wastewater to the drainfield. Figure 16 shows the groundwater elevations and contours calculated from data collected on February 12, 1992.

Based on the initial piezometer measurements, the general direction of groundwater flow at the Groseclose site was determined to be generally north toward the canal. Subsequent groundwater elevation data confirmed the initial groundwater flow direction, although groundwater flow direction in the immediate vicinity of the drainfield was effected by groundwater mounding and tended to flow radially from the drainfield. The groundwater gradient was calculated to be 0.033 ft/foot in the area immediately downgradient of the drainfield. This gradient decreased to 0.025 ft/foot between wells G-9, G-10, G-11 and wells G-13, G-14, and G-15.

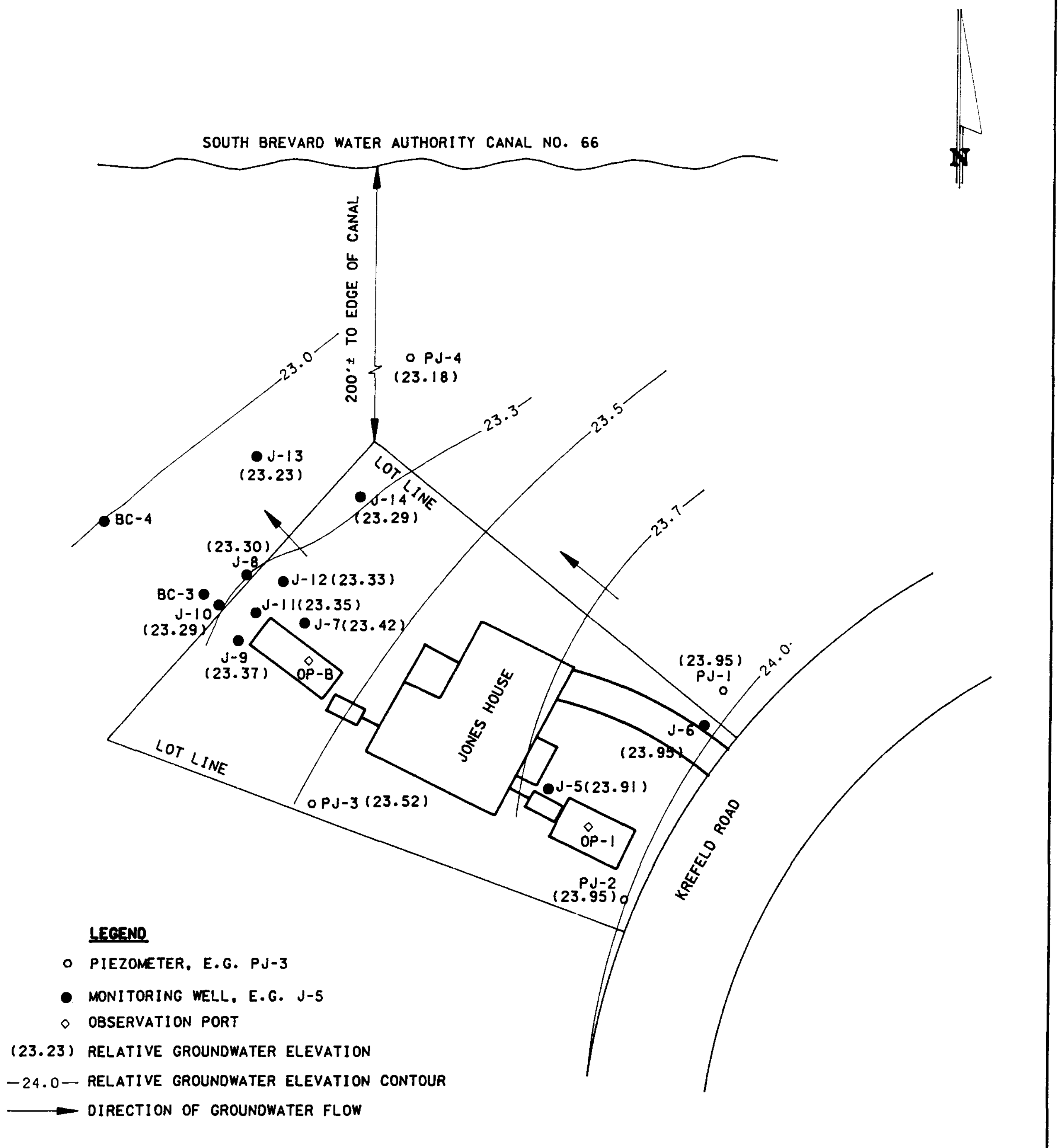
Depth to groundwater at the Jones site ranged from approximately 3.0 feet to 5.0 feet bgs. The lowest relative groundwater elevations were recorded at PJ-4 located in the northern portion of the site and the highest relative groundwater elevations were calculated to be at monitoring wells PJ-2, J-6, and PJ-1. Figure 17 shows the groundwater elevations and contours calculated from data collected on August 6, 1991.



LEGEND

- ● MONITORING WELL
- o PIEZOMETER
- ▽ SEEPAGE METER
- (23.23) RELATIVE GROUNDWATER ELEVATION
- —24.0— RELATIVE GROUNDWATER ELEVATION CONTOUR
- ——► DIRECTION OF GROUNDWATER FLOW

SCALE: 1"=30'	DRAWN BY:	DATE:	RELATIVE GROUNDWATER ELEVATION CONTOUR MAP FEBRUARY 12, 1992 GROSECLOSE RESIDENCE	FIGURE: 16
AYRES ASSOCIATES	CHECKED BY:	DATE: 2/4/3		
	APPROVED BY:	DATE:		



SCALE SCALE: 1"=40'	DRAWN BY: LAF	DATE: 2/12	RELATIVE GROUNDWATER ELEVATION CONTOUR MAP AUGUST 6, 1991 JONES RESIDENCE	FIGURE: 17
AYRES ASSOCIATES	CHECKED BY: JWW	DATE: 2/13		
	APPROVED BY:	DATE:		

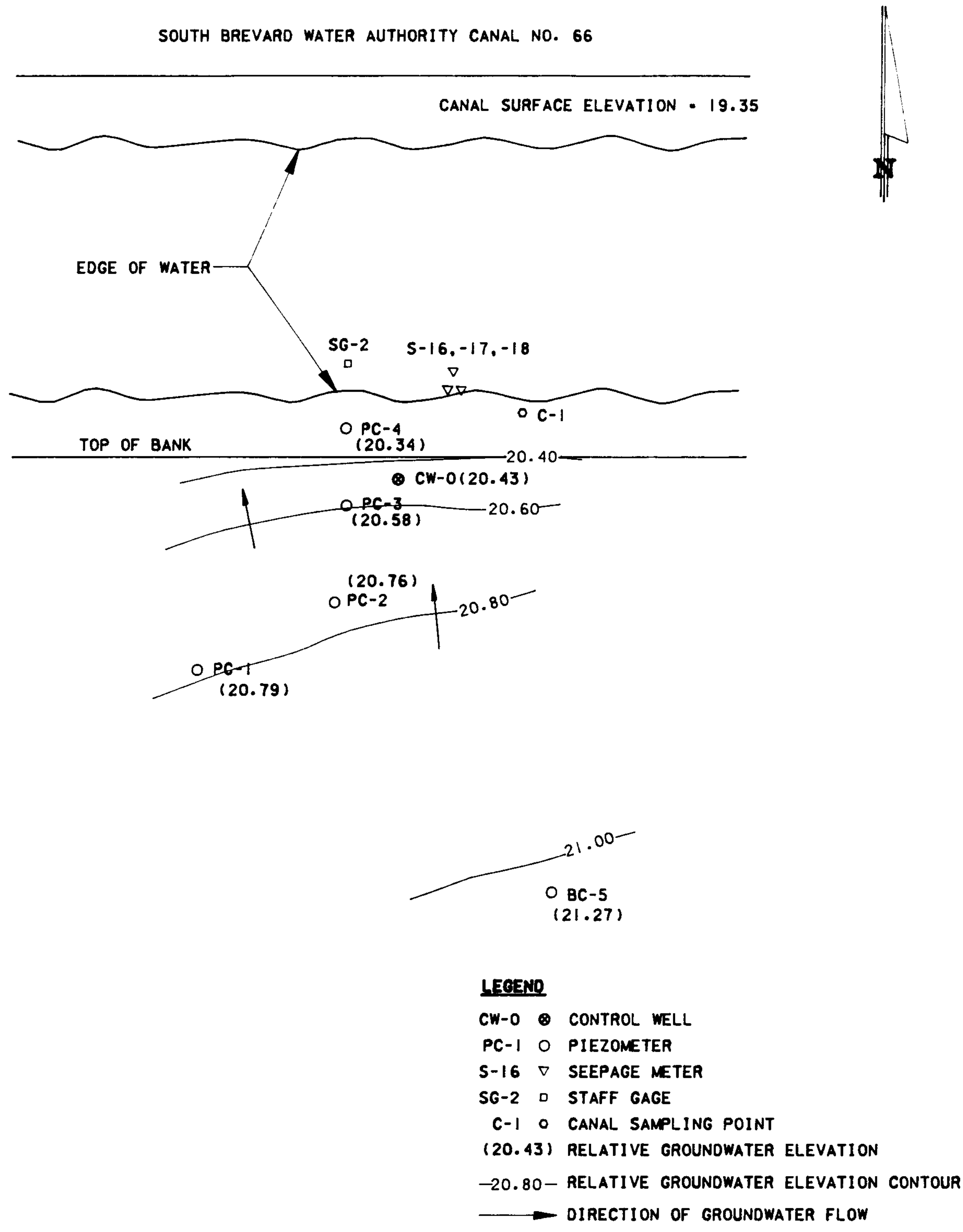
Initial piezometer measurements indicated the general direction of groundwater flow at the Jones site was north-northwest, toward Canal No. 66. Subsequent depth to groundwater measurements confirmed the groundwater flow direction. The groundwater gradient was relatively constant and flat across the site and was calculated to be approximately 0.006 feet/foot.

Depth to groundwater at the control site ranged from approximately 4.0 feet to 6.5 feet bgs. The lowest relative groundwater elevations were recorded at monitoring well PC-4 located in the northern portion of the site, however this well was located on a berm of spoil adjacent to the canal. The highest relative groundwater elevations were calculated to be at monitoring well BC-5 located in the southern portion of the site, and this is more representative of the control area in general. Figure 18 shows the groundwater elevations and contours calculated from data collected in February, 1990.

Initial piezometer measurements indicated the general direction of groundwater flow at the control site was north, toward Canal No. 66. Subsequent depth to groundwater measurements confirmed the groundwater flow direction. The groundwater gradient was calculated to be approximately 0.0029 ft./ft. in the wooded area between monitoring wells BC-5 and PC-2, but increased to approximately 0.012 ft./ft. near the canal in the area of monitoring well CW-0.

Unsaturated Zone Thickness: Water table elevations were obtained each time water samples were collected from the monitoring wells. The average water table elevation and the ranges of water table elevation are presented in Table 6, and the raw data are presented in Appendix D. Table 6 also shows the range of unsaturated soil thickness between the drainfield infiltrative surface and the water table for wells near the drainfields at the Groseclose and Jones sites.

At the time of highest measured groundwater levels (August 1991), the depth to groundwater at the Jones site for the monitoring well closest to the blackwater drainfield located in the back yard (monitoring well J-7) was 4.19 feet. At that time, there was



SCALE SCALE: 1"=40'			RELATIVE GROUNDWATER ELEVATION CONTOUR MAP FEBRUARY 1990 CONTROL SITE	FIGURE: 18
AYRES ASSOCIATES	DRAWN BY: WAF CHECKED BY: [illegible] APPROVED BY:	DATE: 2/92 DATE: 2/93 DATE:		

Table 6. Average Water Table Elevations During the Study Period (February 7, 1990 - March 26, 1992)

Well ID	Average Water Table Elevation (ft NGVD)	Water Table Elevation Range (ft NGVD)	Unsaturated Zone Range* (ft below infiltrative surface)
J-5	22.49	21.41 - 23.47	N/A
J-6	22.64	21.18 - 23.54	N/A
J-7	22.04	21.06 - 23.01	5.21 - 3.26
J-8	21.89	20.86 - 22.88	N/A
J-9	21.94	20.91 - 22.98	5.36 - 3.29
J-10	21.89	20.87 - 23.28	N/A
J-11	21.93	20.91 - 22.91	5.36 - 3.36
J-12	22.39	22.03 - 22.94	N/A
J-13	22.21	21.81 - 22.79	N/A
J-14	22.36	22.20 - 22.81	N/A
BC-5	21.07	20.10 - 22.06	N/A
CW-0	20.86	19.84 - 20.82	N/A
G-5	22.59	21.65 - 23.32	2.90 - 1.23
G-6	22.23	21.21 - 22.96	3.34 - 1.59
G-7	22.35	21.13 - 23.11	3.42 - 1.44
G-8	22.23	21.43 - 22.90	3.14 - 1.65
G-9	21.70	20.92 - 22.54	N/A
G-10	21.88	21.01 - 22.58	N/A
G-11	22.05	21.14 - 22.81	N/A
G-12	22.63	21.10 - 23.38	N/A
G-13	21.69	21.53 - 22.04	N/A
G-14	21.66	21.51 - 21.89	N/A
G-15	21.77	21.63 - 21.95	N/A

* Based on elevation of 24.55 ft. for infiltrative surface of Groseclose drainfield and 26.27 for infiltrative surface of Jones drainfield.

N/A Wells not in area of drainfield.

approximately 3.26 feet of unsaturated soil below the blackwater drainfield. The thickness of unsaturated soil below the blackwater drainfield (based on J-7 data) ranged from 3.26 to 5.21 feet during the study period, well above the current requirement for 2 feet of unsaturated soil found in Chapter 10D-6, FAC.

At the Groseclose site, the highest groundwater level measured near the drainfields (monitoring well G-5) was 4.18 feet below ground surface and also occurred in August of 1991. At that time there was only approximately 1.23 feet of unsaturated soil below the drainfields, however, due to the lower elevation of the Groseclose OSDS. This unsaturated soil thickness ranged from 1.23 to 2.90 feet over the study period, and was considerably less than at the Jones site. The water table at the Groseclose residence was commonly within 2 feet of the drainfield infiltrative surface.

As the data indicates, water table elevation in the monitoring wells varied by as much as two feet during the study period. The water table in most of the wells was at its highest level during the February 28, 1990 or August 19, 1991, sampling events. Estimated rainfall totals for one and seven days prior to each sampling event are presented in Table 7. These data were used in regression analysis of the data, discussed later.

Table 7. Rainfall Totals for Periods 1-Day and 7-Days Prior to Sampling Date*

Sample Date	1-Day Rainfall (inches)	7-Day Rainfall (inches)
07-Feb-90	0.00	0.00
28-Feb-90	0.05	0.05
15-Mar-90	0.02	0.02
04-Apr-90	0.60	1.08
25-Apr-90	0.00	0.07
09-May-90	0.00	0.01
31-May-90	0.00	1.91
27-Jun-90	0.50	2.90
23-Jul-90	2.05	2.88
07-Aug-90	0.01	1.12
19-Aug-91	0.00	0.25
20-Aug-91	0.10	0.35
18-Sep-91	0.00	0.00
19-Dec-91	0.00	0.00
08-Jan-92	0.00	1.70
03-Mar-92	0.00	0.00
26-Mar-92	1.20	1.45

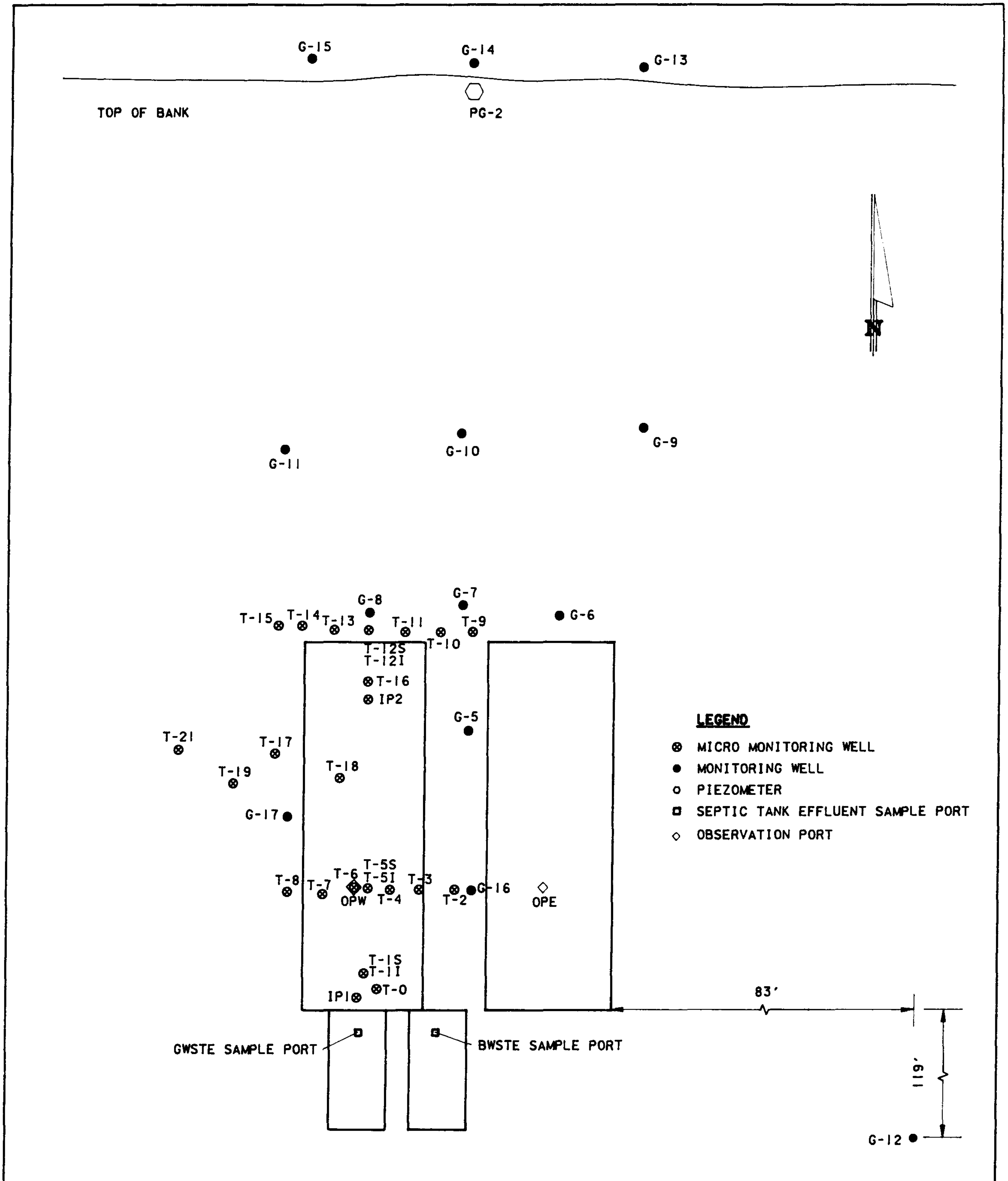
* Based on an average of rainfall data collected at Wilbro Dairy, 2 miles South of study site, and west Melbourne WWTP, 1.5 miles NE of study site.

Tracer Tests

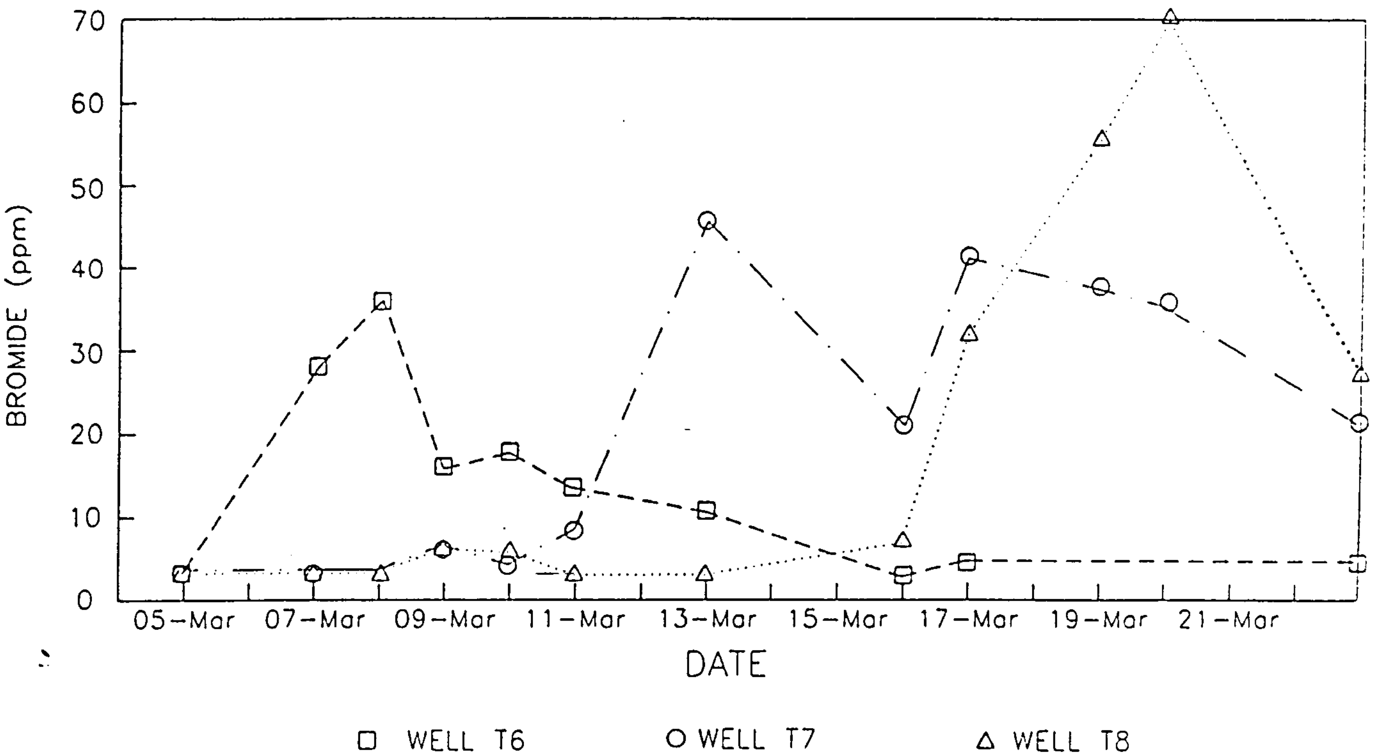
The tracer test sampling locations are shown in Figure 19 and the complete tracer test results are included in Appendix B. Travel time through the unsaturated zone could not be determined using data from Tracer Test #1. Concentrations of bromide were never detected in wellpoint T-0 adjacent to the IP-1 injection port. The tracer slug was injected in IP-1 during a graywater discharge by the home residents, and instead of seeping into the unsaturated zone below the injection port, the slug entered the gravel-filled drainfield with the wastewater. Bromide tracer was detected in wellpoints T-6, T-7, and T-8. The Br^- concentration curves for these wellpoints are shown on Figure 20. During Test #1, the highest Br^- concentrations were recorded in samples from Wellpoint T-8.

The direction of groundwater tracer movement from IP-1 northwestward toward T-8 is consistent with the groundwater flow direction indicated by the detailed water-level contours in the vicinity of the drainfields (Figure 21). The northwestward flow direction is the result of water-table mounding beneath the drainfields and subsequent radial flow outward. After tracer was observed in the southern row of wellpoints (T-6, T-7, and T-8), additional wellpoints (T-17, T-18, T-19, and T-21) were installed in a downgradient direction and sampled for Br^-. The maximum or peak concentration of bromide occurred in wellpoint T-21 38.13 days after the peak concentration was recorded in wellpoint T-8 (Figure 22). The peak concentration recorded in wellpoint T-21 was less than half of the peak concentration recorded in wellpoint T-8, demonstrating the effects of dilution and dispersion on the tracer slug as it moved in the downgradient direction with the flow of groundwater.

Using the peak-to-peak tracer travel time between wellpoints T-8 and T-21, the hydraulic conductivity of the soils between the wellpoints can be calculated (Todd 1980; Davis et al. 1985). The average interstitial velocity (V_a) of the tracer slug can be expressed using



SCALE		MINIATURE WELLPOINT & MONITORING WELL LOCATION MAP	FIGURE:
SCALE: 1"=10'	DRAWN BY: WPF DATE: 2/92	**GROSECLOSE RESIDENCE**	19
AYRES ASSOCIATES	CHECKED BY: DATE: 2/93		
	APPROVED BY: DATE:		



AYRES ASSOCIATES

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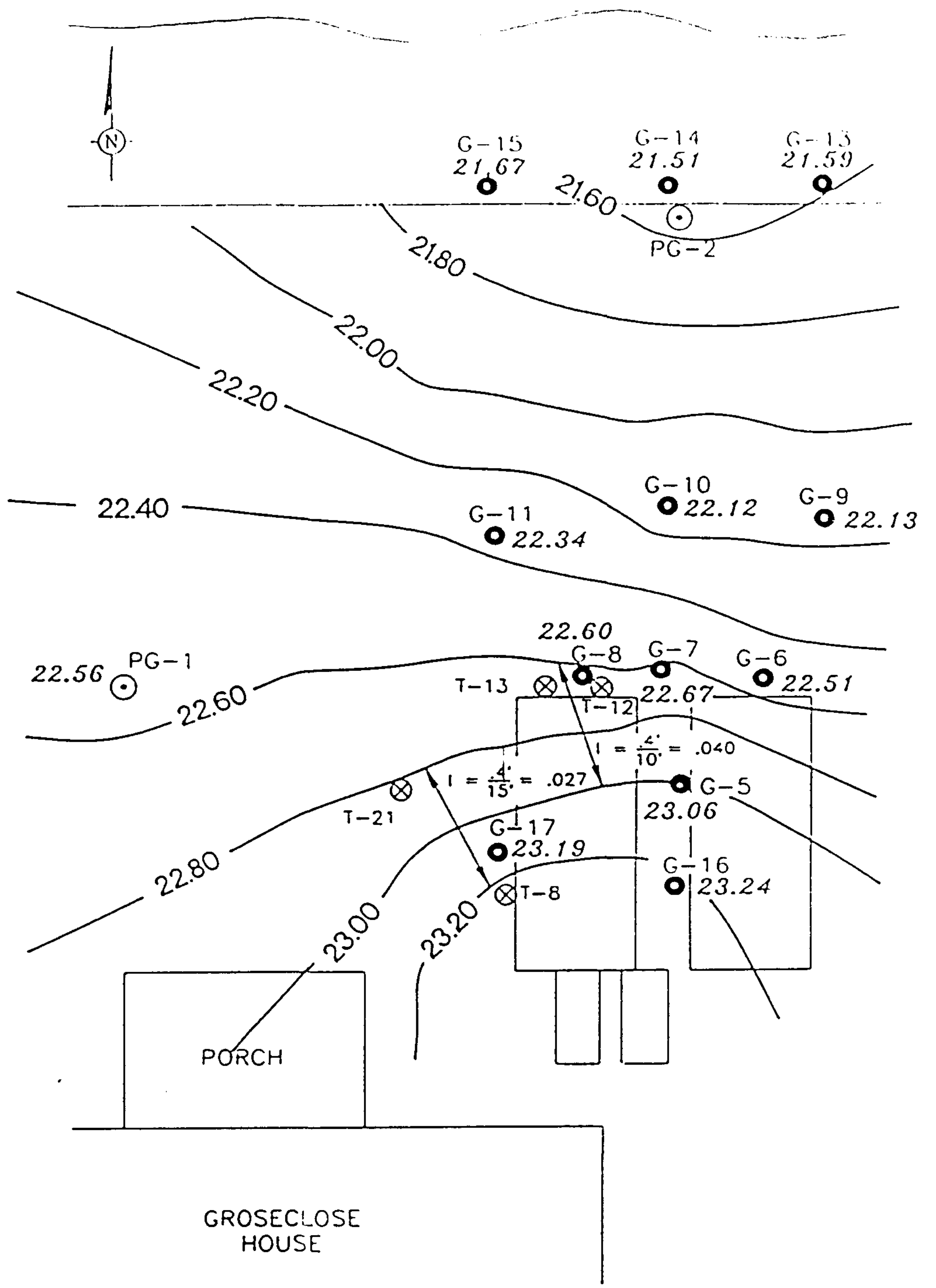
TRACER TEST •1 - BROMIDE CONCENTRATION vs. TIME, WELLPOINTS T6, T7, AND T8

FIGURE: 20



AYRES ASSOCIATES

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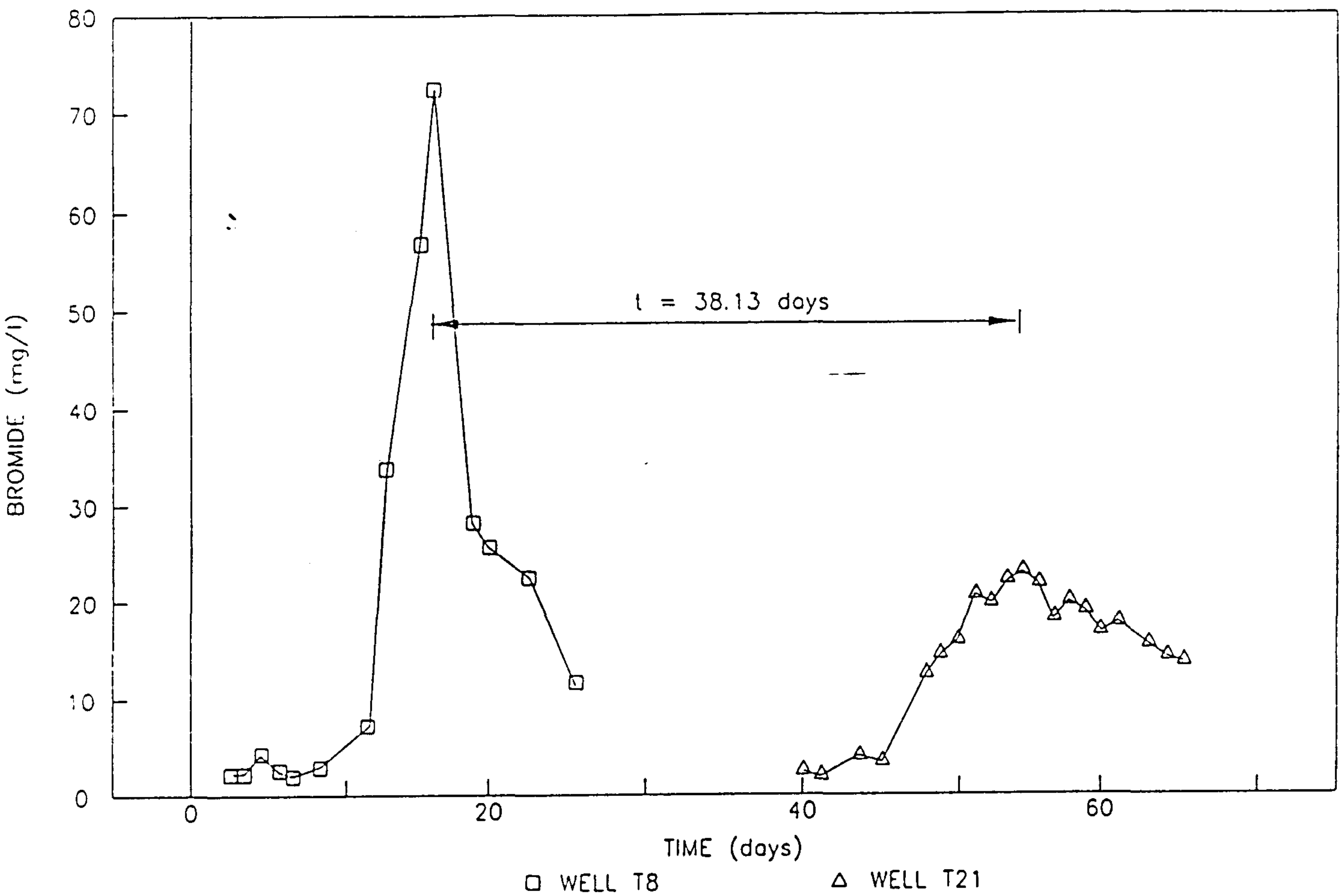
CHECKED BY:	DATE: 2/93
APPROVED BY:	DATE:

DETAILED WATER LEVEL CONTOURS IN VICINITY OF DRAINFIELD

FIGURE: 21



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WOF	2/92
CHECKED BY:	DATE:
JMcG	2/93
APPROVED BY:	DATE:

TRACER TEST #1 - BROMIDE CONCENTRATION vs. TIME, WELLPOINTS T8 AND T21

FIGURE: 22

a form of the Darcy Equation:

$$V_a = \frac{K}{\eta_e} \frac{\Delta h}{L}$$

Where K is hydraulic conductivity of the soil, η_e is the effective porosity of the soil, Δh is the hydraulic head difference between the wellpoints and L is the distance between the wellpoints. The average velocity (V_a) can also be expressed using the equation:

$$V_a = \frac{L}{t}$$

Where L is equal to the distance between the wellpoints and t is the travel time. Equating both velocity terms yields:

$$\frac{K\Delta h}{\eta_e L} = \frac{L}{t}$$

and solving for K:

$$K = \frac{\eta_e L^2}{\Delta ht}$$

To find K using Tracer Test #1 data:

η_e = 0.20 *(estimated)*
L = 0.38 *feet (ft) the distance between wellpoints T-8 and T-21*
Δh = 0.38 *ft, the head difference between wellpoints T-8 and T-21*
t = 38.13 *days, the peak-to-peak tracer travel time between wellpoints T-8 and T-21*

$$K = (0.20)\ (14.00\ ft)^2\ /\ (0.38\ ft)\ (38.13\ days)$$

$$K = 2.70\ ft/day$$

Travel time through the unsaturated zone was measured during Tracer Test #2. After the initial tracer slug injection in IP-2, the Br^- tracer was detected in Wellpoint T-16 after approximately 2 days (wellpoint T-16 was located adjacent to IP-2). A peak tracer concentration (Br^- 6200 ppm) was detected in wellpoint T-16 approximately 5 days after initial injection of the slug beneath the drainfield.

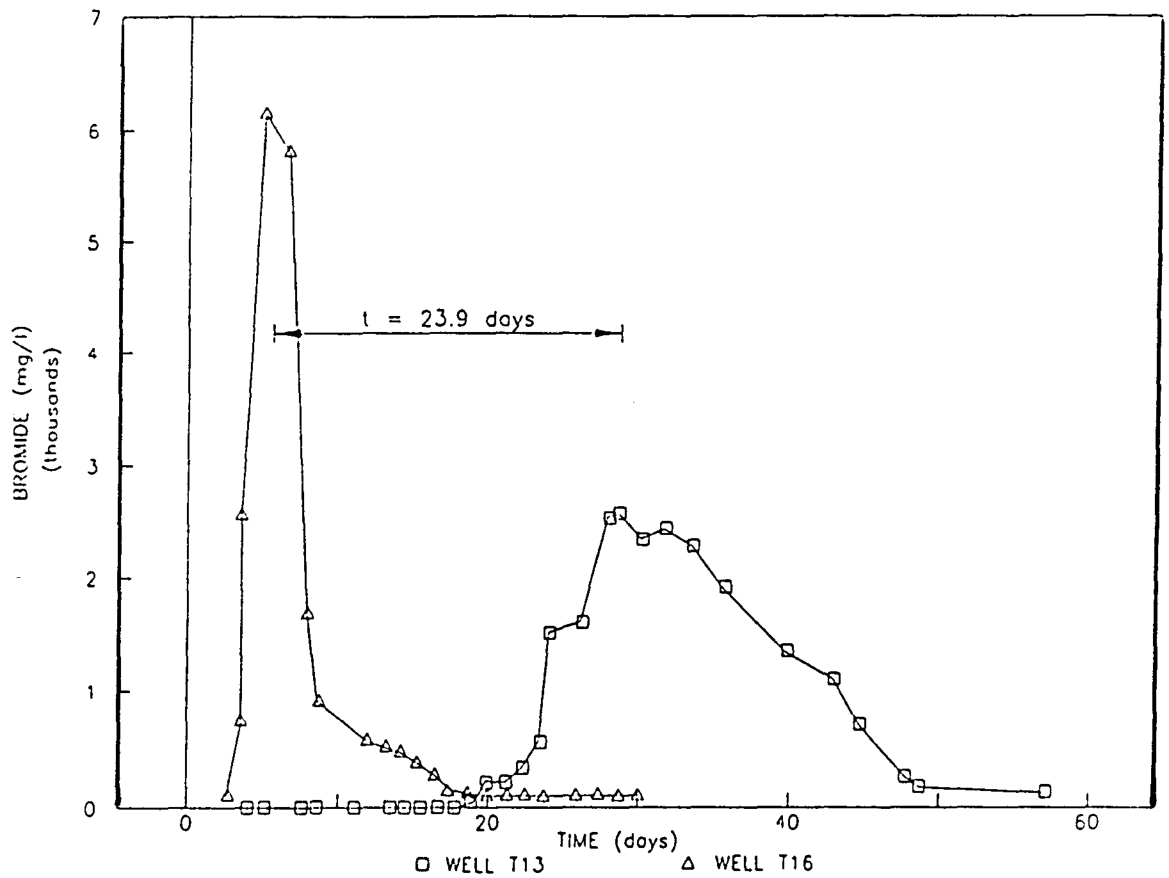
Approximately 12 days after initial slug injection, the tracer was detected in the northern row of wellpoints in wellpoint T-12S. Wellpoint T-12S was located 4.5 feet from IP-2 and approximately due north. A peak Br^- concentration (72.6 ppm) was detected in wellpoint T-12S approximately 16 days after initial slug placement in IP-2. Approximately 18 days after the slug was placed in IP-2 bromide was detected in wellpoint T-13. Wellpoint T-13 was located 5.0 feet from IP-2 and slightly to the northwest. A peak Br^- concentration (2600 ppm) was detected in wellpoint T-13 approximately 29 days after initial injection of the slug. The tracer was not detected in any of the other wellpoints during Test #2. Based on the Br^- analyses from all the wellpoints, it is apparent that the bulk of the tracer slug passed through wellpoint T-13. This flow path of the tracer is consistent with the ground-water flow direction indicated by the water-level contours (Figure 21). Groundwater flows from the drainfield toward the canal adjacent to the northern property boundary. Evaluation of the Br^- data also suggested an essentially horizontal groundwater flow path. Bromide was not detected in the intermediate depth wellpoint T-12i completed two feet below T-12S.

The peak-to-peak tracer travel time between wellpoints T-16 and T-13 was 23.9 days as shown on Figure 23. Using this travel time, the hydraulic conductivity of the shallow soils can be calculated using the same methods as those used for Test #1.

$$K = \eta_e \, (L)^2 / \Delta h t$$

η_e = 0.20 (*estimated*)
L = 5.70 *ft, the distance between wellpoints T–16 and T–13*
Δh = 0.20 *ft., the head difference between wellpoints T–16 and T–13*
t = 23.9 *days, the peak to peak tracer travel time between wellpoints T–16 and T–13*

$$K = 1.36 \ ft/day$$



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JMW	2/93
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TRACER TEST #2 - BROMIDE CONCENTRATION vs. TIME, WELLPOINTS T13 AND T16

FIGURE: 23

Hydraulic conductivities calculated from slug tests conducted at the Groseclose site during subsurface characterization ranged from 0.47 to 4.74 ft./day, with an average of 1.40 ft/day. The hydraulic conductivities calculated from the tracer tests, 2.70 ft./day and 1.36 ft./day, agree well with those previously calculated using slug test data.

Based on tracer test #2, average groundwater seepage velocity immediately downgradient of the drainfield was 5.70 feet in 23.9 days, or approximately 0.24 ft./day. Assuming a constant velocity, this data would indicate an approximate travel time of 272 days from the drainfield to the canal, approximately 65 feet downgradient.

Tracer test #2 yielded on unsaturated zone travel time of approximately 5 days based on peak bromide concentrations. At the time of this test, the approximate thickness of unsaturated soil between the infiltrative surface of the drainfield and the water table was 1.75 feet. This yields an estimated unsaturated zone flow rate of 0.35 feet/day under the drainfield during the study period. This value is higher than the saturated seepage velocity calculated above and would suggest that relatively saturated flow was occurring from the drainfield to the water table immediately below. This is not surprising considering the small unsaturated thickness, the potential capillary fringe in the fine sandy soil, and the constant wastewater loading from the drainfield above.

Septic Tank Effluent (STE) Characterization

STE Quality: The blackwater and graywater septic tank effluent was sampled at the Groseclose site to assess the quality of wastewater discharged to the OSDS drainfields. The blackwater septic tank received wastewater from the toilets and the kitchen. The graywater septic tank received wastewater from the bath and showers, clotheswashing machine, and all other wastewater sources. The results of the septic tank effluent sampling at the Groseclose site are summarized in Table 8.

The blackwater STE contained considerably higher concentrations of all parameters monitored except chloride, foaming agents, and oil and grease. As would be expected due to the anaerobic conditions in the septic tanks, almost all nitrogen occurred in the organic and ammonia form. Chloride concentrations were similar between the two wastewater flowstreams while the concentrations of foaming agents and oil and grease were higher in the graywater STE.

Table 8. Septic Tank Effluent Characteristics, Groseclose Residence (mg/L)

PARAMETER	Graywater			AVERAGE Graywater	BLACK WATER		AVERAGE BLACK WATER
	11/20/91	2/12/92	4/19/92		2/12/92	4/9/92	
COD	350	160	280	263	340	460	400
BOD	100	117	113	110	229	176	202.5
TSS	44	20	35	33	37	80	58.5
TDS	478	442	456	459	598	562	580
NO_3-N	0.13	0.02	0.02	0.06	<0.01	0.03	<0.02
TKN	3.6	3.3	4.6	3.8	120	100	110
TP	2	0.56	0.87	1.14	19	17	18
CL^-	160	120	150	143	150	170	160
FOAMING AGENTS	58	19	32	36	4.8	7.8	6.3
OIL & GREASE	19	38	30	29	13	23	18
FECAL COLI	49,000	24,000	1,300	24,767	150,000	130,000	140,000
FECAL STREP	<10	<10	<10	<10	48,000	51,000	49,500

These results are typical of blackwater/graywater septic systems with the exception of the total phosphorus data. Typically graywater STE contains as much or more phosphorous than blackwater STE. The phosphorus results at the Groseclose residence suggest the use of a phosphate-free laundry detergent, leading to reduced total phosphorus levels in the graywater waste stream. The higher phosphorus levels in the blackwater waste stream suggested that an automatic dishwasher in the home discharged to the blackwater septic tank.

STE Quantity: Water meter readings were collected from the Groseclose and Jones residences at 6 different dates over the course of the study from December 1989 to April 1992. These data were used to estimate the average wastewater flow to the OSDS at the two homes. Table 9 summarizes the data and presents estimates of wastewater loading to the OSDS drainfields.

Water use ranged from 372 to 435 gpd at the Groseclose home and from 166 to 257 at the Jones home based on the water meter data collected over the study. Average per capita water use was 80 gallons/capita/day (gpcd) at Groseclose and 101 gpcd at Jones.

Because the Groseclose home utilized canal water for lawn watering, the average water use was used as an estimate of wastewater flow to the OSDS. Assuming an equal split between blackwater and graywater flows, this resulted in an estimated average wastewater loading of 0.71 gallons/ft^2/day to the drainfields at the Groseclose site.

The Jones home utilized municipal water for lawn watering and had a higher per capita water use during the study than the Groseclose home. For this reason the minimum water use interval was used as an estimate of wastewater flows at the Jones home. Assuming an equal split between blackwater and graywater flows, this resulted in an estimated average wastewater loading of 0.44 gal./ft^2/day to the drainfields at the Jones site.

Table 9. Water Use and Estimated Wastewater Flows, Groseclose and Jones Residences

(Based on 6 water meter readings from 12/89 to 4/92)

PARAMETER	HOME	
	Groseclose Residence	Jones Residence
Average Water Use (gpd)	400	203
Range (gpd)	372-435	166-257
Average Per Capita Water Use (gpd)	80	101
Drainfield Area (ft^2)	560	380
Estimated Wastewater Loading ($gal/ft^2/day$)	0.71*	0.44*

* Wastewater loading at the Groseclose based on average flow of 400 gpd because canal water was used for lawn irrigation. Wastewater loading at Jones is based on the minimum average water use of 166 gpd as water use figures included water for lawn irrigation.

Groundwater and Surface Water Quality Results

Physical/Chemical Data: Groundwater samples were obtained from monitoring wells on seventeen different sampling dates (for most locations) from February 7, 1990, through March 26, 1992. Raw data for each monitoring well are presented in Appendix D; average concentrations are summarized in Table 10. Figures 24 and 25 show water quality results for selected parameters on maps of the Jones and Groseclose sites. In general, contaminant levels were elevated over background levels in wells near the OSDS drainfields.

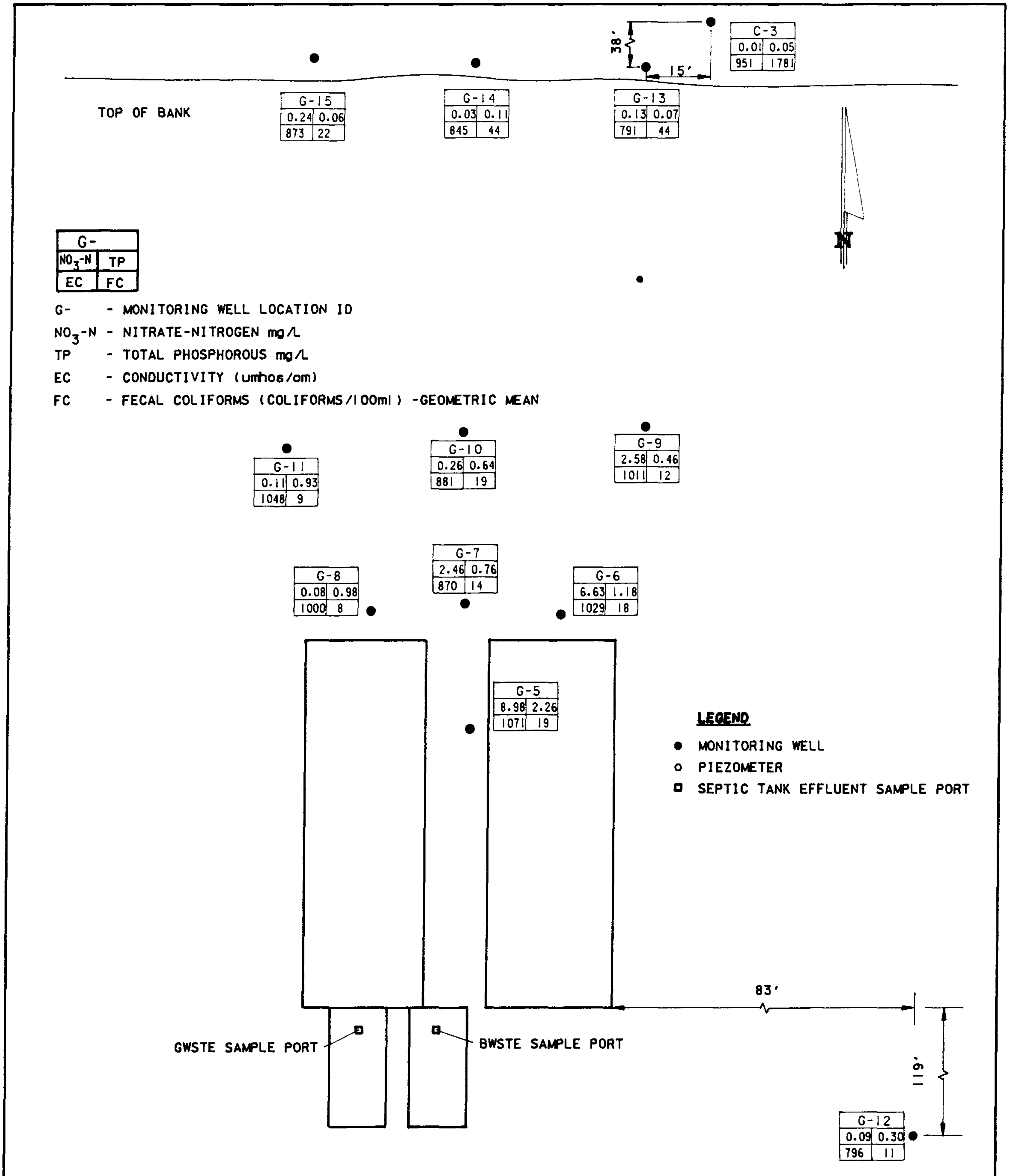
Background concentrations were determined from wells located in the front yard of each residence (G-12, J-6) and from wells at the control site (BC-5, CW0). The front yard wells were located considerable distances from the septic systems and were also up-gradient from the drainfields. Sometimes there were significant ($p \leq 0.05$) differences in parameter concentrations between the various background wells. Statistical testing based on the Fisher PLSD test (as discussed in the methods section) indicated significant differences at the 0.05 significance level between the control wells for the parameters indicated in Table 11. Scatterplots and other ANOVA testing calculations are presented in Appendix H.

As a result of these differences, it was decided that the most valid comparisons were generally between monitoring well and background well data at each individual residence (background wells G-12, J-6).

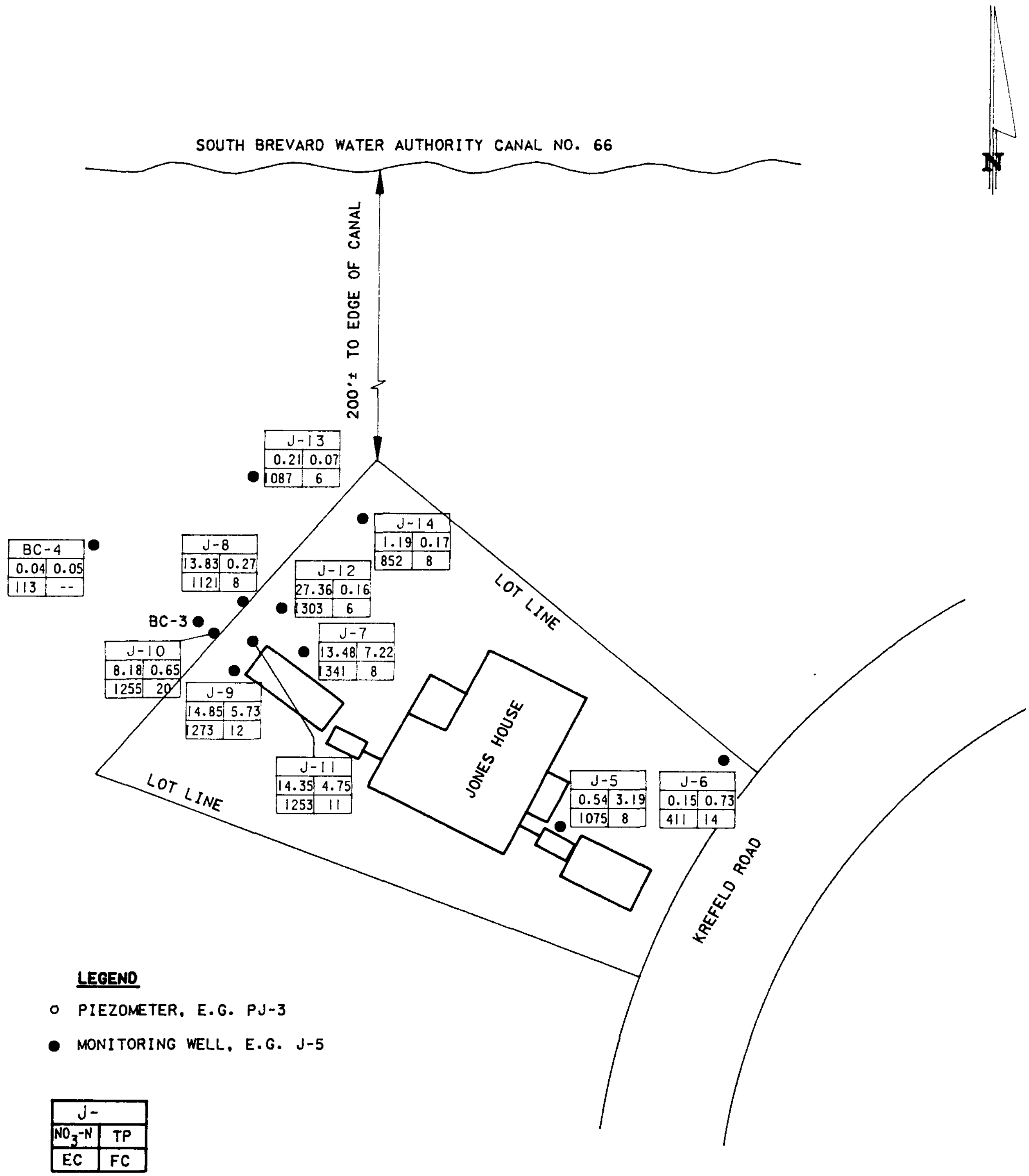
The concentration of nitrate (NO_3,NO_2-N), total kjeldahl nitrogen (TKN), total phosphorus (TP) and conductivity were generally higher in the wells in the vicinity of the drainfield when compared to the background wells. The concentrations of these parameters were always at or below the site background concentrations in wells fifty (50) feet or more from the drainfield (G-13, G-14, G-15, J-13).

Table 10 **Monitoring Well Average Values for the Sampling Periods (February 7, 1990 - March 26, 1992)**

Site	Samples (n)	C.O.D. (mg/l)	Color (CPU)	Conductivity (μmhos/cm)	D. O. (mg/l)	NO_3,NO_2-N (mg/l)	pH	Temp. (°C)	T.K.N. (mg/l)	TP (mg/l)	Turbidity (NTU)
G- 5	15	58	217	1070	4.6	8.98	6.78	23.7	2.51	2.26	110
G- 6	16	80	199	1029	5.1	6.63	6.76	23.1	3.30	1.18	102
G- 7	16	76	297	870	4.4	2.46	6.63	23.3	2.62	0.76	105
G- 8	16	62	190	1000	4.0	0.08	6.83	23.6	1.87	0.98	106
G- 9	15	69	174	1011	4.8	2.58	6.94	23.7	1.74	0.46	128
G-10	16	68	323	881	4.2	0.26	6.81	23.6	2.13	0.64	100
G-11	16	80	374	1048	4.9	0.11	6.79	23.7	1.48	0.93	108
G-12	17	72	206	796	4.6	0.09	6.62	24.8	1.67	0.30	95
G-13	7	45	285	711	3.7	0.07	6.71	24.1	1.03	0.03	13
G-14	7	24	127	794	4.2	0.03	6.79	24.8	1.03	0.05	12
G-15	7	35	197	830	3.2	0.01	6.65	24.2	1.91	0.02	6
J- 5	14	75	123	1075	4.6	0.54	7.04	26.0	2.00	3.19	141
J- 6	15	64	126	411	5.4	0.15	7.13	25.7	1.94	0.73	132
J- 7	14	62	125	1341	4.5	13.48	4.63	23.9	7.19	7.22	139
J- 8	15	37	64	1121	3.6	13.83	5.02	23.6	5.90	0.27	49
J- 9	14	70	182	1273	4.8	14.85	6.24	24.0	5.88	5.73	99
J-10	14	90	113	1255	4.6	8.18	6.09	23.7	5.06	0.65	151
J-11	15	66	161	1253	3.8	14.34	5.13	23.7	8.01	4.75	63
J-12	7	37	145	1303	3.4	27.36	5.95	25.3	11.92	0.16	18
J-13	7	71	149	959	3.7	0.21	5.20	25.5	1.69	0.07	15
J-14	6	145	445	852	3.4	1.19	6.84	26.2	3.01	0.17	15
BC-5	13	50	200	270	6.0	0.03	6.40	22.9	0.95	0.14	52
CW-0	15	62	175	393	4.2	0.06	6.60	23.0	1.51	0.35	118



SCALE: 1" = 10'		AVERAGE CONCENTRATIONS OF SELECTED WATER QUALITY PARAMETERS GROSECLOSE RESIDENCE - MONITORING WELLS	FIGURE: 24
AYRES ASSOCIATES	DRAWN BY: DATE: 12/92 CHECKED BY: DATE: 2/93 APPROVED BY: DATE:		



LEGEND

o PIEZOMETER, E.G. PJ-3

● MONITORING WELL, E.G. J-5

J- - MONITORING WELL LOCATION ID

NO_3-N - NITRATE-NITROGEN mg/L

TP - TOTAL PHOSPHOROUS mg/L

EC - CONDUCTIVITY (umhos/cm)

FC - FECAL COLIFORMS (COLIFORMS/100ml) -GEOMETRIC MEAN

SCALE: 1"=40'	DRAWN BY: WAF	DATE: 12/92	AVERAGE CONCENTRATIONS OF SELECTED WATER QUALITY PARAMETERS JONES RESIDENCE	FIGURE: 25
AYRES ASSOCIATES	CHECKED BY:	DATE: 2/93		
	APPROVED BY:	DATE:		

Table 12 summarizes the results of the surface water sampling at the canal sites C0 through C5 over the study period. Raw data for each sample point are included in Appendix E. In general, key physical/chemical parameters especially nutrients, were low in the canal samples relative to groundwater.

Table 11. Parameters which Showed Significant Differences (at $\leq$0.05) Between Control Sites Determined using the Fisher PLSD Test

Comparisons	Parameter
J-6 vs BC-5	NO_3
J-6 vs CWO	TP
J-6 vs G-12	
J-6 vs BC-5	
J-6 vs BC-5	TKN
CWO vs BC-5	
CWO vs BC-5	D.O.
J-6 vs G-12	Conductivity
J-6 vs BC-5	
CWO vs BC-5	
CWO vs G-12	

Bacterial Data:

Septic tank effluent generally contains 10^5-10^6 fecal coliforms per 100 ml of effluent (Ayres Associates 1989), while fecal streptococcus levels in septic tank effluent have been reported to range from 10^3-10^7 colonies per 100 ml of effluent (Hagedorn et al. 1981; Cogger, et al. 1988). In view of these reports, bacterial contamination of ground and surface water is a concern at any OSDS site.

Fecal streptococci have been used with fecal coliforms to differentiate human fecal contamination from that of other warm-blooded animals. Fecal coliform/fecal streptococcus (FC/FS) ratios may provide information on possible sources of pollution.

Table 12. **Canal Surface Water Average Values for the Sampling Periods (February 7, 1990 - March 26, 1992)**

Site	Samples (n)	C.O.D. (mg/l)	Color (CPU)	Conductivity (μmhos/cm)	D. O. (mg/l)	NO_3,NO_2-N (mg/l)	pH	Temp. (°C)	T.K.N. (mg/l)	TP (mg/l)	Turbidity (NTU)
C0	14	39	131	1351	6.8	0.02	7.49	24.1	0.99	0.08	4
C1	15	38	138	1133	7.1	0.01	7.56	25.0	0.72	0.05	4
C2	15	45	138	1152	6.9	0.01	7.48	24.9	1.06	0.04	5
C3	16	53	176	951	7.1	0.01	7.50	24.3	1.43	0.05	8
C4	13	65	214	1030	8.7	0.01	7.65	26.3	1.03	0.05	9
C5	12	71	256	791	5.4	0.01	7.53	26.0	1.23	0.03	8.5

Estimated per capita contributions of fecal coliforms and fecal streptococci for animals have been used to develop the following FC/FS ratios (Standard Methods, 1985):

Human	4.4
Duck	0.6
Sheep	0.4
Chicken	0.4
Pig	0.4
Cow	0.2
Turkey	0.1

A ratio greater than 4.1 is considered indicative of pollution derived from domestic wastes composed of human excrement whereas ratios less than 0.7 suggest pollution due to nonhuman sources (Standard Methods, 1985). The FC/FS ratio of the Groseclose STE samples ranged from 3.0 to 4.1 based on a limited number of samples.

The validity of the FC/FS ratio to differentiate between human and animal sources of pollution has recently been questioned. Apparently, some fecal streptococcus group species survive longer then others (Feacham, R., 1975) and some methods of enumerating fecal streptococci, such as the KF membrane filter procedure, apparently give false-positive rates (Fujioka, R.S., et al., 1984; Olivieri, V.P., et al. 1977; Ericksen, T.H., et al. 1983). FC/FS ratios were nonetheless evaluated in this study due to the observation of numerous animal sources along the study canal.

Monitoring wells and canal sites C0-C5 were sampled for fecal coliform and fecal streptococcus bacteria at each sampling event. Results of the bacterial sampling are presented in Tables 13 and 14. Downgradient canal sites NC-1 through NC-4 (see Figure 2) were sampled on four occasions over the study period, these data are shown on Table 15.

Fecal coliform counts in the monitoring wells were generally below 10 col./100 ml in 66% of the samples (Table 13). The highest fecal coliform counts were found in wells G-13 and G-14 with an average of 44col./100 ml (geometric mean). Fecal coliform counts in

Table 13. Fecal Coliform Counts During the Study Period (col./100 ml)

ID	7 Feb 90	28 Feb 90	15 Mar 90	4 Apr 90	25 Apr 90	9 May 90	31 May 90	27 Jun 90	23 Jul 90
J-5	<1	<1	<1	<10	<10	<10	<10	<10	<10
J-6	<1	<1	10	<10	<10	<10	<10	<10	<10
J-7	<1	<1	<1	<10	<10	<10	<10	<10	<10
J-8	2	<1	<1	<10	<10	<10	<10	<10	<10
J-9	<1	<1	<1	<10	<10	<10	10	<10	<10
J-10	92	30	<10	<10	<10	73	<10	<10	<10
J-11	20	<1	<1	<10	<10	<10	10	<10	<10
J-12	**								<10
J-13	**								
J-14	**								
G-5	15	<1	18	<10	<10	<10	<10	<10	<10
G-6	<1	<1	240	<10	20	37	<10	15	<10
G-7	<1	<1	<1	<10	<10	<10	<10	10	<10
G-8	<1	10	<1	<10	<10	<10	<10	10	<10
G-9	<1	<1	<1	<10	<10	<10	<10	10	<10
G-10	51	10	<1	<10	<10	<10	<10	230	<10
G-11	1	<1	2	<10	<10	<10	<10	10	30
G-12	<1	<1	2	20	<10	<10	<10	<10	<10
G-13	**								490
G-14	**								<10
G-15	**								150
BC 5	<1	1	<1	<10	<10	10	<10	10	<10
CW 0	<1	30	6	<10	<10	<10	<10	<10	<10
C0	300	880	1035	420	310	680	120	400	1200
C1	765	350	1260	560	1380	390	420	700	1400
C2	1900	400	940	1515	440	300	70	800	700
C3	4100	1385	2300	3300	4500	8100	8300	1300	700
C4	160	770	100	200	290	430	dry	dry	180
C5	48	80	705	920	40	40	40	190	200

Table 13. (Contd) Fecal Coliform Counts During the Study Period (col./100 ml)

I D	8 Aug 90	14 Sep 90	19 Aug 91	20 Aug 91	18 Sep 91	19 Dec 91	8 Jan 92	26 Mar 92	Geo. Mean
J-5	<10	50	<10	--	11	12	--	80	8
J-6	<10	10	<10	--	6600	<10	<20	220 (40)	14
J-7	<10	10	<10	--	350	<4	--	10 (<10)	8
J-8	<10	40	<10	--	16	<10	<4	40	8
J-9	<10	<10	<10	--	>2000	<10	--	620	12
J-10	<10	<10	<10	--	510	<20	--	20	20
J-11	<10	<10	<10	--	2360	<2 (<2)	<4	89	11
J-12	10	<10	<2 (<1)	--	4	<2	<2	52	6
J-13	<10	<10	<1	--	50	<2	<2	<10 (<10)	6
J-14	<10	<10	<10	--	21	<2 (<2)	--	8	8
G-5	<10	<10	--	450	1180	<20 (<20)	--	10 (70)	19
G-6	<10	10	--	340	1300	<10	4	20	18
G-7	<10	<10	--	440	17600	<10	<4	40	14
G-8	<10	<10	--	105	100	940	<4	60	8
G-9	<10	<10	--	780	310	<20	--	<20	12
G-10	<10	<10	10	890	150 (170)	20 (<20)	--	<10	19
G-11	<10	<10	10	20	50	<20	--	<20	9
G-12	<10	<10	<10	140	<1	<20	<4	100	11
G-13	50	<10	<10	240	500	<10	10	20	44
G-14	<10	<10	130	>200	6300	<20	<10	<20 (<20)	44
G-15	10	20	17	80	50	<4 (12)	<2	36	22
BC 5	<10	(')	--	--	3 (5)	4	--	--	5
CW 0	<10	<10	36 (24)	--	8	26	4	5	9
C0	1100	540	--	1210	270	470 (660)	--	2084	585
C1	790	290	--	2400	220	1000	750	1120	711
C2	900	(')	--	310	250	920	560	1860	591
C3	800	1800	--	11000	1030	90	204	2840 (3008)	1783
C4	1100	(')	--	400		18	--	3470	296
C5	30	40	--	725	50	280	--	2930	143

( ) Denotes Duplicate Samples; average of 2 was taken in Geometric Mean.
Values of >1000 taken to be 1000 for calculating Geometric Mean.
-- No Sample
** These wells were not installed until later in the study.

Table 14. Fecal Streptococcus Counts During the Study Period (col./100 ml)

ID	7 Feb 90	28 Feb 90	15 Mar 90	4 Apr 90	25 Apr 90	9 May 90	31 May 90	27 Jun 90	23 Jul 90
J-5	1300	4000	2800	6100	20000	5600	2900	5500	1600
J-6	8400	4500	1500	6000	7000	5400	1700	5200	1400
J-7	2300	>10000	8000	>10000	6300	24100	11000	21000	4500
J-8	4500	>10000	1500	3200	2400	31000	5400	6200	1200
J-9	10500	>10000	3000	6300	5100	2700	3700	5000	1000
J-10	6000	11500	3100	800	2500	2050	1200	4400	1200
J-11	5200	4400	600	4900	800	1100	1100	7200	3000
J-12	**								900
J-13	**								2400
J-14	**								4100
G-5	1400	5000	2100	5500	1100	6500	1400	3500	880
G-6	3700	6600	3300	4600	300	600	800	3400	3800
G-7	12100	>10000	4000	7100	1300	1100	950	8200	1200
G-8	>10000	4600	3600	1900	2200	26000	4500	38000	5600
G-9	2700	4500	2500	1700	500	5000	580	6500	800
G-10	12300	2400	4100	5900	1000	1100	800	800	700
G-11	1300	3000	1300	3600	700	700	400	1400	1000
G-12	>1000	540	2200	420	400	280	260	290	340
G-13	**								650
G-14	**								1000
G-15	**								500
BC 5	2500	5100	1600	500	4600	680	2200	4400	400
CW 0	9700	12300	400	370	320	870	400	2600	700
C0	5100	10000	8000	4900	13500	2300	7900	13000	10000
C1	5000	>10000	12000	3900	18800	3100	78000	41000	7900
C2	10000	>10000	9000	5000	11000	2600	3300	8000	4700
C3	28000	103000	16000	24000	4900	34000	23500	49000	34000
C4	11400	6500	5300	3500	600	8500	dry	dry	4600
C5	6500	3800	4600	3300	1900	840	3900	4600	5000

Table 14. (Contd) Fecal Streptococcus Counts During the Study Period (col./100 ml)

I D	8 Aug 90	19 Aug 91	20 Aug 91	18 Sep 91	19 Dec 91	8 Jan 92	26 Mar 92	Geo. Mean
J-5	2700	6650		170	40		250	1896
J-6	880	750		2900	<20	10	610 (350)	1252
J-7	600	650		20	40		<10 (<10)	1626
J-8	510	700		10	<20	<10	<10	699
J-9	1000	2150		1500	20		<10	1512
J-10	300	1100		300	<20		10	854
J-11	1100	750		70	20 (20)	10	50	622
J-12	4300	1300		<10	<20	<10	<10	100
J-13	4000	750		30	<20	<10	<10 (<10)	123
J-14	4500	2000		10	<20 (<20)		<10	204
G-5	800		3150	1400	<20 (<20)		610 (880)	1417
G-6	700		4300	190	<20	40	200	832
G-7	1100		4250	<10	<20	<10	200	849
G-8	700		16000	40	60	920	<10	1746
G-9	1400		1000	110	<20		100	859
G-10	700	6500	2700	170 (<10)	240 (80)		130	1150
G-11	2100	2750	3350	<10	60		1400	846
G-12	300	700	1050	290	440	1400	1540	557
G-13	250	400	1150	260	40	2120	820	437
G-14	1500	2200	2600	30	<20	10	20 (20)	178
G-15	4500	100	2450	260	<20 (<20)	10	140	212
BC 5	870			<10 (10)	<10			685
CW 0	760	1100 (1000)		20	<10	<8	10	324
C0	660		7750	350	510 (670)		3740	3982
C1	4500		9800	570	2500	>40	1740	4770
C2	7500		2000	500	460	180	2940	3145
C3	7400		28000	990	120	360	4510 (5060)	9241
C4	9800		2000		240		4100	3363
C5	8000		2100	320	480		4300	2585

( ) Denotes Duplicate Samples; average of 2 was taken in Geometric Mean.
Values of >1000 taken to be 1000 for calculating Geometric Mean.
** These wells were installed later in the study.

the canals were considerably higher than the wells, especially at canal station C-3 near the Groseclose site, where the geometric average was 1793 col./ml (Table 13). Other Canal 66 sampling sites averaged between 143 to 711 fecal coliform col./100 ml (geometric mean). Average fecal coliform levels at other canal stations downstream in the drainage basin, designated NC-1 through NC-4, ranged form 37 to 242 col./100 ml (Table 15). All averages are calculated as geometric means, the preferred statistic for microbial data.

Fecal streptococcus levels were high and variable in both wells and canals, ranging nearly three orders of magnitude (Table 14). Values ranged from a geometric mean of 100 col./100 ml at well J-12 to 1896 col./100 ml at well J-5. Most well fecal streptococcus levels varied between 100 and 1500 col./ml (geometric mean). Canal 66 geometric mean levels ranged from 2585 (C5) to 924 (C3) col./100 ml (Table 14), while stations NC-1 through NC-4 exhibited lower levels, ranging from 1941 (NC-3) to 2755 (NC-4) col./ml (geometric mean), shown in Table 14.

The canal geometric means over the entire study for fecal streptococcus and fecal coliform data were 5778 col./100 ml and 978 col./100ml, respectively. This results in an average FC/FS ratio of 0.17:1, strongly indicating fecal inputs from wildlife or sources other than human domestic waste. The average monitoring well FC/FS ratio was 0.04. FC/FS ratios for the various wells and canal site locations are indicated in Table 16. These data all indicate low ratios, and the dominance of non-human inputs. Interestingly, the highest well ratios were found in wells G13, G14, and G15, the farthest wells from the drainfield at the Groseclose site, and the front yard control well G12 (Table 16). Also, the mean canal FC/FS ratio (0.17:1) was higher than the mean ratio in the wells (0.04:1).

Table 15. Fecal Coliform and Fecal Streptococcus Counts During the Study Period at the Downstream Canal Sites, NC-1 through NC-4, Phase I (col./100 ml)

Fecal Coliform

I D	15 May 90	9 Jun 90	5 Jul 90	28 Aug 90	Geo. Mean
NC-1	50	500	60	750	183
NC-2	50	415	140	1180	242
NC-3	30	80	110	340	97
NC-4	10	49	10	400	37

Fecal Streptococcus

I D	15 May 90	9 Jun 90	5 Jul 90	28 Aug 90	Geo. Mean
NC-1	700	3770	3000	2700	2150
NC-2	1100	910	2000	12000	2214
NC-3	500	1490	2800	6800	1941
NC-4	1400	2680	1600	9600	2755

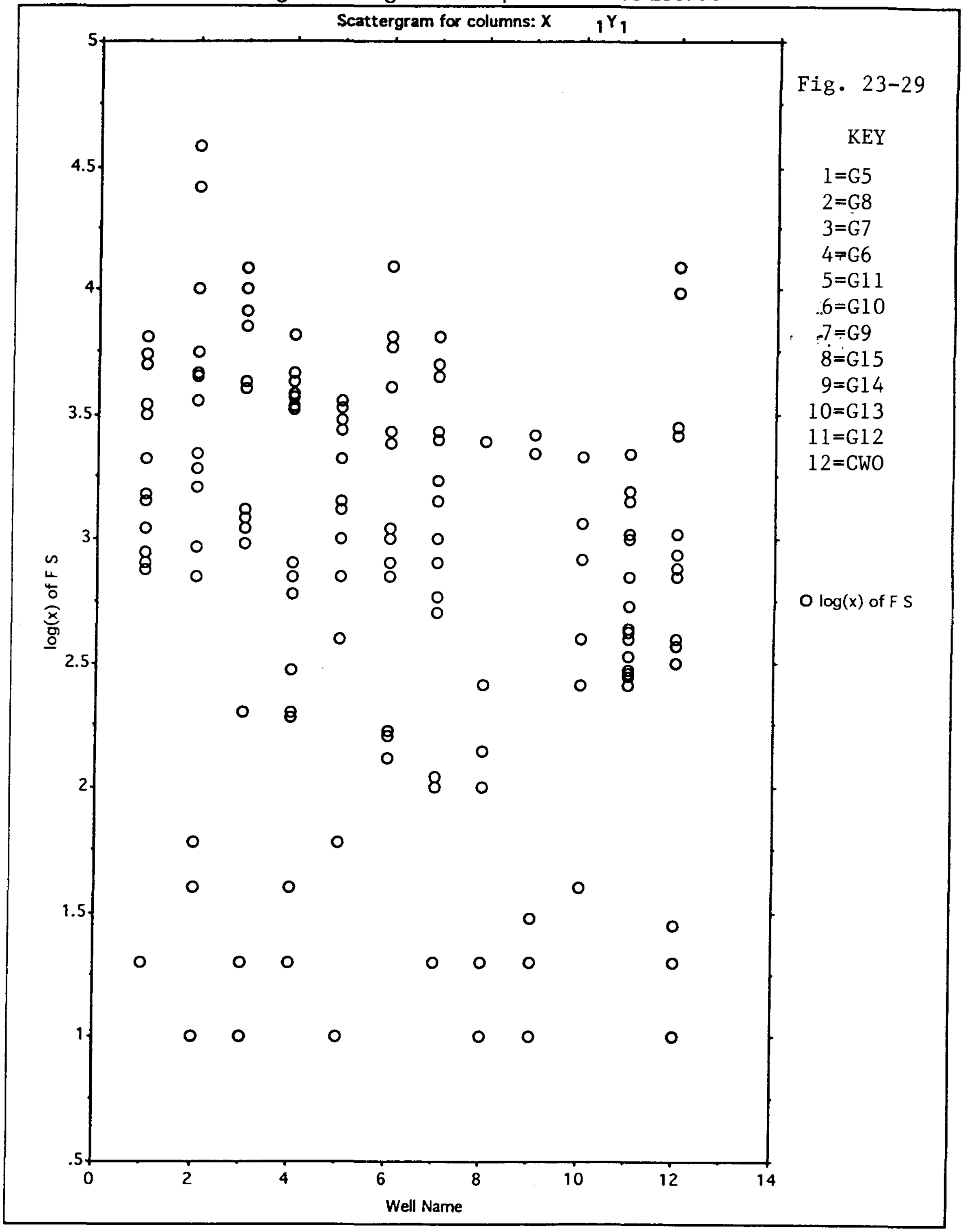
Assessment of Water Quality Results

Scatterplots showing levels of various parameters with increasing distance from the drainfield at each residence generally show decreasing concentrations. Groseclose residence scatterplots are shown in Figures 26 through 32. Jones residence scatterplots are presented in Figures 33 through 37. Reference numbers 11 (J-5), 9 (BC-5), 10 (J-6), and 12 (CW0) at the Jones site may be misplaced, as they represent front yard and control site data, but the plots still indicate the general decreasing trend. ANOVA's run on well concentration data between various well combinations verify the scatterplot trends. Significant differences ($p \leq 0.05$) between selected wells and well groups for various parameters determined from ANOVA analysis, are summarized in Appendix I and J.

Table 16. FC/FS Ratios in Monitoring Wells and Canals

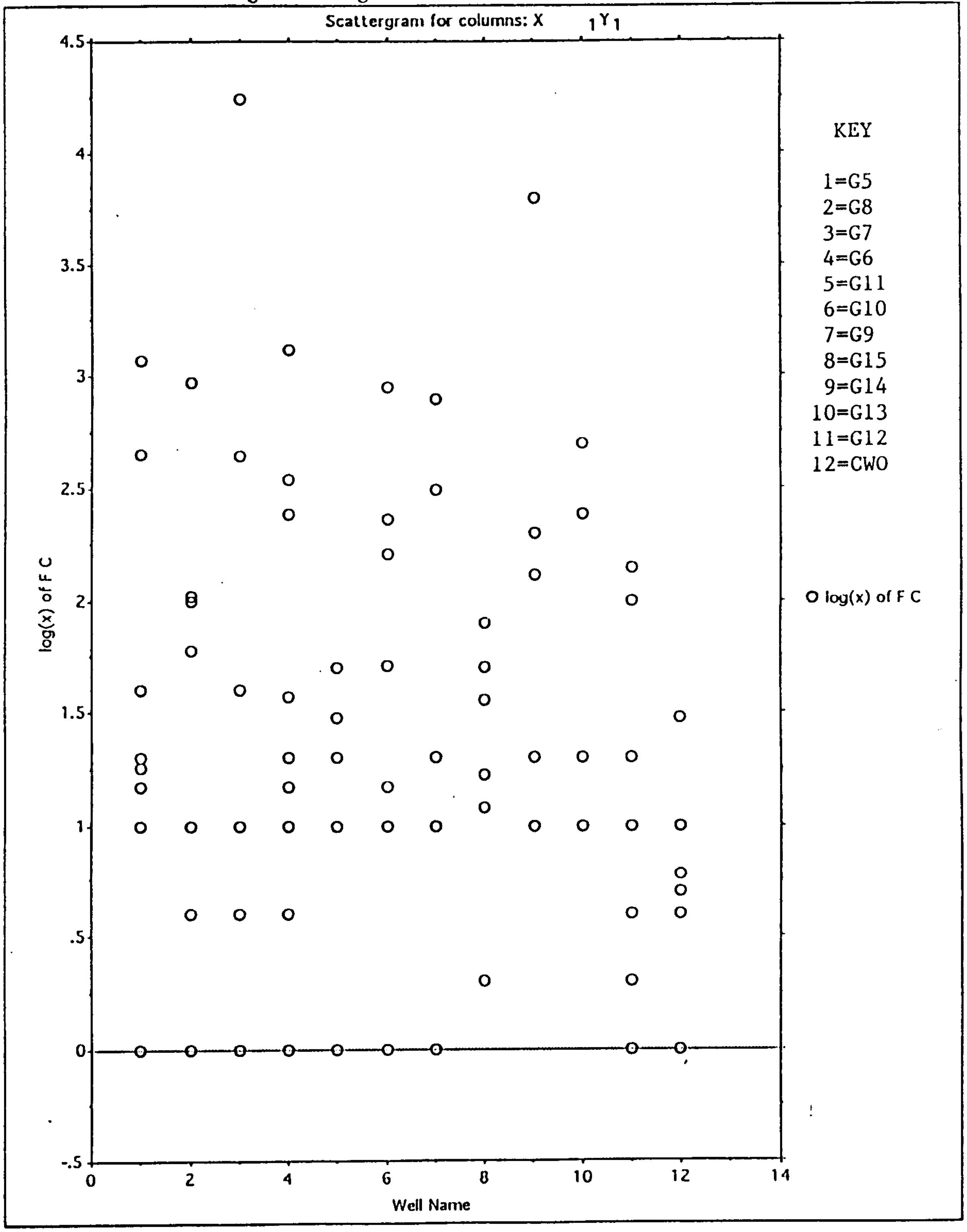
Station ID	FC/FS Ratios	Station ID	FC/FS Ratios
	WELLS	CANAL	
J-5	0.004	C0	0.147
J-7	0.005	C1	0.149
J-8	0.011	C2	0.188
J-9	0.008	C3	0.193
J-10	0.023	C4	0.088
J-11	0.018	C5	0.055
J-12	0.060		
J-13	0.048		
J-14	0.039		
J-6 (background)	0.011		
BC-5(background)	0.007		
CW0 (background)	0.028		
G-5	0.013		
G-6	0.022		
G-7	0.016		
G-8	0.005		
G-9	0.014		
G-10	0.016		
G-11	0.011		
G-13	0.101		
G-14	0.247		
G-15	0.104		
G-12(background)	0.100		

Scattergram of Log Fecal Strep - Groseclose Location



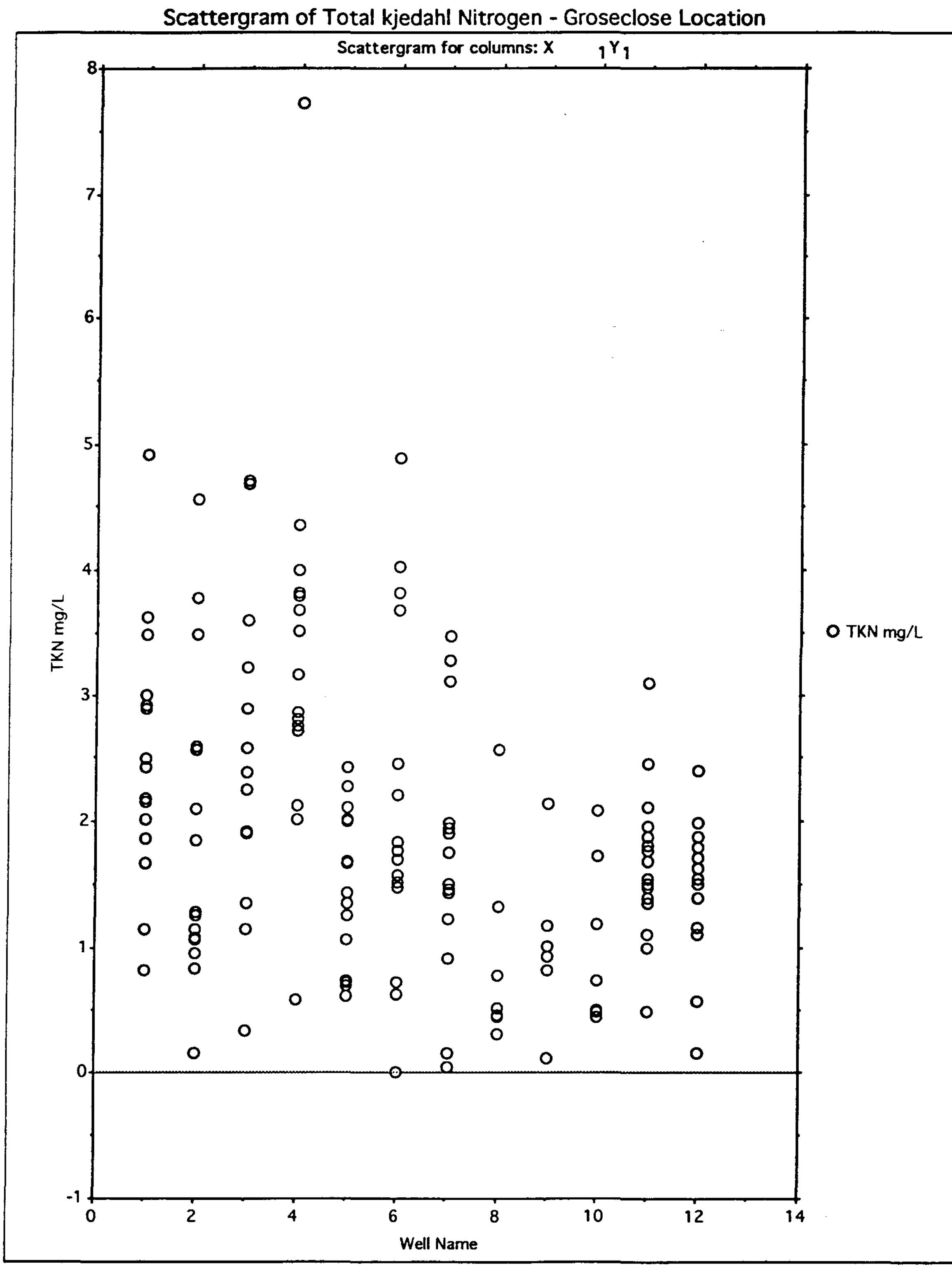
AYRES ASSOCIATES	DRAWN BY:	DATE:	SCATTERPLOT OF LOG FS vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE	FIGURE: 26
	CHECKED BY:	DATE:		
	APPROVED BY:	DATE:		

Scattergram of Log Fecal Coliform - Groseclose Location



AYRES ASSOCIATES	DRAWN BY: TPB DATE: 8/92 CHECKED BY: DATE: 2/93 APPROVED BY: DATE:	SCATTERPLOT OF LOG FC vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE

FIGURE: 27



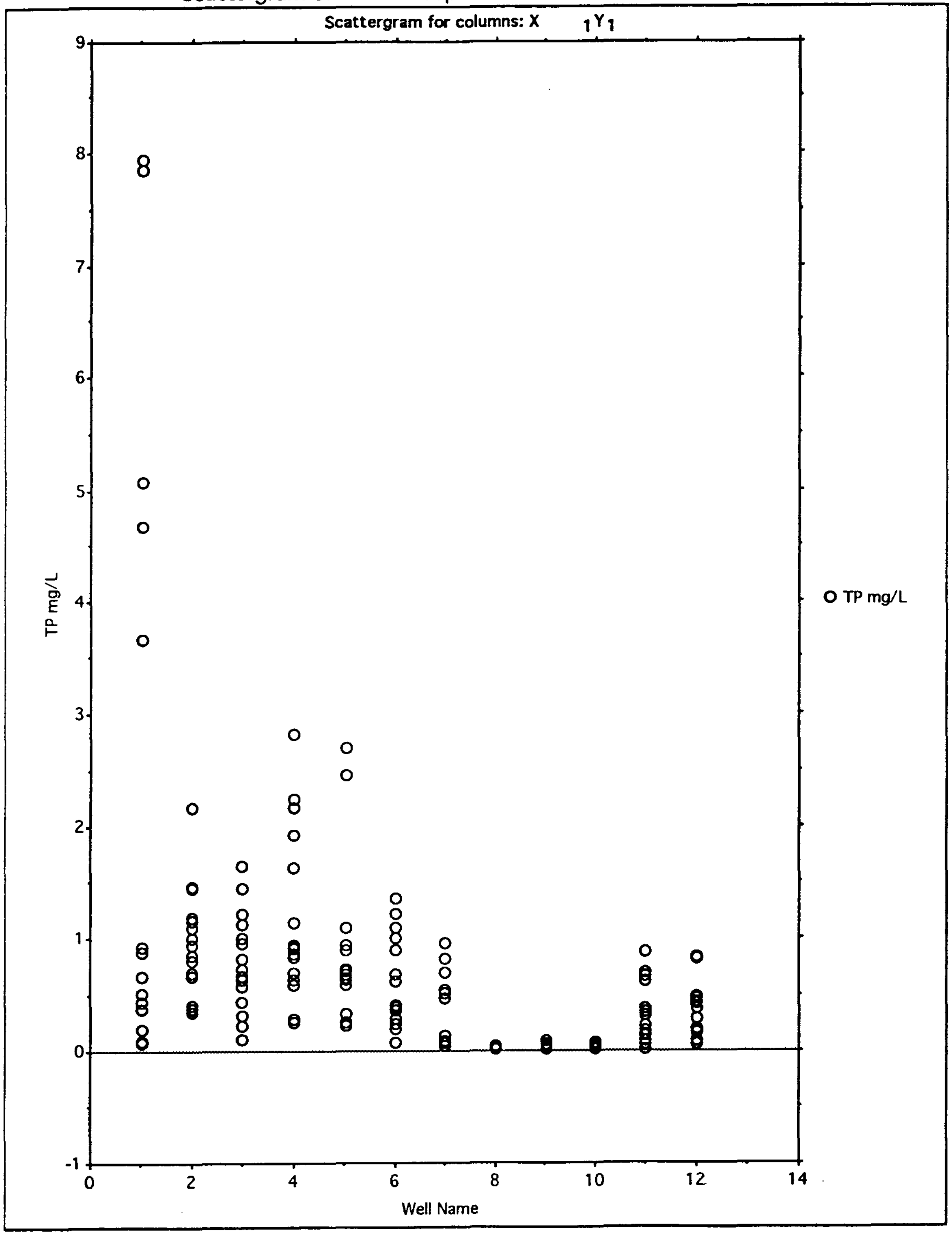
AYRES ASSOCIATES

DRAWN BY:	DATE:
CHECKED BY:	DATE:
APPROVED BY:	DATE:

SCATTERPLOT OF TKN vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE

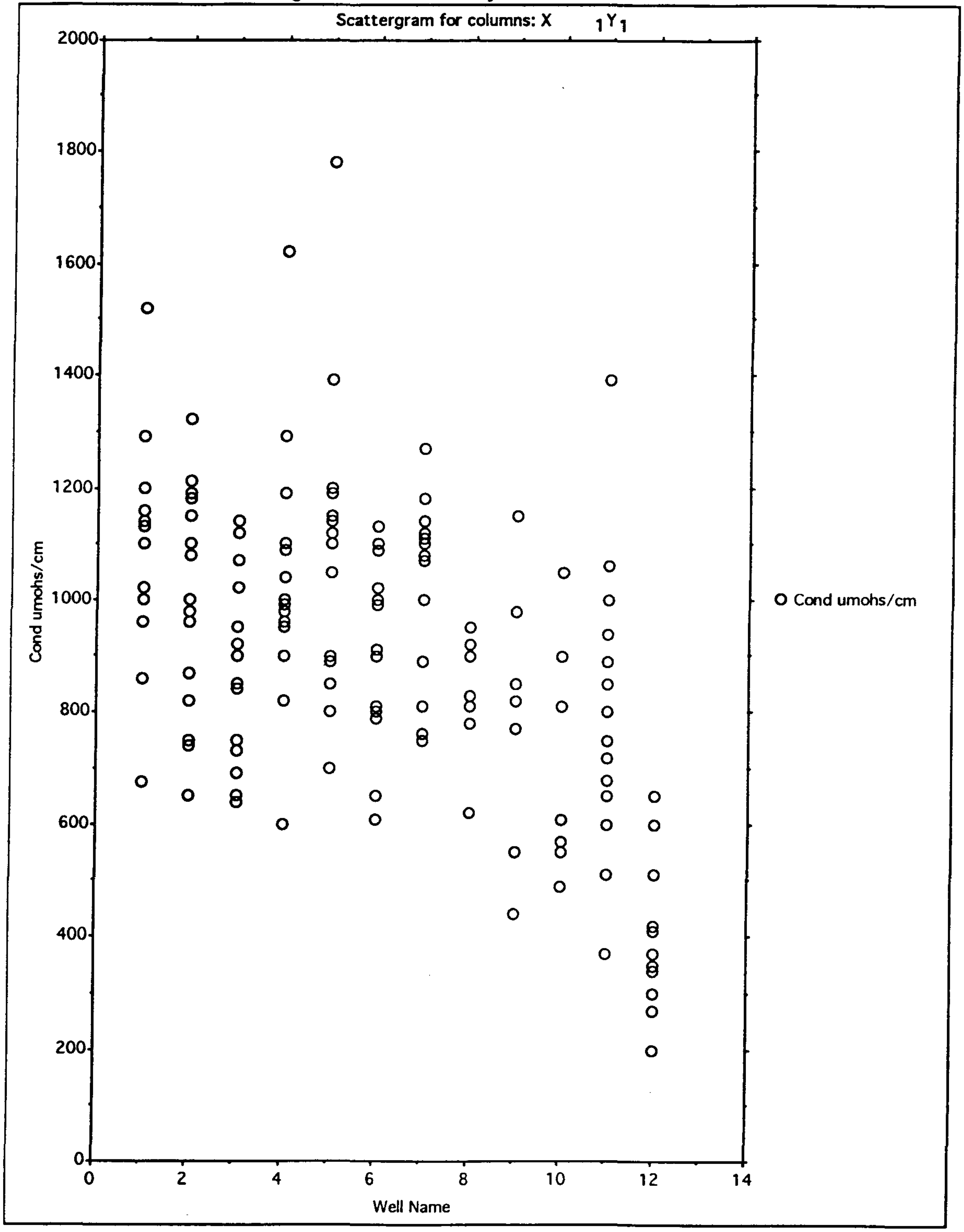
FIGURE: 28

Scattergram of Total Phosphorus - Groseclose Location



AYRES ASSOCIATES	DRAWN BY: TJB	DATE: 3/92	SCATTERPLOT OF TP vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE	FIGURE: 29
	CHECKED BY: TMN	DATE: 2/93		
	APPROVED BY:	DATE:		

Scattergram of Conductivity - Groseclose Location

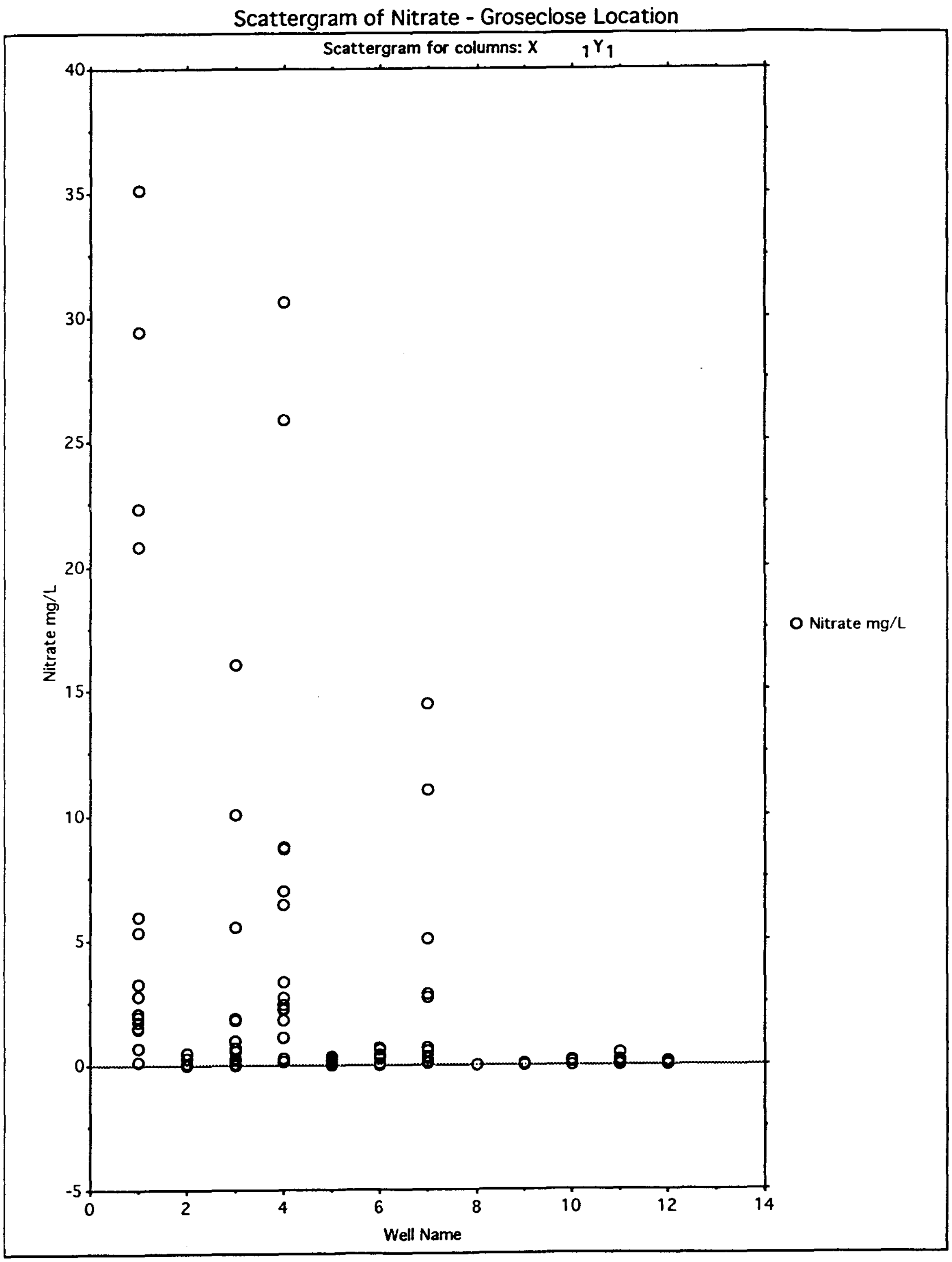


AYRES ASSOCIATES

DRAWN BY:	DATE:
[illegible]	12/[illegible]
CHECKED BY:	DATE:
[illegible]	2/[illegible]
APPROVED BY:	DATE:

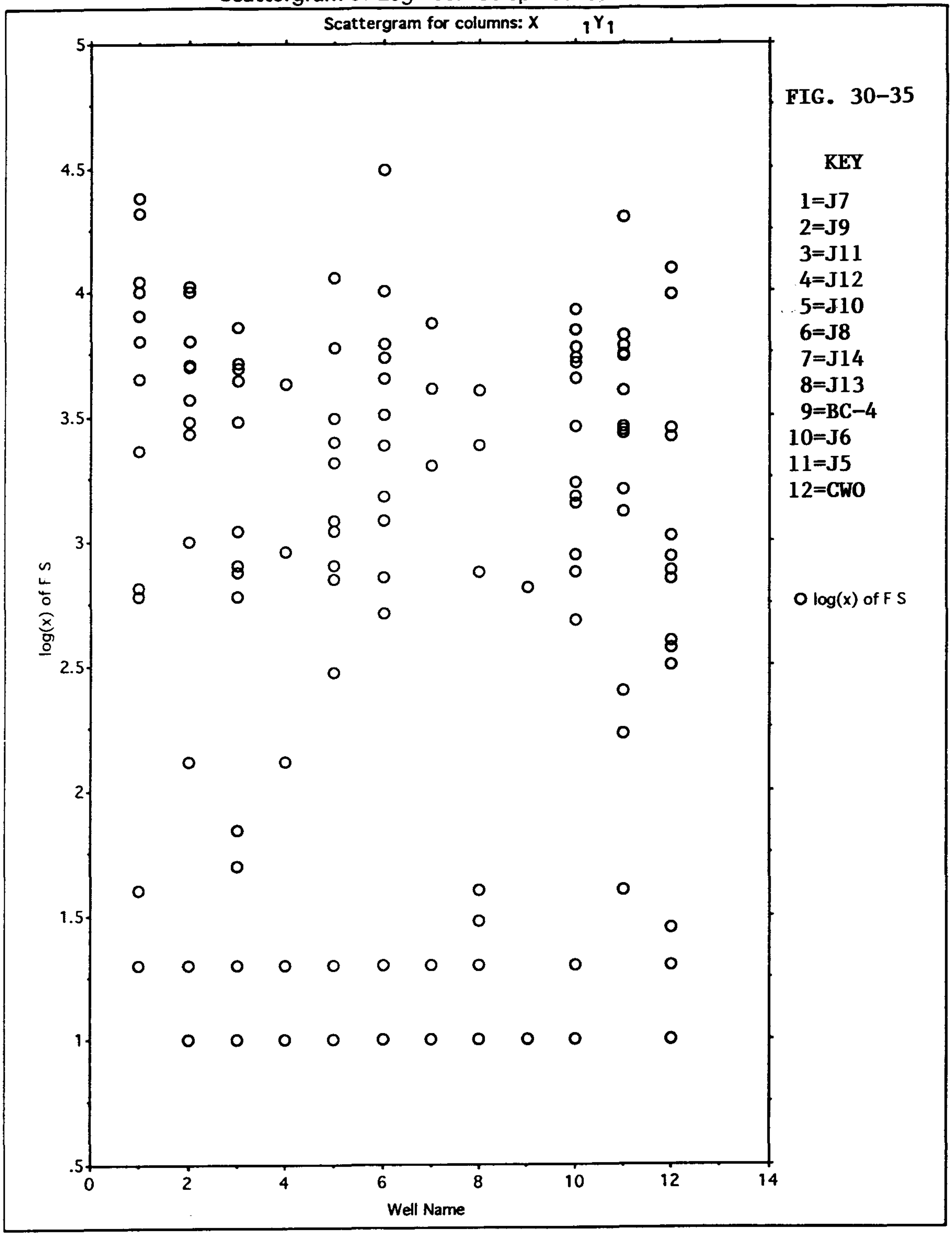
SCATTERPLOT OF CONDUCTIVITY vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE

FIGURE: 30



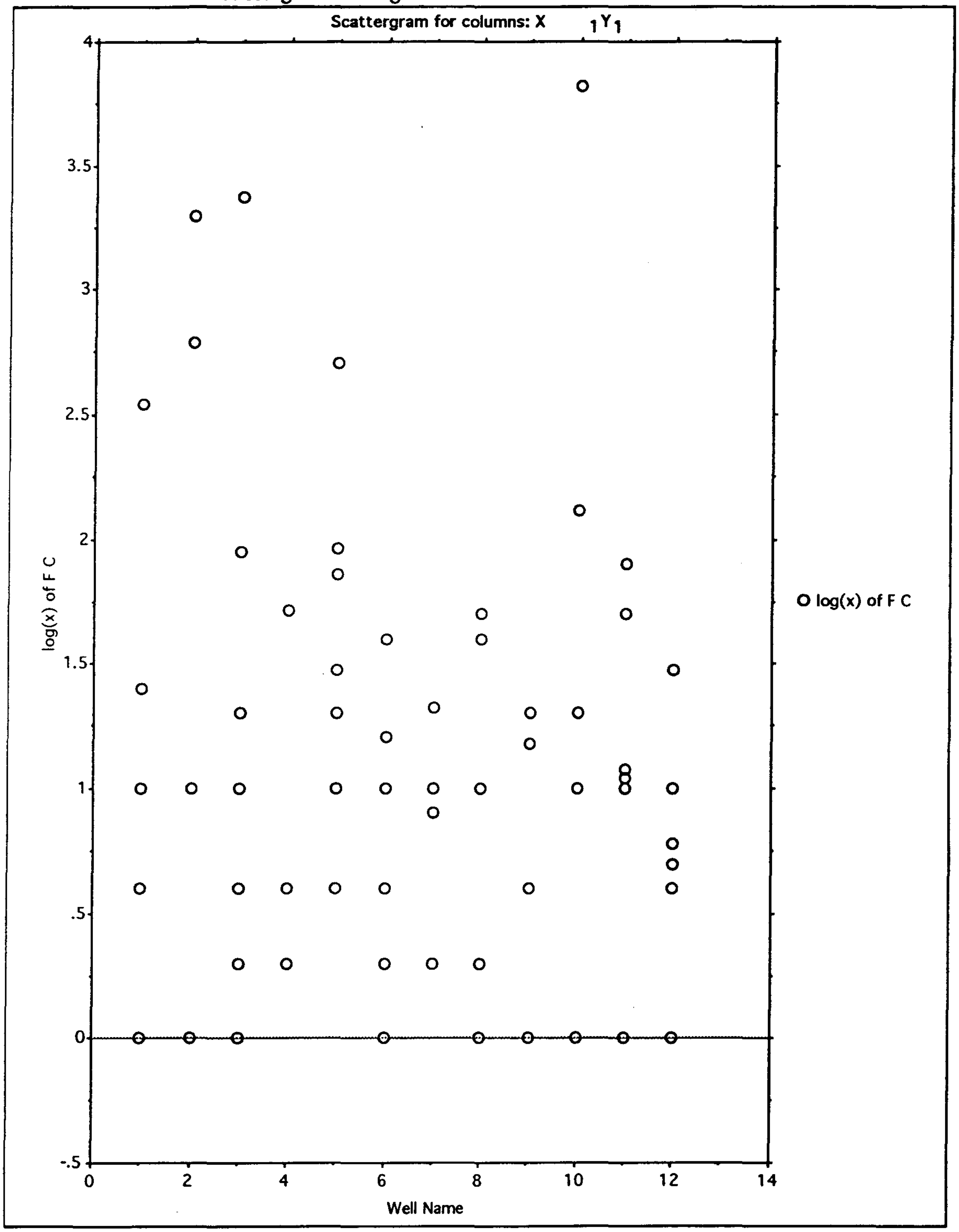
AYRES ASSOCIATES	DRAWN BY: TVB — DATE: 12/92 CHECKED BY: — DATE: 2/93 APPROVED BY: — DATE:	SCATTERPLOT OF NITRATE, NITRITE-NITROGEN vs. DISTANCE FROM DRAINFIELD AT THE GROSECLOSE SITE	FIGURE: 31

Scattergram of Log Fecal Strep - Jones Location



AYRES ASSOCIATES	DRAWN BY: / DATE: 12/82 CHECKED BY: / DATE: APPROVED BY: / DATE:	SCATTERPLOT OF LOG FECAL STREPTOCOCCUS vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE	FIGURE: 32

Scattergram of Log Fecal Coliform - Jones Location



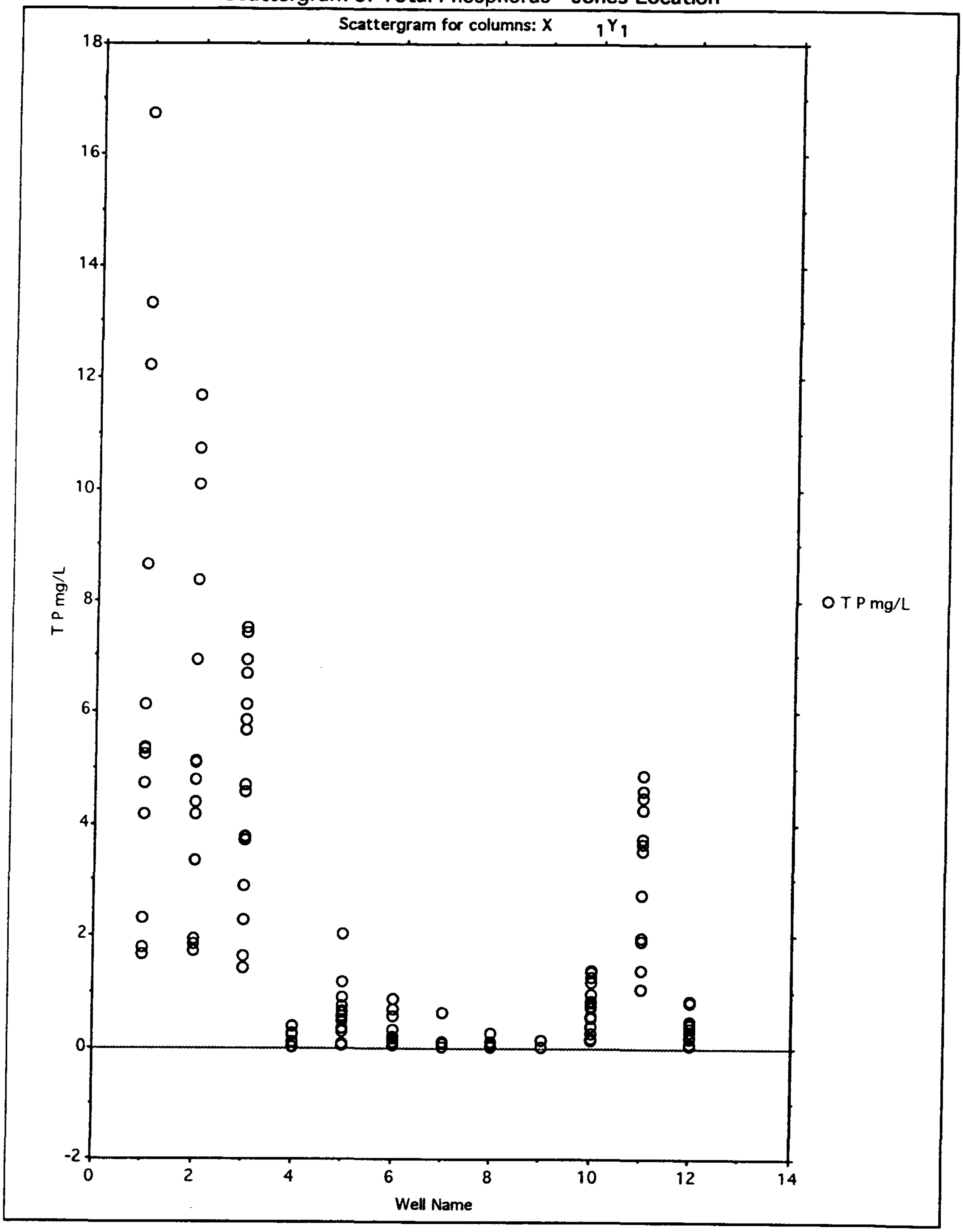
AYRES ASSOCIATES

DRAWN BY:	DATE:
CHECKED BY:	DATE:
APPROVED BY:	DATE:

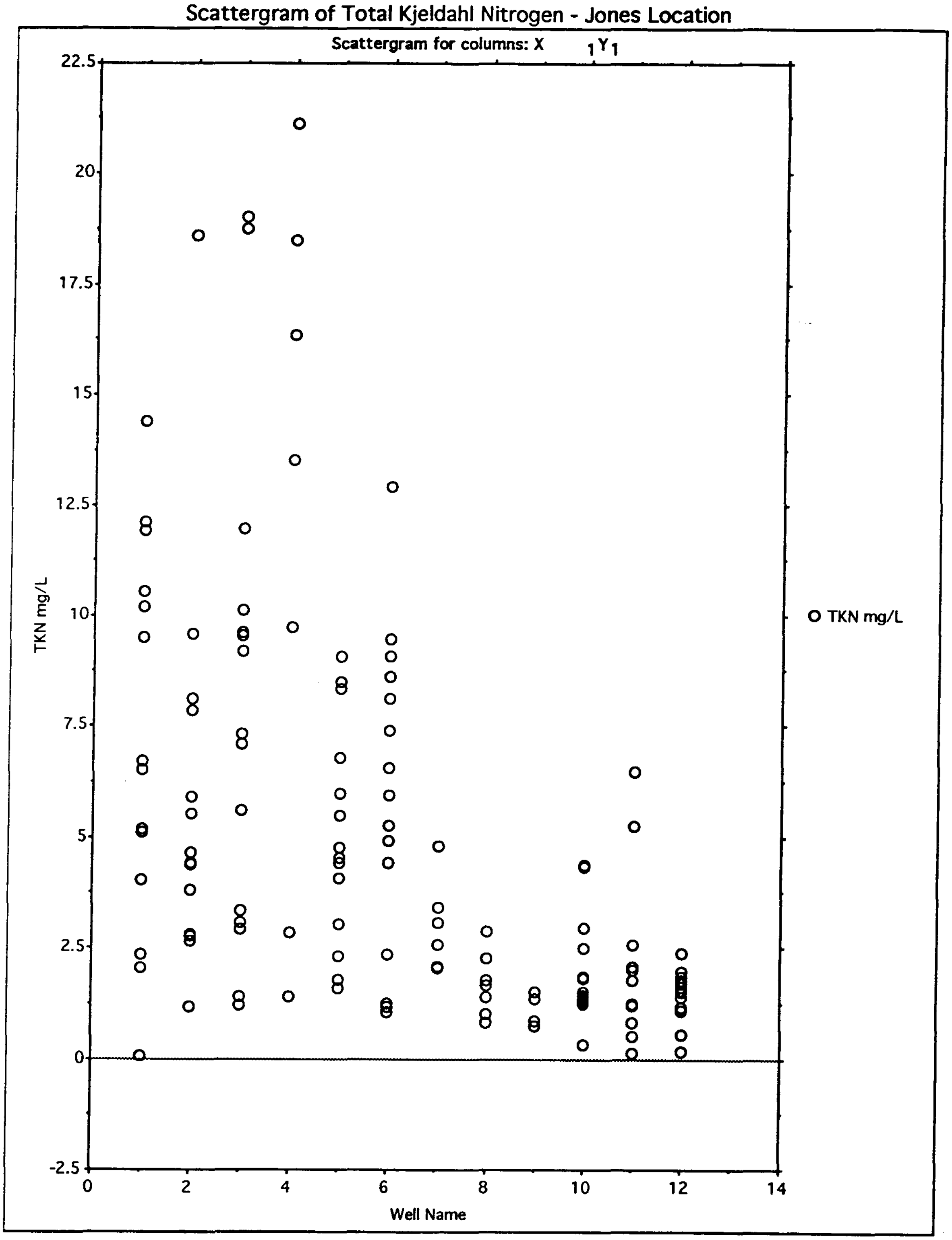
SCATTERPLOT OF LOG FECAL COLIFORMS vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE

FIGURE: 33

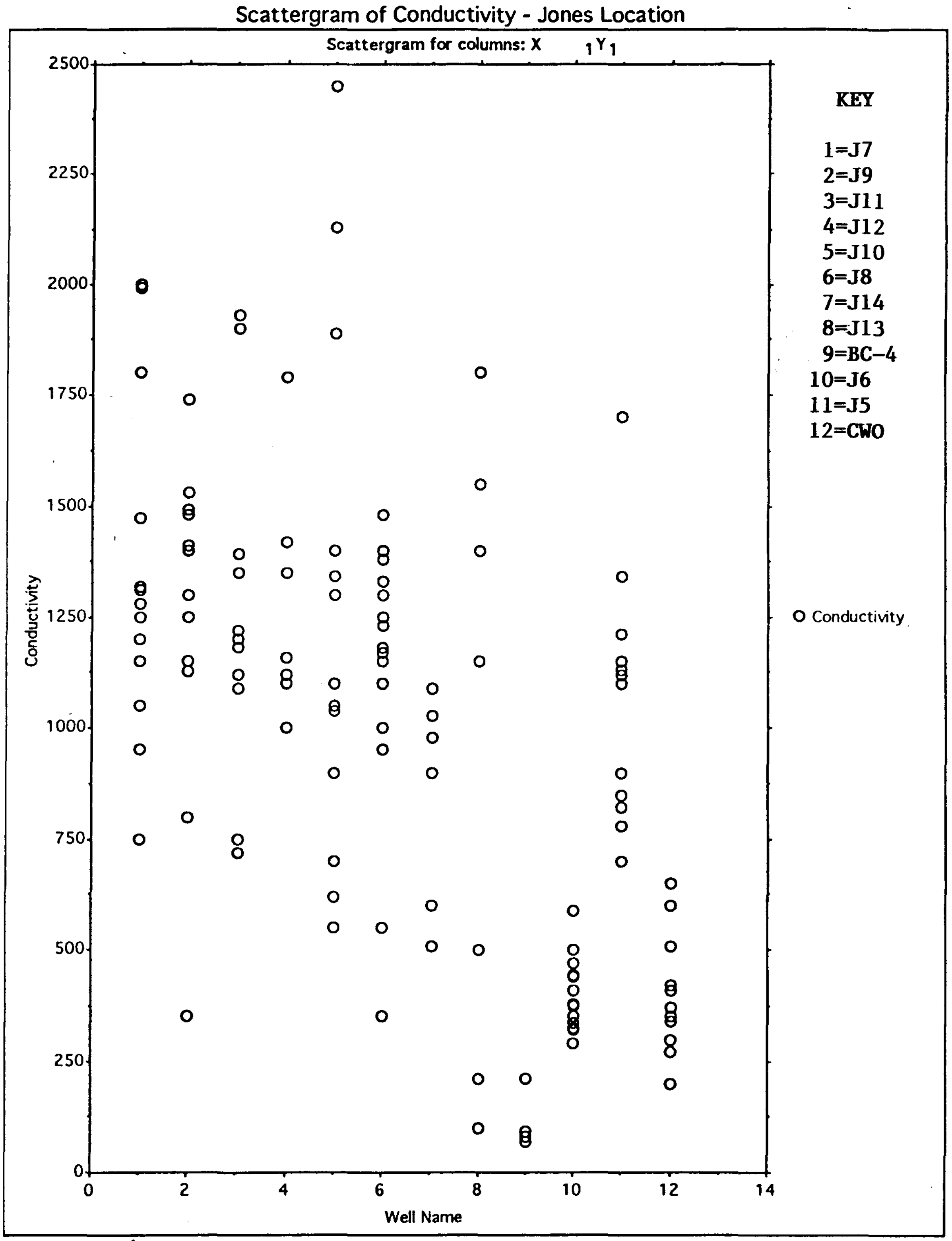
Scattergram of Total Phosphorus - Jones Location



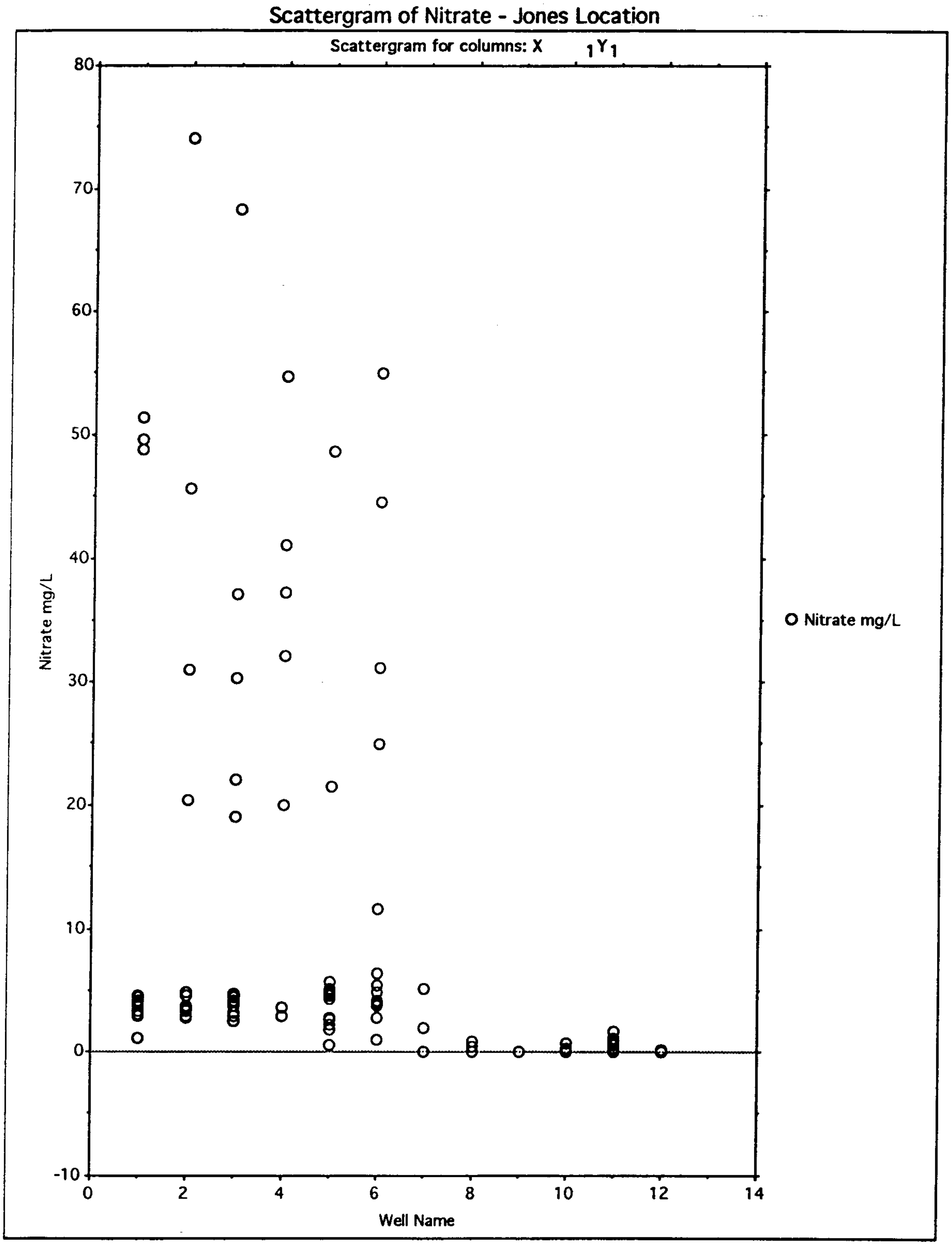
AYRES ASSOCIATES	DRAWN BY: DM DATE: 12/92 CHECKED BY: DATE: 2/93 APPROVED BY: DATE:	SCATTERPLOT OF TP vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE	FIGURE: 34



AYRES ASSOCIATES	DRAWN BY: DATE:	SCATTERPLOT OF TKN vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE	FIGURE: 35
	CHECKED BY: DATE:		
	APPROVED BY: DATE:		



AYRES ASSOCIATES	DRAWN BY:	DATE:	SCATTERPLOT OF CONDUCTIVITY vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE
	CHECKED BY:	DATE:	FIGURE: 36
	APPROVED BY:	DATE:	



AYRES ASSOCIATES	DRAWN BY: JSD	DATE: 2/42	SCATTERPLOT OF NITRATE, NITRATE-NITROGEN vs. DISTANCE FROM DRAINFIELD AT THE JONES SITE	FIGURE: 37
	CHECKED BY:	DATE: 2/43		
	APPROVED BY:	DATE:		

Strongest decreasing trends with distance from the drainfield were found for conductivity, TKN, nitrate, fecal coliform and TP at the Jones locations; while TP, TKN, conductivity, and nitrate exhibited the strongest trend at the Groseclose site. Dissolved oxygen and log fecal streptococcus levels showed no relationship ($p \geq 0.05$) with distance.

Assessment of Groseclose Site Results: Septic system drainfields over suitable unsaturated soil depth typically convert nearly all organic and ammonia nitrogen to nitrate nitrogen before effluent reaches groundwater.

At both the Groseclose and Jones residences, significant concentrations of ammonia nitrogen were measured downgradient of the drainfields. At the Groseclose site total kjeldahl nitrogen (TKN) data showed significant reductions with distance from the drainfield. TKN was significantly ($p \geq 0.05$) higher at G-6 (mean = 3.30 mg/l), near the end of the blackwater drainfield, than any other well except the adjacent G-7 (2.62 mg/l). These results are not surprising considering that the Groseclose drainfield was often less than two feet from the water table. These TKN levels seem low relative to the Jones site however, where a greater unsaturated thickness existed yet even higher levels of TKN were detected. The well group G-6, G-7, G-8 mean TKN level (2.60 mg/l) was significantly higher ($p \geq 0.05$) than G-13, G-14, G-15 (0.99 mg/l), G-12 (1.66 mg/l), or CW0 (1.51 mg/l) at the Groseclose site. The G-9, G-10, G-11 well row, located approximately 20 feet from the drainfield, also exhibited significantly ($p \geq 0.05$ higher TKN levels (1.76 mg/l) than G-13, G-14, and G-15. The G-13, G-14, G-15 mean TKN concentration was not significantly ($p \geq 0.05$) different from the background level at G-12. Some of this reduction is no doubt due to dilution.

At the Groseclose site, nitrate was significantly ($p \leq 0.05$) higher at G-5, between the drainfields, than at any other well (mean = 8.98 mg/l). Well G-6, at the east end of the first row, was also high (mean = 6.63 mg/l), and reflects loading from the blackwater drainfield. This well had significantly ($p \leq 0.05$) higher concentrations of nitrate than other wells at the end of the two drainfields. A significant ($p \leq 0.05$) difference existed between nitrate levels in G-6, six feet from the edge of the drainfield, and levels in G-9 (2.58 mg/l), or G-10 (0.26 mg/l), located 20 feet from the edge of the drainfield. There were also significant ($p \leq 0.05$) concentration differences between G-6, G-7, G-8 (3.06 mg/l) well group data and G-9, G-10, G-11 (0.95 mg/l); and G-13, G-14, G-15 (0.037 mg/l), located 50 feet away. This indicates significant attenuation in the 14 feet between G-6, G-7, G-8, and G-9, G-10, G-11, and further reduction in the 30 feet from G-9, G-10, G-11 and G-13, G-14, G-15. Since nitrate moves readily with groundwater, this reduction was thought most likely due to dilution, although some denitrification may have contributed to the reduction.

Total phosphorus (TP) concentrations also decreased significantly downgradient of the Groseclose drainfields. The mean G-13, G-14, G-15 TP concentration at the Groseclose site (0.037 mg/l) was below G-12 (0.91 mg/l) or CWO (0.62 mg/l) background concentrations, indicating a reduction to background in a distance of 50 feet. Phosphorus (TP) at the Groseclose site was significantly ($p \leq 0.05$) higher in G-5 (mean = 2.83) than any other well, and was not significantly different ($p \leq 0.05$) on either the blackwater or graywater side of the first row of wells. The G-6, G-7, G-8 first row (0.97 mg/l) and G-9, G-10, G-11 second row (0.68 mg/l) mean concentration TP data were significantly ($p \leq 0.05$) higher than the G-13, G-14, G-15 concentrations (0.33 mg/l), fifty (50) feet away from the drainfield. These data indicate that TP levels were effectively reduced to background levels before reaching the canal, 65 feet from the drainfield. This was no doubt due to a combination of phosphorus attenuation by soils and dilution with groundwater.

Mean conductivity was highest at G-5 (1070 μmhos/cm) and lowest at the control site CW0 (393 μmhos/cm). The G-6, G-7, G-8 mean concentration (966 μmhos/cm) was significantly different from G-13, G-14, G-15 (778 μmhos/cm), located 50 feet from the drainfield. The G-9, G-10, G-11 mean level (979 μmhos/cm) was also significantly

($p \leq 0.05$) different from G-13, G-14, G-15, but there was not a significant difference between G-6, G-7, G-8 and G-9, G-10, G-11 or between G-13, G-14, G-15 and the background well G-12 (796 μmhos/cm), indicating negligible impact from the Groseclose drainfield, in terms of conductivity, in at least a 50 foot linear distance from the drainfield to G-13, G-14, G-15. These results suggest that the development of the subdivision may have impacted the overall conductivity concentrations of groundwater, since the undeveloped control area shows such significantly lower conductivity values.

Bacterial data were highly variable in the well samples. The mean fecal streptococcus level was highest in well G-8 (1746 col./100 ml geometric mean), while the mean fecal coliform value was highest in wells G-14 and G-15 (44 col./100 ml geometric mean). Lowest respective geometric mean levels were found in G-14 (178 col./100 ml) and G-8 (8 col./100 ml). G-6, G-7, G-8 and G-9, G-10, G-11 log mean fecal streptococcus levels were significantly ($p \leq 0.05$) lower than G-13, G-14, G-15. However, none of the well group mean levels were significantly ($p \leq 0.05$) different from the G-12 control.

Fecal coliforms seemed to vary in an inverse manner with distance from the drainfield, as the G-6, G-7, G-8 and G-9, G-10, G-11 log mean level was significantly ($p \leq 0.05$) lower than G-13, G-14, G-15, and G-13, G-14, G-15 was significantly ($p \leq 0.05$) lower than the control. The fecal coliform data therefore indicate no discernible septic system impact on downgradient wells at the Groseclose site. The number of bacteria found in all wells make this data difficult to interpret, however. The variability in bacteria data may be attributed to the Groseclose residence using canal water for lawn irrigation, as high levels of fecal coliforms and fecal streptococcus were found in the canals.

Assessment of Jones Site Results: At the Jones residence, fecal coliform geometric mean levels varied from 6 col./100 ml (J-12, J-13) to 20 col./100 ml (J-10) in the wells, with higher values occurring later in the study. The background site, J-6, also exhibited a high geometric mean of 14 col./100 ml, indicating possible inputs from road runoff and wildlife. There were no significant ($p \leq 0.05$) differences between wells or well groups for log fecal coliform levels at the Jones residence, as levels were generally low.

Fecal streptococcus levels (geometric mean) ranged from 100 col./100 ml (J-12) to 1896 col./100 ml (J-5). J-5 is located in the front yard near the graywater septic tank. The background well (J-6) also exhibited a high geometric mean value of 1252 col./100 ml. This site is located in a swale area near the road and high bacterial levels are possibly contributed from runoff inputs and local animal populations. There were a considerable number of significant ($p \leq 0.05$) differences between individual wells at the Jones residence (Appendix J), but no significant differences between well groups. The individual well differences are difficult to interpret, but levels were highest at the western side of the drainfield and were near CW0 and BC 5 background levels at J-8 and J-10, approximately 15 feet from the drainfield.

Well comparisons with significant difference in mean total kjeldahl nitrogen (TKN) levels also existed at the Jones site. TKN ranged from an average of 1.69 mg/l at J-13 to 11.92 mg/l at J-12. The mean J-7, J-12, J-9, J-11 (7.73 mg/l) concentration was significantly ($p \leq 0.05$) greater than J-8, J-10 (5.49 mg/l) and J-8, J-10 was significantly greater than mean concentrations at the background sites J-6 (1.94 mg/l), CW0 (1.51 mg/l) or the more distant J-13 (1.69 mg/l). These data indicate impact on J-8, J-10, fifteen feet from the drainfield, but not on J-13, fifty feet away. Considering that the unsaturated thickness at the Jones drainfield was in excess of 2 feet, these results are somewhat surprising.

Well comparisons with significant differences ($p \leq 0.05$) in mean nitrate-nitrite levels were also observed at the Jones site, and very high nitrate concentrations were measured. Average nitrate-nitrite nitrogen (NO_3,NO_2-N) at the Jones residence ranged from 0.15 mg/l at the background well (J-6) to 27.4 mg/l at J-12, near the blackwater drainfield. The high nitrate and TKN levels at J-12 indicate high nitrogen loading near that location. ANOVA analysis indicated significant differences ($p \leq 0.05$) between J-7, J-12, J-9, J-11 (16.1 mg/l) and J-8, J-10 (11.1 mg/l) mean levels and background levels at J-6 (0.15

mg/l) and CWO (0.06 mg/l), indicating significant impact within 15 feet of the drainfield. The mean level at J-13 (0.21 mg/l), fifty (50) feet from the drainfield, was similar to the J-6 background level.

Average total phosphorus (TP) concentrations at the Jones site ranged from 0.16 mg/l at J-12 to 7.22 mg/l at J-7. There were many individual well significant ($p \leq 0.05$) differences (Appendix J), but it is important to note the mean J-7, J-12, J-9, J-11 level (5.07 mg/l) was significantly ($p \leq 0.05$) higher than the mean J-8, J-10 level (0.45 mg/l), and the J-8, J-10 mean concentration was lower than the background well (J-6) concentration (0.76 mg/l), indicating significant phosphorus attenuation within 15 feet of the drainfield. The J-5 concentration (3.19 mg/l) near the graywater drainfield in the front yard was significantly ($p \leq 0.05$) higher than the J-8, J-10; J-13; J-6; CWO, and J-14 concentrations, reflecting some phosphorus input from the graywater system.

The high TKN nitrogen and total phosphorus concentrations at the Jones residence are disturbing considering the relatively high unsaturated zone thickness below the blackwater drainfield in the Jones backyard. This unsaturated thickness was based on measurements of the infiltrative surface elevation at OPB, the observation port for the blackwater drainfield (See Figure 8). The ground surface over the drainfield sloped considerably downward away from the house towards the back lot line, and it was suspected that perhaps the drainfield infiltrative surface also sloped downward. Thus, the infiltrative surface may have been lower at the far end of the drainfield and this would have resulted in a reduced unsaturated thickness in that zone. Further investigation would be required to confirm this, but this may be an explanation for the results obtained.

Many well comparisons indicated significant ($p \leq 0.05$) differences in mean conductivity levels at the Jones site (Appendix J). Mean conductivity levels ranged from 411 μmhos/cm at the background well (J-6) to 1341 μmhos/cm at J-7. J-7, J-12, J-9, J-11 (1287 μmhos/cm) and the J-8, J-10 (1186 μmhos/cm) mean group levels were significantly ($p \leq 0.05$) greater than background levels at J-6 and CW0 (393 μmhos/cm), indicating impact from the drainfield.

Fecal coliform levels were slightly lower than at the Groseclose site, and there were no significant ($p \leq 0.05$) differences between wells or well groups, indicating minimal OSDS impact. There were many significant differences between individual wells and fecal streptococcus data at the Jones site, but no significant ($p \leq 0.05$) differences between well groups varying distances from the drainfield. Also, the control well, J-6, exhibited a high geometric mean (1252 col./100 ml) and was greater than most of the wells. At both residential sites, there was no statistically significant ($p \leq 0.05$) difference between well bacterial levels and water table height, temperature, or rainfall.

Assessment of Canal Water Quality Results: The canal water had low concentrations of NO_3, NO_2-N, TKN, TP, and COD for the entire length of canal sampled (Table 11). There was no significant difference ($p \leq 0.05$) between water samples collected above, next to, and below the study sites (C5-C0) for NO_3, NO_2-N, TKN, TP or turbidity. The COD concentrations did increase slightly as the water flowed east to west along the study sites, probably because of the increase in organic matter and runoff along the direction of canal flow. As the canal flows westward, the amount of organic muck increases because of the abundant vegetation growing in the canal. Dissolved oxygen was variable, ranging from an average of 5.4 mg/l at C5 to 8.6 mg/l at C4. With the exception of C4, conductivity tended to decrease in a westerly direction with increasing flow and dilution. Color, however, which increased from C0 to C5, reflected increasing seepage and runoff in a westerly direction.

Multiple regressions were run on mean canal and station C-3 data to determine the importance of selected independent variables in predicting parameter concentrations. These analyses are presented in Appendices K and L, respectively. The independent variables listed were Rain 1 day (1 day rain level prior to sampling), Rain 7 day (7 day cumulative rain total prior to sampling), stage height and water temperature. The multiple regression parameters were not significant, although they did suggest possible relationships between temperature and stage height and fecal coliform and log fecal streptococcus levels, particularly at station C-3. Additionally, C-3 station analysis indicated that stage height appeared significant to nitrate and TKN levels. Rain 1 and 7 indicated possible significance in predicting mean canal total phosphorus and Rain 7 was a possible significant predictor of conductivity.

Because the multiple regression data did not reveal conclusive relationships, only possible relationships (Appendices K and L), linear regression relationships were run between the above variables and parameters using mean canal and station C-3 data. These analyses are shown in Appendices M and N, respectively.

Results of linear regression analyses indicated that a significant inverse relationship existed between the average canal log fecal streptococcus levels and mean canal stage height ($p \leq 0.02$), and a significant positive relationship existed between the average canal log FS and mean canal temperature. Using only station C3 data, located behind the Groseclose residence, significant linear relationships were found between log fecal coliform and stage height (inverse, $p < 0.06$); between log fecal coliform and temperature ($p \leq 0.05$); between log fecal streptococcus and stage height (inverse, $p < 0.024$); between TKN and stage height (inverse, $p < 0.07$); and between conductivity and temperature ($p \leq 0.029$).

Fecal coliform counts in surface water obtained from the canals were considerably higher than in the groundwater obtained from the monitoring wells. The fecal coliform counts at Canal 66 sampling site averaged between 157 and 755 fecal coliform col./100 ml (geometric mean).

Fecal streptococcus levels were high and variable in surface water obtained from canals. Canal 66 geometric mean levels ranged from 2585 (C5) to 9240 (C-3) Col./100 ml, while stations NC-1 through NC-4 exhibited lower levels ranging from 1941 (NC-3) to 2755 (NC-4) Col./m (geometric mean).

Bacterial levels found in canals corresponded fairly well with other published data for levels found in surface waters under similar environmental conditions. Cowan et al., (1989) found fecal coliform levels of up to 2400 MPN/100 ml in a multi-use catchment basin in Newfoundland, with most samples containing less than 500 MPN/100 ml. In river samples in Kentucky, Geldreich (1976) reported average fecal coliform and fecal streptococcus densities of 1443 col./100 ml and 4271 col./100 ml, respectively. Bacterial levels ranging approximately one order of magnitude lower than those found in this study have been frequently reported for rivers, lakes and streams. Factors such as geographical area, temperature, flow rate, and land use must be taken into account when making accurate comparisons between surface water densities found in this study and densities reported in other studies. The relatively high temperatures and slow rate of flow found in this study are conducive to growth and recovery of organisms in water samples. As discussed previously, linear regression analysis indicated a significant ($p \leq 0.05$) inverse relationship existent between the mean canal fecal streptococcus level (log) and stage height (NGVD). Also log fecal streptococcus level were significantly related to temperature ($p \leq 0.05$). Using only Station C3 data, significant inverse relationships were found between the log fecal coliform level and stage height ($p \leq 0.06$) and between log fecal streptococcus and stage height ($p \leq 0.007$); while significant positive relationships were observed between log fecal coliform and temperature ($p \leq 0.05$) and between log fecal streptococcus and temperature ($p \leq 0.08$). Rainfall was not significantly ($p \leq 0.05$) related to canal bacterial levels.

The FC/FS ratio in canal 66 near the OSDS research sites averaged 0.17:1, which is in agreement with most published literature for surface waters that do not receive significant loading from sources containing human waste. This ratio is higher than the well mean ratio, but still very low. A ratio of greater than 4:1 has been reported to be indicative of a human source of contamination, whereas a ratio of 0.71:1 or less indicates a wildlife or animal source (Standard Methods, 1985). Ratios in between are ambiguous, but the higher the ratio, the more likely that recent contamination from a human source has occurred. The ratio of FC/FS in stormwater runoff is usually 0.7:1 or less, and is often 0.4:1 or less (Geldreich, 1976). The ratio of 0.17:1 in canal 66 may, therefore, indicate that stormwater runoff is also a considerable source of bacteria.

The high correlation found between fecal coliform and fecal streptococcus levels in canal 66 ($r = 0.91$; $p<0.01$) during Phase I sampling indicates that the two bacterial groups may originate from common non-human sources. In general, fecal coliform and fecal streptococcus levels varied from an average low at canal station C5 to a peak C3, and then decreased or remained similar from C3 to C0, although not returning to C5 levels. C1 levels were slightly higher than C0 or C2 levels.

Ten to fifteen ducks were owned by a resident of the home adjacent to and upstream from the Groseclose residence, and they have often been observed in the canal. Also, a chicken coop containing ten to twenty chickens was located across the canal from this site. It is highly likely, therefore, that the presence of the ducks and chickens are responsible for the increased bacterial densities found at the C3 station.

The FC/FS ratio at downstream canal stations NC-1 through NC-4 averaged approximately 0.7:1 during Phase I sampling, with bacterial levels showing a general decrease from NC-2 to NC-4, where canal 1 merges with Turkey Creek. This slight ratio increase from canal 66 to NC-1 through NC-4 may be due to a greater cumulative impact of OSDS from numerous sites along the canal relative to non-human inputs from runoff. Fecal coliform and fecal streptococcus levels did not correlate significantly with each other at stations NC-1 through NC-4, perhaps due to more varied inputs as flow increases downstream.

Differences in Water Quality Results With Time: Considerable differences in water quality were measured between the Phase I and Phase II sampling periods (See Appendix D). At the Groseclose Site, nitrate-nitrogen concentrations were considerably higher in monitoring wells G-5, G-6, G-7 and G-9 during the Phase II sampling (1991 sample dates). These wells were all within twenty feet of the blackwater drainfield and peak NO_3-N concentrations exceeded 30 mg/L at wells G-5 and G-6. Also, phosphorus concentrations where higher in G-5 and TKN concentrations were higher in well G-9.

At the Jones Site, nitrate-nitrogen concentrations were considerably higher in wells J-7, J-8, J-9, J-10, J-11 and J-12 during the Phase II sampling. Peak nitrate-nitrogen concentrations exceeded 50 mg/L at wells J-7, J-8, J-11 and J-12. In addition, total phosphorus concentrations were substantially higher during Phase II in wells J-7, J-9, and J-11 and TKN concentrations were higher in wells J-11 and J-12. Monitoring wells J-7 through J-12 are also located within twenty feet of the Jones blackwater drainfield.

The reason for the increased concentrations of nitrogen and phosphorus in Phase II is unknown. A check of laboratory calculations and QA/QC analyses revealed no reason to suspect laboratory errors. Phase II results were obtained one year after the end of Phase I monitoring due to delays in project contracting. Therefore, groundwater movement during that time may have resulted in the contaminant plume moving past the wells in question. Since the impacted wells were within twenty feet of the drainfields, however, it seems unlikely that it would take eight years for the plume of conservative parameters such as nitrate to reach them. Results of aquifer testing at the sites indicated a range of mean seepage velocities from 25 to 84 ft/year, which confirms this premise.

Another potential reason for the change from Phase I to Phase II is the water table fluctuations at the sites. Both the Groseclose and Jones site water table monitoring yielded the highest water table elevations in August and September 1991 during Phase II monitoring. This was also the time when concentrations of nitrogen species and total phosphorus began to increase in wells near the drainfields. This may explain the increase in TP and TKN, but not nitrate-nitrogen. Nitrate is a conservative parameter which generally moves freely with groundwater, so one would have expected to see it continuously with time. One plausible explanation however, may be that the increased water table elevations resulted in a shift in groundwater flow direction. This shift could have been vertical or horizontal, but may have caused the plume to be centered more on the monitoring wells.

An increase in fecal coliforms was also noted in monitoring wells located at the Jones and Groseclose sites during August and September of 1991. The increase, however, can not definitely be attributed to septic tank leachate as background wells and wells located greater distances from the drainfield showed approximately the same increase.

No difference in canal water quality was noted between Phase I and Phase II.

Effects of Rainfall Events: Rainfall totals for periods one day and seven days prior to sampling are shown in Table 16. On April 4, June 27, and July 23, 1990, during Phase I, the monitoring wells and canal surface water sampling points were sampled after periods of relatively heavy rainfall. The estimated rainfall for the seven days prior to these three sampling events were 1.68, 2.90, and 2.88 inches, respectively. The estimated rainfall for one day prior to each rainfall sampling event was 0.60 inches for the April 4 event, 0.50 inches for the June 27 event and 2.05 inches for the July 23 sampling event. The water quality of the monitoring wells did not change significantly ($p \leq 0.50$) during these sampling events compared to previous levels recorded. Many of the wells had lower concentrations during these two sampling events (Appendix D). This was most likely due to dilution. The canal samples also had lower concentrations during these sampling events. The three rain events raised the water table somewhat, but due to the dry conditions the previous months, the water table was still within 2 to 3 feet below the bottom of the drainfields.

During Phase II, two additional rainfall events were monitored. These occurred on August 20, 1991, and March 26, 1992, and represented one day rainfall totals of 0.10 and 1.20 inches, respectively. The water quality of the wells and canal stations after the two rainfall events in Phase II also did not change significantly ($p \leq 0.05$) from the averages without any rainfall influence, in nearly every case. In addition, regression analyses with mean canal and station C3 water quality data indicated no significant ($p \leq 0.05$) linear relationship of canal water quality with rain totals 1 day and 7 days prior to sampling. Parameters tested were nitrate, TKN, TP, conductivity, D.O., FC and FS. The August 20, 1991, sampling represented a unique opportunity, as sampling at many of the Groseclose residence wells occurred on August 19th, prior to the rain, for comparison. Data are presented in Table 17. These data are of value because all other environmental variables are similar and, therefore, the comparison is particularly valid. The 0.10 inch rain event represented an average water table increase in G-10 through G-15 of 0.04 ft. The data indicated conflicting results, however. Generally, COD, conductivity, temperature and turbidity decreased and D.O. and nitrate-nitrite-nitrogen increased (particularly in G-10 and G-13). TKN, TP, chloride, color and TDS data were variable. For example, TKN increased significantly in G-10 and G-12, but decreased in G-11 and G-13. TP increased in G-10, decreased in G-11, but concentrations were similar in other wells (Table 17).

Ideally, sampling should have been conducted on a daily basis after the rainfall events. Bromide tracer testing beneath the drainfield indicated an approximately 5 day travel time through the unsaturated zone and then a movement downgradient in groundwater at approximately 0.24 feet per day. Although heavy rainfall may change the travel times somewhat, it nevertheless appears that sampling several days after a rainfall would yield a better idea of the effects. Also, since the true travel time after a rainfall would be unknown, sampling on a daily basis for a period of time equal to the estimated travel time would be recommended.

Table 17. **Water Quality Data Collected at Selected Groseclose Residence Stations on August 19, 1991 and August 20, 1991, Prior to and After 0.10 Inch Rainfall.**

Station:	G10		G11		G12	
Parameter Units	19 Aug	20 Aug	19 Aug	20 Aug	19 Aug	20 Aug
Water Table (NGVD)	22.56	22.58	22.77	22.81	23.28	23.27
FC (col./100 ml)	10.00	890.00	10.00	20.00	<10.00	140.00
FS (col./100 ml)	6500.00	2700.00	2750.00	3350.00	70.00	1050.00
C.O.D (mg/l)	61.00	53.00	50.00	43.00	81.00	49.00
Color (CPU)	344.00	330.00	340.00	330.00	184.00	186.00
Conductivity (μmhos/cm)	650.00	610.00	850.00	890.00	1000.00	940.00
D.O. (mg/l)	3.60	4.30	3.50	4.40	3.30	4.60
NO_3,NO_2-N (mg/l)	0.00	0.33	0.04	0.05	0.00	0.04
pH	6.90	6.48	6.54	6.38	6.22	6.38
Temperature (°C)	29.10	27.20	30.60	27.20	29.70	28.00
TKN (mg/l)	0.01	0.62	1.07	0.74	0.49	1.48
TP (mg/l)	0.07	0.24	0.73	0.59	0.01	0.01
Turbidity (mg/l)	20.00	17.00	26.00	15.00	11.00	15.00
Chloride (mg/l)	117.00	114.00	159.00	158.00	145.00	143.00
TDS (mg/l)	509.00	502.00	562.00	523.00	567.00	579.00

Table 17. (Continued) Water Quality Data Collected at Selected Groseclose Residence Stations on August 19, 1991 and August 20, 1991, Prior to and After 0.10 Inch Rainfall.

Station:	G13		G14		G15	
Parameter Units	19 Aug	20 Aug	19 Aug	20 Aug	19 Aug	20 Aug
Water Table (NGVD)	22.00	22.04	21.80	21.89	21.90	21.95
FC (col./100 ml)	<10.00	240.00	130.00	>200.00	17.00	80.00
FS (col./100 ml)	400.00	1150.00	2200.00	2600.00	100.00	2450.00
C.O.D (mg/l)	41.00	66.00	25.00	29.00	43.00	47.00
Color (CPU)	259.00	258.00	118.00	121.00	232.00	249.00
Conductivity (μmhos/cm)	570.00	550.00	550.00	439.00	950.00	920.00
D.O. (mg/l)	3.80	4.70	5.30	7.40	3.30	3.60
NO_3,NO_2-N (mg/l)	0.00	0.23	0.00	0.04	0.00	0.02
pH	6.70	6.70	6.80	6.91	6.67	6.38
Temperature (°C)	29.90	26.70	26.90	27.00	32.00	27.10
TKN (mg/l)	1.73	0.74	----	0.82	0.78	0.52
TP (mg/l)	0.01	0.02	0.01	0.05	0.01	0.01
Turbidity (mg/l)	11.00	16.00	9.00	8.00	6.00	6.00
Chloride (mg/l)	74.00	72.00	120.00	122.00	97.00	116.00
TDS (mg/l)	437.00	451.00	562.00	580.00	580.00	571.00

Seepage Rates and Seepage Water Quality

Seepage rates were determined on twenty-two dates from March 20, 1990, through March 26, 1992. Seepage rates varied depending on the location along the canal; however, the variation for each meter was much less than the variation between meter locations. Raw seepage meter data are presented in Appendix F; while average seepage rates are presented in Table 18.

Table 18. Average Seepage Rate (L/m²-hr) for Sites Sampled From March 20, 1990 through March 26, 1992.

Seepage Meter	Average (L/m²-hr)	Average Rate (ft/day)	Standard Deviation
S1	0.02	1.58×10^{-3}	0.02
S2	0.02	1.58×10^{-3}	0.03
S3	0.01	7.89×10^{-4}	0.01
S4	0.08	6.31×10^{-3}	0.09
S5	0.10	7.89×10^{-3}	0.05
S6	0.08	6.31×10^{-3}	0.04
S7	0.05	3.94×10^{-3}	0.04
S8	0.71	5.6×10^{-2}	0.18
S9	1.06	8.36×10^{-2}	0.18
S10	0.45	3.5×10^{-2}	0.45
S11	0.91	7.18×10^{-2}	0.82
S12	0.40	3.15×10^{-2}	0.30
S13	0.69	5.4×10^{-2}	0.21
S14	2.17	0.171	0.53
S15	1.83	0.144	0.47
S16	0.77	6.1×10^{-2}	0.35
S17	0.04	3.15×10^{-3}	0.06
S18	0.13	1.0×10^{-2}	0.06
S19	0.10	7.89×10^{-3}	0.15
S20	0.23	1.8×10^{-2}	0.15
S21	0.06	4.7×10^{-3}	0.10

The seepage rates at the Groseclose site were consistent, but very low (0.02 - 0.16 L/m²-hr). This appears to be a direct effect of the lower hydraulic conductivity of the soil at this site. The majority of the remaining seepage meters were highly variable with location but have average seepage rates less than 1.00 L/m²-hr. The only seepage meters which were significantly higher than this were S14 and S15, with average seepage rates of 2.17 and 1.78 L/m^2-hr, respectively. The seepage rate of each individual seepage meter varied little during the entire study. There was no relationship between cumulative seven day rainfall, head difference between canal and nearest well, or distance from shore, and seepage rate. The variables which seem to contribute the most to the seepage rate were conditions which were specific to each location such as: the hydraulic conductivity of the surrounding soil, the depth to a confining layer, the type of soil in the immediate area, or the water table configuration. In order to be able to estimate a seepage rate for a specific area all these variables must be considered. Higher seepage rates west of the Groseclose site are probably related to higher hydraulic conductivity (k) and hydraulic gradients. As discussed previously, the average Jones site k value (2.46 ft/day) was nearly twice that of the mean Groseclose site k (1.40 ft/day).

On June 12, 1990, August 2, 1990, January 8, 1992, March 3, 1992, and March 26, 1992, water quality samples were collected from various locations in and near the canal for possible comparison with seepage rate data for loading calculations. The samples were collected from seepage meter bags, adjacent wells, nearest piezometers, canal piezometers, littoral interstitial pore water samplers, and the canal surface water. The raw data are presented in Appendix G. Bacterial counts from canal seepage analysis are also presented in Appendix G. As the data show, concentrations for the various samples do not always agree. For example, total phosphorous levels were elevated in the seepage meters when compared to adjacent monitoring well data. These elevated levels can probably be attributed to anaerobic activity within the seepage meters and subsequent release of P from the sediments and is not indicative of nutrient contributions from groundwater. The highly variable seepage rates and water quality data make comparisons with the monitoring well data difficult and, therefore, calculations of nutrient loading are difficult. It was felt that the monitoring well and tracer test data was most valid for the sites studied, and therefore those results were used for further analysis of the data.

Estimates of Nutrient Loading

Canal sampling points and adjacent monitoring wells were sampled together on several occasions at the Groseclose site to obtain additional data for estimating nutrient loading from the OSDS to canal 66. On June 2, 1990; August 2, 1990; January 8, 1992; March 3, 1992; and March 26, 1992 water samples were collected from seepage meter bags, adjacent monitoring wells, littoral interstitial pore water samplers (LIP), canal piezometers, and the canal surface water itself. The data from these sampling events are included in Appendix G. Concentrations of nitrate-nitrite-N, total phosphorus, TKN, and COD in the canal piezometers were low and not significantly ($p < 0.05$) different from the surface water samples. These low concentrations, coupled with low seepage rates at the various canal sites (particularly the Groseclose site), indicate that nutrient loading into the canal via groundwater seepage may be low. Data from the monitoring wells also indicates that levels of contaminants from septic system leachate are near background levels close to the canal.

These results at first appear contradictory to the bromide tracer results. All monitoring data near the canal indicate that levels of nutrients, bacteria and other parameters are near background levels, suggesting that a plume of septic system contaminants has not reached the canal. The bromide tracer results, however, suggest that a travel time of only slightly greater that 270 days would be required for mobile contaminants such as nitrate to reach the canal from the drainfield. Since the homes studied were at least 8 years of age, a nitrate plume should have reached the canal, especially at the Groseclose site, if nitrate is truly a mobile contaminant. One plausible explanation for these results is that the mixing of renovated septic tank effluent downgradient of the drainfield results in sufficient dilution to result in concentrations of mobile contaminants equal to background levels. To check this hypothesis, sample calculations were made for dilution of mobile contaminants (i.e. those that move freely with groundwater) downgradient of the drainfields at the Groseclose site.

Figure 38 shows a schematic of the assumed groundwater flow system for these examples. The groundwater system consists of the background groundwater moving under the drainfield from upgradient and the renovated septic tank effluent entering this groundwater zone from the unsaturated soil below the drainfield. The two flows mix over some distance downgradient resulting in a flow of "impacted" groundwater. The mass of a given mobile pollutant in any flowstream can be calculated as concentration (C) x flow (Q). This results in the following mass balance for the system in Figure 38:

$$C_0Q_0 + C_eQ_e = C_i\,(Q_e + Q_0)$$

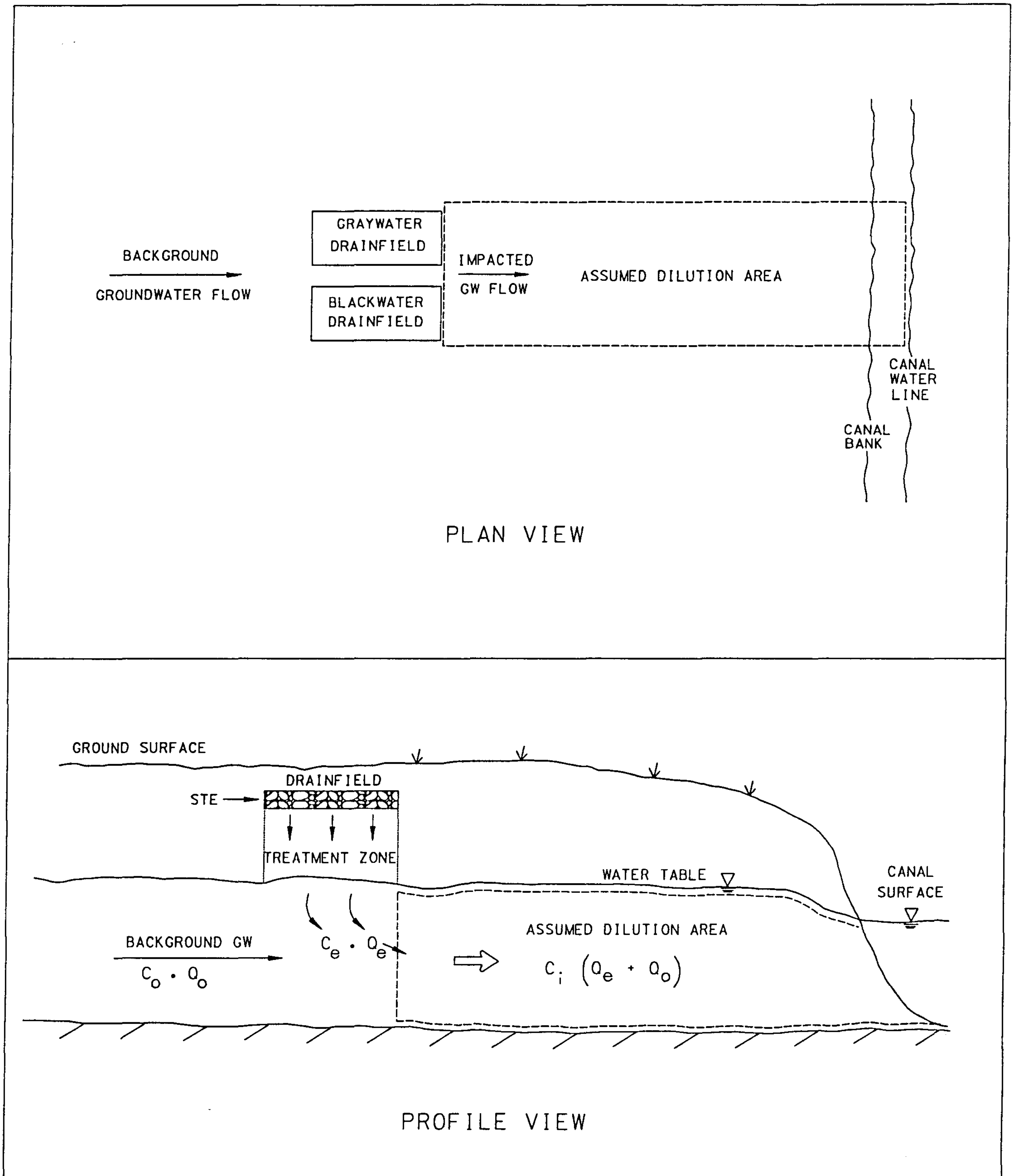
where:

C_0 = concentration of pollutant in background groundwater, mg/L
Q_0 = flow of groundwater from upgradient, L/day
C_e = concentration of pollutant in renovated effluent, mg/L
Q_e = flow of septic tank effluent to drainfield, L/day
C_i = concentration of pollutant in impacted groundwater, mg/L

Rearranging and solving for Q_0/Q_e (the "dilution factor") yields the following:

$$Q_0 / Q_e = C_e - C_i / C_i - C_0$$

Data from tracer test #2 indicated a bromide concentration of approximately 6200 mg/L entering the groundwater below the drainfield and a concentration of approximately 2600 mg/L five feet further downgradient. Substituting these values into the above yields a "dilution factor" of approximately 1.4 assuming zero bromide in the background groundwater. This would indicate that the background groundwater flow was approximately 1.4 times the flow of wastewater applied to the drainfield. Applying this dilution factor to an STE flow of 400 gallons/day, assuming an average total nitrogen concentration of 50 mg/L, and assuming complete mixing of the two flows down gradient would result in a calculated total nitrogen concentration of C_i = 21 mg/L between the drainfield and canal. Actual total nitrogen concentrations at monitoring well G-6 at the Groseclose site, in the area of highest nitrogen concentrations downgradient, were only about 10 mg/L. Forty feet further downgradient total nitrogen levels were at or below background levels. This contradiction would indicate that either the dilution factor was higher than estimated or that some removal of nitrogen occurred through denitrification.



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SYSTEM SCHEMATIC FOR DILUTION CALCULATIONS

FIGURE: 38

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Since the above dilution factor calculation was based on bromide tracer test data for a distance of only five feet downgradient of the drainfield, additional dilution would probably occur as groundwater moved toward the canal. Additional monitoring of the tracer plume would have given more accurate estimates of the dilution factor, and further work in this area is recommended. However, based on the results of other parameters it does not appear that much additional dilution occurs. Based on chloride and total dissolved solids concentrations at well G-5 compared to concentrations at the canal wells (G-13, G-14, G-15), dilution factors of 2.5 and 2.9 were calculated for chloride and total dissolved solids (TDS), respectively. Using a dilution factor of 2.5 in the example above for total nitrogen would result in a downgradient nitrogen concentration of 14.3 mg/L, still much higher than actually measured at the canal wells. It therefore appears that substantial denitrification may be occurring below and downgradient of the OSDS drainfield at the Groseclose site. The organic-coated sand layer described in the soils description may be contributing to this phenomena.

A similar result was obtained at the Jones site except nitrogen levels, especially nitrate, were significantly higher immediately downgradient of the blackwater drainfield in the backyard. Nevertheless, total nitrogen levels still appeared to be at or below background levels within 40 feet of the drainfield. Monitoring well density was not nearly as great at the Jones site however and it is felt that the contaminant plume may have escaped detection past well J-8. No tracer testing was conducted at the Jones site which limited further analyses of contaminant transport.

Estimates of nutrient loading to the canal system from OSDS in the Port Malabar area are difficult to make in light of the results obtained from this study. Neither nitrogen or phosphorus levels above background concentrations were documented further than 30 feet downgradient of the drainfields studied despite the fact that both homes were over 8 years of age. Although some dilution is responsible for the reduction in groundwater concentrations, it appears that phosphorus attenuation by the onsite soils is significant and that denitrification is contributing to significant nitrogen removal as well. While the results of this study indicate that OSDS are not significantly impacting the canals at present, they do not indicate that this will continue indefinitely. Phosphorus concentrations should gradually increase downgradient of the OSDS after sediments at the site have reached their absorption capacity and eventually may enter the canal. Assuming it took 10 years at the Groseclose residence to measure increased levels of TP 20 feet downgradient of the blackwater drainfield, it would take more than 20 additional years to reach the canal at the same rate of movement. This is only a crude estimate and further tracer experiments and soils analyses would be required to make accurate estimates of phosphorus and nitrogen transport.

Table 19 has been prepared to estimate nutrient mass loading to the canals, by homes bordering the canal, if and when nitrogen and phosphorus reach the canals. These estimates assume various levels of nutrient attenuation or removal between the OSDS and the canal. The purpose of this analysis is not to present accurate estimates of nutrient loading to the canals, but rather to be used as a tool to determine if nutrient loading from OSDS is a significant part of overall nutrient budget to the drainage system which leads to the Indian River Lagoon. A previous investigation of nutrient loading to the Crystal River/Kings Bay system from OSDS indicated that nutrient loading from OSDS under worst case conditions was a very small part of the total nutrient budget compared to wastewater treatment facilities, stormwater, and other inputs (Ayres Associates, 1991). A complete preliminary nutrient balance should be determined for the Indian River Lagoon to determine the significance of the estimates presented in this report.

Table 19 indicates a relatively wide range of nutrient loading depending on attenuation level between the OSDS and the canal. Assuming that very little regeneration of phosphorus adsorption sites occurs with time, total phosphorus loading may eventually reach 2.0 kg/home/year for homes bordering the canals. While one would normally assume little attenuation of nitrogen, results of this investigation would indicate that substantial nitrogen removal is occurring. Therefore, a nitrogen loading of approximately

Table 19. **Estimated Ultimate Nutrient Loading to Canals from OSDS Bordering Canals, Assuming Various Levels of Nutrient Reduction***

Attenuation or Removal (%)	Mass Loading to Canal (kg/canal home/year)	
	Total Phosphorus	Total Nitrogen
10	2.0	7.5
25	1.7	7.2
50	1.1	4.1
75	0.6	2.1
90	0.2	0.8

* Assumes average STE total nitrogen concentration of 40 mg/L, phosphorus concentration of 11 mg/L and a per capita STE flow of 50 gal/pers/day with an average household population of 3 (based on Ayres Associates, 1989).

4.0 kg/home/year may be a reasonable estimate for homes along the canals at some point in the future. Further investigation should be conducted at additional sites such as the ones studied herein to more accurately define the transport of nutrients and to better estimate nutrient loading to the drainage system and eventually the Indian River Lagoon system.

V. CONCLUSIONS AND RECOMMENDATIONS

This report presents the methods and results of a study conducted to determine the potential impact from OSDS on water quality in the Indian River Lagoon Basin. Two residential OSDS and an undeveloped control area of the Port Malabar subdivision in Palm Bay, Florida were studied over a two year study period. The impact of the OSDS on a nearby drainage canal which subsequently emptied to the Indian River Lagoon via the Turkey Creek drainage system was investigated. Monitoring wells were installed upgradient and downgradient of the OSDS drainfields. Seepage meters and canal piezometers were installed in the canal bottom. Septic tank effluent, groundwater, and canal surface water and seepage water samples were collected and analyzed for key water quality parameters indicative of OSDS impact. Aquifer testing and a bromide tracer test were conducted to determine key groundwater characteristics and to assist in analysis of the data.

Based on the data collected in this study, the following conclusions can be drawn:

Site Characteristics

1. The residences studied were typical of those in the Port Malabar subdivision and utilized separate blackwater and graywater septic tanks and drainfields. Water use monitoring indicated wastewater loading rates to the drainfields were below design loading rates per Chapter 10D-6, F.A.C.

2. The soils of the study area were typical of the South Florida Flatwoods land resource area and consisted of Myakka sand at the Jones site, Oldsmar sand at the Grosclose site, and EauGallie sand at the control site. A sandy clay loam layer was encountered at depths of five to seven feet at the Groseclose site.

3. Groundwater flow direction at both residences and the control site was in the general direction of canal 66, to the north. Groundwater elevation monitoring indicated an unsaturated soil thickness below the drainfields which varied with

season from 1.2 to 2.9 ft. at the Groseclose site and from 3.3 to 5.2 ft. at the Jones site.

4. Bromide tracer testing at the Groseclose site indicated an unsaturated zone travel time of 5 days below the drainfield and an average groundwater seepage velocity of 0.24 ft./day towards the canal.

Groundwater Water Quality

1. Nutrient concentrations (NO_3,NO_2-N, TKN, TP) were generally higher in those wells in the immediate vicinity of the OSDS drainfields as compared to concentrations in wells further downgradient, and significant differences ($p \leq 0.05$) generally occurred for these parameters in well groups with distance from the drainfields. However, nutrient concentrations were at or below background levels within 20 to 40 feet of the drainfields.

2. Fecal coliform counts in the monitoring wells were generally below 10 cols/100 ml on two-thirds of the sampling dates. Fecal streptococcus levels were high in all wells, generally ranging from 100 to 2000 col./100 ml (geometric mean). Fecal streptococcus and fecal coliform bacterial data did not statistically ($p \leq 0.05$) indicate significant reductions in number with increasing distance from the drainfield. The high levels of fecal streptococcus encountered in the groundwater at the Groseclose site were thought to be attributable to the utilization of canal water for lawn irrigation at this site and the presence of ducks, geese, and chickens nearby.

3. Comparison of fecal coliform bacterial data to fecal streptococcus indicated a wildlife or animal rather than human source of contamination. The FC/FS ratios in monitoring wells and canals were very low. The average monitoring well FC/FS ratio was 0.04 which is indicative of a non-human source of fecal contamination.

4. Based on multiple regression analysis, there were no significant ($p \leq 0.05$) correlations between chemical parameters and bacterial levels in the wells and environmental parameters such as water table height, temperature and rainfall levels.

5. Phase II results were obtained one year after the end of the Phase I monitoring. Considerable differences in water quality were measured between Phase I and Phase II sampling periods. Nitrate-nitrogen concentrations increased in groundwater obtained from monitoring wells located within twenty feet of the blackwater drainfields. Phosphorous and TKN concentrations also increased in

some of the wells. At the Jones site, peak nitrate-nitrogen concentrations exceeded 50 mg/L at several wells located within 20 feet of the blackwater drainfield. Total phosphorous and TKN concentrations were also elevated. It was speculated that these increases were due to higher water table elevations or a shift in groundwater flow.

Surface Water Quality

1. Bacterial counts were high and variable in the surface water obtained from canal 66. Fecal coliform and fecal streptococcus levels peaked at canal station C3 which is located near the Groseclose site. The peak levels of bacteria appear to be related to the presence of numerous ducks, geese, and chickens located near this sampling station. This was supported by FC/FS ratios at the receiving canal stations (C-0 through C-5) which averaged 0.17:1, indicating the likelihood of non-human sources of pollution. The FC/FS ratio also suggested that stormwater runoff may be a source of bacteria.

2. A significant ($p \leq 0.05$) inverse linear relationship existed between mean canal fecal streptococcus levels (log) and stage height, and a significant positive relationship ($p \leq 0.05$) was found between mean canal fecal streptococcus and temperature. At station C3 additional relationships were found between fecal coliform levels (inverse) and stage height ($p \leq 0.06$) and temperature ($p \leq 0.05$). Also, nitrate was inversely related to stage height ($p \leq 0.03$) and conductivity was positively related to temperature ($p \leq 0.03$).

3. Based on canal water sampling, OSDS impacts on the receiving canal water quality were not evident. There were no significant ($p \leq 0.05$) relationships between nutrient concentrations in the canal surface water and sampling locations in the canal.

4. Based on the data collected after rainfall events (five {5} occasions), no conclusive cause-effect relationships on either groundwater or surface water quality were found.

5. Seepage rates of groundwater into the canal were low and very site specific. No correlation ($p \leq 0.05$) was observed between head difference, rainfall, distance from shore, and seepage rate. Seepage meter rate data were very low compared to groundwater fluxes to the canal predicted from monitoring well data.

6. Seepage fluid water quality data generally indicated that seepage meter water quality may not be directly comparable to monitoring well or even canal piezometer data and, in turn, may not be useful for the determination of nutrient loading to the

canal. Based on parameter concentrations encountered in the seepage water, seepage meter water quality is probably effected by conditions within the seepage meter itself.

Summary

1. Estimates of nutrient loading to the canal system at Port Malabar were not definitive. Tracer test data collected during site characterization indicated that conservative parameters such as nitrate and chloride should reach the canal in approximately 270 days, yet concentrations of these compounds were measured at or below background levels within 20 to 40 feet of the drainfields. Calculations of "dilution factors" indicated that, although some dilution may be responsible for these results, it also appeared that phosphorous was significantly attenuated by onsite soils and that denitrification was contributing to substantial nitrogen removal.

2. The results of the study indicated that while OSDS were impacting groundwater in their immediate vicinity, they were not impacting canal water quality significantly at the time of this study. This may not continue indefinitely however, and it was estimated that total phosphorous loading to the canal may eventually reach a maximum of 1 to 2 kg/home/year for homes bordering the canal. Although nitrogen was significantly reduced at the study sites (especially Groseclose), it was estimated that under unfavorable conditions, total nitrogen loading from homes bordering the canal could be as high as 4 to 7 kg/home/year.

3. Fecal bacterial impacts to the canal could not be assessed due to the variability of the data collected and the impacts from wildlife. A better indicator of fecal bacterial impacts than fecal coliform is needed.

Based on the results of this study and the conclusions listed above, Ayres Associates recommends that the Water Management District complete a preliminary nutrient budget for the Indian River Lagoon from all sources, utilizing the estimated loadings above for OSDS inputs. If the OSDS nutrient loading is a significant part of the overall nutrient budget to the lagoon, additional study to refine the nutrient loading estimates above would be recommended. If these estimates proved accurate, an investigation of nutrient reduction techniques for OSDS should be initiated.

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