# A COMPARATIVE STUDY OF A CONVENTIONAL AND A MOUND ON-SITE WASTEWATER DISPOSAL SYSTEM IN A COASTAL ENVIRONMENT

BY

EDUARDO J. MILES TOUYA

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Approved:

Daniel L. Gallagher

D.L. Gallagher
Chairman

G.M. Simmons, Jr

G.M. Simmons, Jr
Alumni Distinguished Professor

C.W. Randall

H. Kator
Virginia Institute of Marine Science

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(ABSTRACT)

Wastewater effluents from two absorption systems were studied in the Assateague Island National Seashore Park, during 1993. The study focused on the wastewater effluents from a mound and a conventional on-site wastewater disposal system (OSWDS). The site was characterized by coastal beach soils. Average wastewater loadings to each drainfield were 10,800 liters per day. The density of the ground water varied due to changes in salinity and the hydraulic gradients fluctuated due to tidal effects. The OSWDS's were used between four to five months of the year, with three to four months at near full capacity.

Dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) immediately adjacent to the conventional drainfield were correlated with bath house usage. The average DIN concentration was 0.76 mg/L in January-March, before the season started, increased to 80.78 mg/L by August, and decreased to 26.41 mg/L in October-November, once the season ended. Likewise the DIP increased from 1.20 mg/L before the season started, to 7.36 mg/L during the season, and decreased to 1.86 mg/L once the season ended.

Conversely, the DIN and DIP in the ground water immediately adjacent to the mound drainfield were not well correlated with bath house usage. The DIN increased from 1.01 mg/L during off-season, to 39.95 mg/L during September and showed almost no appreciable difference after fall closure. The DIP did not show a significant difference from off-season values, with an average value of 0.29 mg/L.

Transport of the wastewater through the ground water toward the surface waters was suggested by increasing DIN concentrations with distance from both drainfields over time with increasing bath house usage.

Elevated *Escherichia coli* concentrations were measured in the ground water immediately adjacent to the conventional drainfield. Concentrations of *E. coli* ≥ 16,000 MPN/100 ml were measured up to 2 meters from this drainfield. On the other hand, the mound did not show, on average, ground water fecal contamination. There was no indication of fecal coliform contamination to the ocean waters in either site. On only three occasions were *E. coli* concentrations, with values between 30 and 130 MPN/100 ml measured at distances greater than 40 meters from the conventional drainfield. At the mound site on only one occasion was a *E. coli* concentration with a value of 23 MPN/100 ml measured at distance of 50 meters from the drainfield.

The conclusions from this study were: 1) the mound drainfield had a better performance in wastewater treatment than the conventional drainfield, 2) direct transport of *E. coli* to surface water via ground water could not be detected or measured, 3) both systems showed direct loadings of nutrients to the ground water, 4) there was some evidence of nutrient transport to surface waters, and 5) the conventional system showed high loadings of *E. coli* to ground water.

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# 1.0 INTRODUCTION AND STUDY OBJECTIVES

## 1.1 Introduction

The septic tank system (STS) has become the on-site wastewater disposal system (OSWDS) most widely used in the United States. In 1976 approximately 70 million Americans were using STS (Hershaft, 1976). The Environmental Protection Agency estimated in 1980 that approximately 25 percent of the new homes built in U.S. used STS (EPA, 1980). For the same year, the Bureau of Census (1983) estimated that 24.1% of the total homes in the U.S. were using STS. The wastewater introduced to the subsurface was estimated to be between 10 to 14 billion liters per day (Bureau of Census, 1983; OTA, 1984). A high risk of ground and surface water pollution exists due to the great volume of wastewater introduced into the upper part of the soil stratum. Several outbreaks cases of waterborne communicable diseases due to contamination of drinking supplies have been documented (McGinnis and DeWalle, 1983). In addition, a number of studies indicate the high probability for these outbreaks to occur (Yates and Yates, 1988; Gerba, 1984; Stramer, 1984; Vaughn et al., 1983).

The septic tank systems are also use in public establishments such as airports, campgrounds, motels, etc., producing an equally high or higher risk of ground and surface water pollution than the residential household STS. The Assateague Island National Seashore Park is an example of a public establishment which utilized the septic tank system for wastewater treatment. In 1992 the park service expressed concern for the possibility of ocean water contamination by wastewater effluents from one of their two bath house drainfields. Monitoring of the drainfield effluent showed that the ocean waters were being contaminated by coliform bacteria. In view of the study results, the park service decided to modify the existing conventional drainfield, and construct a mound system.

This research was conducted for the Assateague Island National Seashore Park Service during 1993 to monitor the performance of the new mound system and the remaining conventional drainfield.

## 1.2 Study Objectives

The study examined the performance of wastewater purification by a mound and a conventional on-site wastewater disposal system under nearly extreme operational conditions at the Assateague Island National Seashore Park. The hypotheses were:

1. There are no differences between the mound and the conventional type drainfield with respect to nutrient loadings to ground and coastal waters.

2. There are no differences between the mound and the conventional type drainfield with respect to fecal coliform loading to the ground and coastal waters.

## 2.0 LITERATURE REVIEW

Much work has focused on the influence of STS effluents in the environment. Research has been conducted in the laboratory (Lotse, 1976; Sawhney, 1977; Vilker, 1978), at individual home sites (University of Wisconsin, 1978; Jones and Lee, 1979; Reneau, 1977, 1978, 1979; Viraraghavan, 1978), in communities (Pitt, 1974; Shoemaker and Porter, 1978), in lakeshore areas (Hayes et al., 1990; Chen, 1988; Hendry and Toth, 1982; Childs et al., 1974), and in coastal areas (Cogger et al., 1988; Stewart and Reneau, 1988; Carlile et al., 1977). The review of literature indicates that the STS is an adequate OSWDS if adequately built and maintained, but has the potential to become a pollution source otherwise.

The literature review chapter is subdivided into four sections describing in 1) the septic tank system; 2) the ground water hydrology in coastal areas; 3) the transport and fate of fecal-coliform bacteria; and 4) nutrient loading beneath septic tank fields.

## 2.1 THE SEPTIC TANK SYSTEM

The basic septic tank system is composed of a septic tank and a subsurface soil absorption system known as drainfield or leachfield (Figure 1). The septic tank removes the settleable and floatable solids from the wastewater. The drainfield functions as a filter and purifier breaking down biodegradable components and retaining any organisms from the human gastrointestinal and urinary tracts.

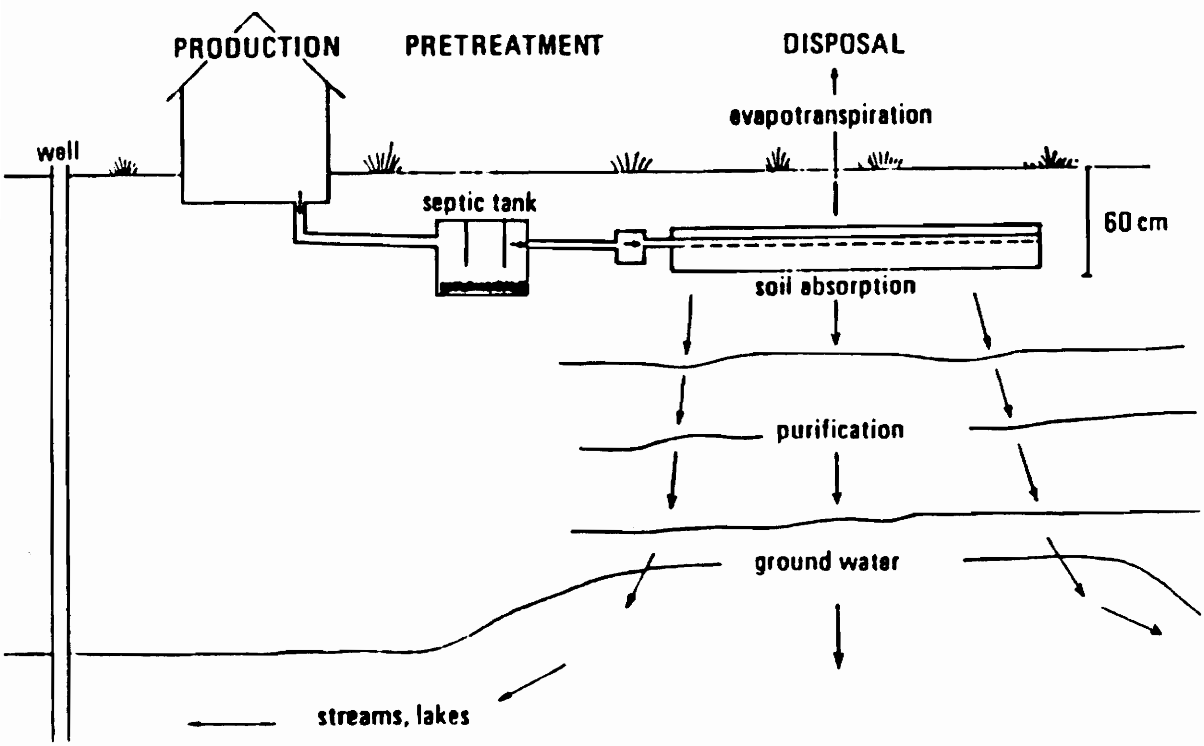



Figure 1. Typical septic tank-soil absorption system (Canter and Knox, 1985).

### 2.1.1 SEPTIC TANKS

Winneberger (1984, Vol.1) described the septic tank (Figure 2) as a system to receive wastewaters, separate settleable and floatable solids, store and digest the separated solids, and transfer the treated wastewater out of the tank. Thus, the septic tank is a sedimentation basin in which physical removal and biological activity occur combine to produce a clarified effluent that is discharged to the soil absorption system.

## 2.1.1.1 Tank Design

In order to accurately design a septic tank, certain parameters must be determined, which include size, number of compartments, path of sewage within the tank, gas conduits, and venting.

Tank size is based on the amount of wastewater to be handled. The size must provide the highest reasonable removal of suspended solids to minimize the possibility of clogging the drainfield through passage of solids via the septic tank (Cotteral and Norris, 1969). Minimum size is determined from the average daily volume of wastewater to be treated with the addition of a safety factor for variations in wastewater loading (Canter and Knox, 1985). Peak wastewater loading flows must be determined for commercial, institutional, or industrial sources. The U.S. Environmental Protection Agency (1980, October) provided information regarding the requirements of minimum tank size in relation to loading volumes. For flow between 750 and 1500 gallons per day, the minimum tank capacity should be equal to 1-1½ days wastewater flow. For flows between 1500 and 15000 gallons per day, the minimum tank capacity can be calculated by the following equation:

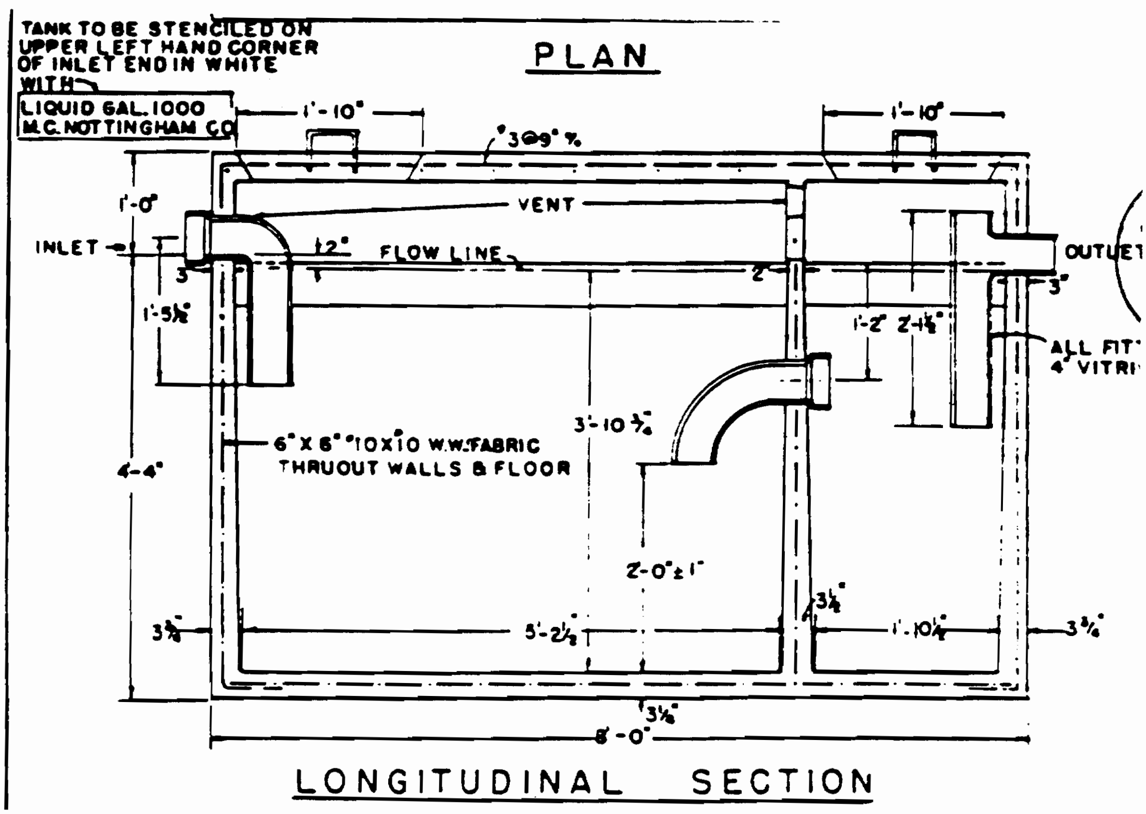



Figure 2. Typical two-compartment septic tank (Baumann and Jones, 1977)

$$V = 1125 + 0.75 \cdot Q$$

where: V = net volume of the tank (gallons)

Q = daily wastewater flow (gallons)

Given that oversize tanks would not be cost effective, the number of compartments and the path of sewage within the tank are used to increase sedimentation and the chemical and the biological processes that take place inside the septic tank. Multiple compartment septic tanks reduce mixing and turbulence induced by the entering wastewater, and provide the function of a multiple settling basin. The most common septic tank has two compartments, with the second compartment being one third to one half the size of the first one. In the first compartment, the heaviest sludge and scum are generated. The second compartment receives the clarified effluent form the first compartment with almost no entering turbulence which enhances settling of low-density solids (Canter and Knox, 1985). The path of the sewage in the septic tank influences the performance of the tank's removal efficiency. The removal efficiency is increased by the lengthening of the path (Kaplan, 1991).

The last important parameter in the septic tank design are the gas vents. The vents allow the different gases, produced during the biodegradation of the sewage material, especially methane, hydrogen sulfide and odorous gases to escape from the tank (Kaplan, 1991).

#### 2.1.1.2 Physical Processes

Gravity separates the wastewater solids within the septic tank. The lighter solids such as grease, oils and some fecal constituents float to the top. This material undergoes microbial biodegradation forming a floating layer called, "scum". The scum layer is increased by settleable solids that are carried by gas produced through biodegradation. The heavier solids, as well as some digested scum solids, settle to the bottom of the tank. These solids undergo a slow biodegradation under anaerobic conditions and produce a blanket of sludge. A partially clarified liquid appears between the scum and the sludge. This liquid flows to the following septic compartment where further settling and biodegradation processes occur (Canter and Knox, 1985; Kaplan, 1991).

On average, two thirds of the volume of the septic tank is reserved for the storage of the sludge and scum. The zone available in the tank for clarification of incoming wastewater is the partially clarified liquid (Cotteral and Norris, 1969). Therefore, regulations exist to establish the detention time in order to provide adequate treatment. The state of Virginia (1980, October) sets the liquid volume sufficient for a 24 hour retention time at maximum sludge depth and scum accumulation.

Other researches studied the composition of the sludge (Metcalf and Eddy, 1916; Kershaw, 1925; Clanton et al., 1984; U.S. E.P.A., 1980), and scum (Dunbar, 1907, Kinnicutt et al., 1910) as well as the composition of the incoming wastewater (Weibel et al., 1954; CEQ, 1974; Bauer et al., 1979) from domestic users. All authors concluded the composition of the scum, sludge, and incoming wastewater is highly dependent on the nature of the wastewater. Table 1 shows the typical composition of untreated domestic wastewater.

Table 1. Typical composition of untreated domestic wastewater. Units in milligram per liter. (Tchobanoglous and Burton, 1991)

| | Concentration | | |
|---|---|---|---|
| **Contaminants** | Weak | Medium | Strong |
| **Solids, total (TS)** | 350 | 720 | 1200 |
| **Dissolved, total (TDS)** | 250 | 500 | 850 |
| **Fixed** | 145 | 300 | 525 |
| **Volatile** | 105 | 200 | 325 |
| **Suspended solids (SS)** | 100 | 220 | 350 |
| **Fixed** | 20 | 55 | 75 |
| **Volatile** | 80 | 165 | 275 |
| **Settleable solids** | 5 | 10 | 20 |
| **Biochemical oxygen demand, mg/L: 5-day, 20°C ($BOD_5$, 20°C)** | 110 | 220 | 400 |
| **Total organic carbon (TOC)** | 80 | 160 | 290 |
| **Chemical oxygen demand (COD)** | 250 | 500 | 1000 |
| **Nitrogen (total as N)** | 20 | 40 | 85 |
| **Organic N** | 8 | 15 | 35 |
| **Free ammonia** | 12 | 25 | 50 |
| **Nitrites** | 0 | 0 | 0 |
| **Nitrates** | 0 | 0 | 0 |
| **Phosphorus (total as P)** | 4 | 8 | 15 |
| **Organic P** | 1 | 3 | 5 |
| **Inorganic P** | 3 | 5 | 10 |
| **Chlorides**[a] | 30 | 50 | 100 |
| **Sulfates**[a] | 20 | 30 | 50 |
| **Alkalinity (as $CaCO_3$)** | 50 | 100 | 200 |
| **Grease** | 50 | 100 | 150 |
| **Total coliform (no/100ml)** | $10^6$-$10^7$ | $10^7$-$10^8$ | $10^7$-$10^9$ |
| **Volatile organic compounds (VOCs) mg/L** | <100 | 100-400 | >400 |

[a] Values should be increased by amount present in domestic water supply

### 2.1.1.3 Biological Processes

Winneberger (Vol.II, 1984) stated that four different zones are generated by the wastewater in a septic tank: the sludge, the liquid partially clarified between sludge and scum, the scum layer, and the zone above the scum. Winneberger continued to say that even though, the septic tank is regarded as an anaerobic environment, in reality it represents a microcosm. The septic tank environment is not strictly anaerobic because the biochemistry of both anaerobiosis and aerobiosis occur concurrently.

Seventy one percent by mass of the organic matter inside the septic tank is converted to gas and leaves the septic tank by the gas vents (Baumann and Jones, 1977). Winneberger (Vol.II, 1984) collected samples of sludge gases as they rose through the liquid phase of seven septic tanks. The average measurements obtained as percentage of volume were: methane, 72.9; nitrogen, 13.5; carbon dioxide, 12.4; hydrogen sulfide, 0.86; oxygen, 0.65 and argon, 0.24. It is interesting to observe, for example, the high concentration of hydrogen sulfide, which varied in the experiment between 0.29 to 1.36% which is equivalent to 2900 to 13000 ppm. A 500 ppm level would kill a human in thirty minutes. Even though the removal of bacteria and viruses is not a purpose of the septic tank treatment, reduction of these organisms were observed (Laak and Crates, 1977; Pitt et al., 1975; Metcalf and Eddy, 1916). Feachem et al. (1983), based on data collected by various researchers, states that a well designed and maintained septic tank, with a retention of 3 days in warm temperatures (>25°C) will reduced fecal bacteria by 50 to 95 percent. Brandes (1978), as cited by Feachem et al. (1983), recorded in three septic tanks, with retention times of 2 to 10 days, mean fecal coliforms values (per 100 ml) of: $4 \cdot 10^5$ to $2 \cdot 10^6$ in the first compartment supernat; $1 \cdot 10^5$ to $1 \cdot 10^6$ in the second compartment supernatant; $9 \cdot 10^5$ to $8 \cdot 10^6$ in the first compartment sludge; $6 \cdot 10^4$ to $6 \cdot 10^5$ in the second compartment sludge; and $5 \cdot 10^5$ to $4 \cdot 10^6$ in the effluent. The University of Wisconsin (1978) in a two year study of seven septic tanks recorded, from septic tank effluent, fecal coliform concentrations of $2.5 \cdot 10^6$-$1.0 \cdot 10^7$ per 100 ml. Ziebell (1974) recorded total coliform mean concentration of $34 \cdot 10^5$ and fecal coliform mean concentration of $42 \cdot 10^4$ per 100 ml from five tank effluents.

#### 2.1.1.4 Septic Tank Performance

Septic tank performance is influenced by the hydraulic loading, maintenance, and temperature (assuming that septic tank design is adequate and that wastewater characteristics do not change). The hydraulic loading will influence the removal performance of suspended solids. Theoretical detention times, which is defined as the volume occupied by the partially clarified liquid, divided by the design flow rate, are set by the regulatory agencies. The required minimum detention time is 24-hr at maximum sludge depth and scum accumulation (E.P.A., 1980). Increasing the detention time translates to an increase in the removal of suspended solids, i.e. an increase in the detention time from 20 hours to 35 hours could increase by 28% the removal of suspended solids (Cotteral et al., 1969).

Septic tanks must be pumped in order to remove the sludge and scum every 3 to 5 years, or when the accumulated sludge reaches or exceeds 1/4 of the total liquid depth, to maintain adequate detention times and prevent clogging of the drainfield pipes (Cotteral et al. 1969; Kaplan, 1991; Canter and Knox, 1985). The efficiency of the septic tank can be determined by measuring wastewater characteristics in the inflow and outflow. Tables 2 and 3 show the quality of the wastewater effluent from septic tanks.

### 2.1.2 SOIL ABSORPTION SYSTEM

The primary functions of a soil absorption system are: 1) to discharge the clarified wastewater from the septic tank into the soil; 2) to purify the wastewater by filtering the microorganisms, and by the aerobic micro-biodegradation of the wastewater components (Kaplan, 1991; Parker et al., 1977). The absorption system design depends on the site, effluent, and volume characteristics. The soil and site characteristics control the purification potential of the system which is mainly governed by the existence of an unsaturated zone between the discharge zone and the groundwater table. In order to provide an adequate soil absorption system, the drainfield performance parameters, types of absorption systems, and soil clogging must be understood.

#### 2.1.2.1 Drainfield Performance Parameters

Cotteral et al. (1961) stated that the most important factors influencing the soil absorption systems are percolative capacity, infiltrative capacity, soil particle size, and drainfield loading rates. They defined these parameters as (1) percolative capacity equals the rate at which effluent can be transmitted through the pores or interstices of the soil; (2) infiltrative capacity equals the measure of rate at which effluent can enter the soil through the surface on which it is applied; (3) soil particle size is a soil characteristic which influence both infiltrative capacity and percolative

Table 2. Summary of Effluent Quality from Seven Septic Tanks.
(University of Wisconsin, 1978)

| Parameter and Statistics | Results (1) | Parameter and Statistics | Results (1) |
|---|---|---|---|
| **Suspended Solids (mg/l)** | | **Total Phosphorus (mg-P/l)** | |
| Mean (# samples) | 49(148) | Mean (# samples) | 13(99) |
| Coef.of Variation | 0.16 | Coef.of Variation | 0.34 |
| 95% Conf. Int. | 44-54 | 95% Conf. Int. | 12-14 |
| Range | 10-695 | Range | 0.7-99 |
| **Volatile Suspended (mg/l)** | | **Orthophosphorous (mg-P/l)** | |
| Mean (# samples) | 35(148) | Mean (# samples) | 11(89) |
| Coef.of Variation | 0.18 | Coef.of Variation | 0.36 |
| 95% Conf. Int. | 32-39 | 95% Conf. Int. | 10-12 |
| Range | 5-320 | Range | 3-20 |
| **$BOD_5$ (Unfiltered) (mg/l)** | | **Total Nitrogen (mg-N/l)** | |
| Mean (# samples) | 138(150) | Mean (# samples) | 45(99) |
| Coef.of Variation | 0.42 | Coef.of Variation | 0.40 |
| 95% Conf. Int. | 129-147 | 95% Conf. Int. | 41-49 |
| Range | 7-480 | Range | 9-125 |
| **$BOD_5$ (Filtered) (mg/l)** | | **Ammonia Nitrogen (mg-N/l)** | |
| Mean (# samples) | 190(130) | Mean (# samples) | 31(108) |
| Coef.of Variation | 0.47 | Coef.of Variation | 0.46 |
| 95% Conf. Int. | 100-118 | 95% Conf. Int. | 28-34 |
| Range | 7-330 | Range | 0.1-91 |
| **COD (mg/l)** | | **Nitrate Nitrogen (mg-N/l)** | |
| Mean (# samples) | 327(152) | Mean (# samples) | 0.4(114) |
| Coef.of Variation | 0.33 | Coef.of Variation | 6.7 |
| 95% Conf. Int. | 310-344 | 95% Conf. Int. | <0.1-0.9 |
| Range | 25-780 | Range | <0.1-74 |

(1) Data from seven sites and collected over time period May 1972-December 1976.

Table 3. Mean and Range of Constituents in Septic Tank Effluents (Otis et al., 1975)

| | **Mean** | **Range** |
|---|---|---|
| **BOD5 (mg/l)** | 158 | 142-174 |
| **TSS (mg/l)** | 54 | 47-62 |
| **Fecal Coliforms (No./ml)** | 4210 | 2879-6158 |
| **Fecal Streptococci (No./ml)** | 38.2 | 20.1-72.4 |
| **Total Nitrogen (mg-N/l)** | 55.3 | 48.9-61.6 |
| **Ammonium-N (mg-N/l)** | 38.7 | 34.3-43.0 |
| **Nitrate-N, Nitrite-N (mg-N/l)** | 0.56 | 0.39-0.82 |
| **Total Phosphorus (mg-P/l)** | 14.6 | 11.4-17.7 |
| **Orthophosphate (mg-P/l)** | 11.5 | 10.2-12.8 |

capacity; and (4) loading rate is the rate of application of effluent to the drainfield infiltrative surface.

The perlocative capacity of the soil is measured by the percolation test, even though the test does not measure percolation or infiltration soil rates accurately (Kaplan, 1991; Winneberger, Vol.I 1984, Anderson et al., 1977; Bouma et al., 1972). The percolation test measures how fast water is absorbed by the soil in a test hole. The problem arises in that measuring how fast the water disappears in a test hole is not indicative of how fast the water moves through a soil. Additionally, the percolation test does not incorporate an adequate requirement to measure infiltration. In spite of the test drawbacks, it has been the most common test for determining the amount of soil interface needed for a soil absorption system. Percolation test procedures can be found in Kaplan (1991); Winneberger (Vol.I 1984); USPHS (1967).

A soil absorption system must have the capability to absorb and adequately treat the effluent wastewater from the septic tank. It is the infiltrative capacity and not the percolative capacity of the liquid-soil interface which determines the life of the absorption system, even though the soil has the adequate percolative capacity to transport the effluent away from the system liquid-soil interface (Cotteral et.al., 1969). The soil infiltrative capacity decreases as clogging appears. The clogging effects are described further in this section. Finally, the distribution system, type of dosing, and volume to surface ratio will impact the absorption system performance (Cotteral et al., 1969).

#### 2.1.2.2 Types of Adsorption systems

In general terms, an adsorption system consists of a hole in the ground with a backfill of gravel and a system of pipes to distribute the wastewater throughout the hole. The main types of soil absorption fields are: the disposal trench or leachline, the disposal or seepage bed, and the disposal or seepage pit (Winneberger, Vol.I 1984; Kaplan, 1991; Canter and Knox, 1985; Cotteral et al., 1969). The trench system consists of shallow trenches (approx. 0.3 to 1.5 m deep) of narrow width (0.3 to 0.9 m) relative to length (15 to 30 m). At the bottom of the trench, a layer (approx. 15 cm) of gravel is overlaid by a single line pipe (i.e., PVC). The infiltrative surfaces of a trench are considered to be the bottom and the sidewalls.

A disposal bed occurs when the width of the trench is greater than 0.9 meters. The disposal bed contains more than one distribution pipe and the infiltration surface is principally the bottom of the bed. The disposal pit is a vertical hole approximately 1.2 to 1.6 meters in diameter and 3 to 12 meters in depth. The vertical cylinder is generally made of concrete blocks with gravel between the soil and the blocks, and at the bottom of the hole. The walls and the bottom of the pit are considered the infiltrative surface.

Effluents from the septic tank go through a distribution box before being discharged to the adsorption system. The distribution box is a small box, and in theory, its function is to distribute the wastewater equally between the drainage pipes. The distribution can be parallel or serial. In parallel distribution, all the drainage pipes are supposed to received equal amounts of wastewater. In serial distribution, all the effluent is given to the first element of the disposal field. Once this element overflows, the effluent continues to the next element, and so forth (Winneberger, Vol.I 1984).

The mound system was developed as an alternative adsorption system to overcome soil and site conditions (high water tables, slowly permeable soils, and steep slopes) which limit the use of the conventional subsurface drainfields (Converse et al., 1975 a,b,c; Wooding, 1975; Machmeier, 1977). The mound system consists of a septic tank, a pumping or siphon chamber, and the mound (Figure 3) (Converse et al., 1977). The basic feature of the mound system is that it increases the wastewater travel time in the aerobic soil environment, thus enhancing purification.

The design of a mound system includes five components (Converse et al. 1977): 1) absorption area within the mound, 2) dimensions of the mound, 3) basal area, 4) distribution network, and 5) pumping system. The absorption area can be a bed or trenches. The mound dimensions are determined by its depth, length, width, and basal area. The depth is determined by the purification potential. To provide adequate purification, an unsaturated flow depth of 0.9 to 1.2 meters of sand and sandy loam is required (McCoy and Ziebell, 1975; Tyler et al., 1978).

The basal area is the interface area between the mound and the natural soil. The basal area size is determined by the infiltrative capacity of the natural soil. The distribution of the effluent must be uniform. Nonuniformity will translate in higher loading per surface area, thus decreasing purification potentials in some areas. Additionally, if the mound system is over low permeable soils, and the effluent is not distributed uniformly, then the effluent would move rapidly through the fill. Once the effluent reaches the interface surface with a slowly permeable soil, it would move laterally to produce surface seepage.

Finally, the pumping system consists of a pumping chamber and a pump, plus control equipment. In the pumping chamber, effluent from the

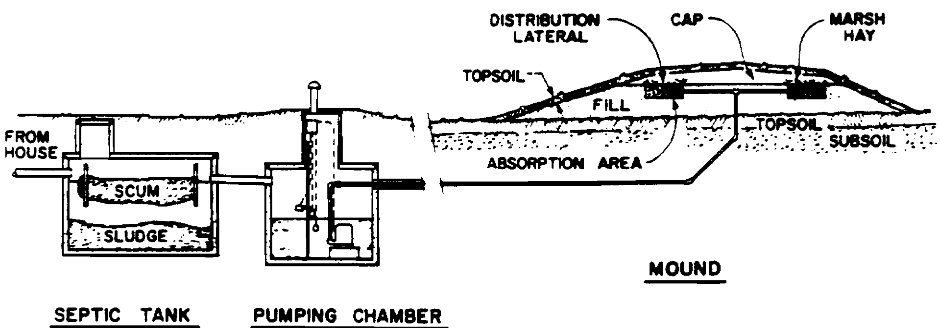



Figure 3. Cross-sectional of a septic tank mound system for on-site disposal ( Converse et al., 1978).

septic tank is collected to assure that dosing will be as uniform as possible. Though the pumping frequency is important, it is even more important to obtain the proper quantity per dose which is assured by the control equipment (Converse et al., 1977). Dosing frequency will depend mainly on the fill material. Bouma (1975) suggested dosing frequencies of four times daily for sand fills and once daily for sandy loam fills.

### 2.1.2.3 Soil Clogging

As previously discussed, the soil's hydraulic conductivity plays a crucial role in the performance of the absorption system. Certain factors that affect hydraulic conductivity in an absorption system are gases produced from biological activity, air trapped below the wetting front, soil swelling, biological breakdown of structural soil units, and accumulation of biological cell mass and cell by-products (Otis, 1984).

Biological, physical and chemical processes affect the loss of a soil's infiltrative capacity and produce soil clogging. Clogging has been described to occur in three or four phases where no appreciable construction damage had occurred (Winterer, 1922; Allison, 1947; Butler and Orlob, 1955; Jones and Taylor, 1965; Thomas et al., 1966; Okubo and Matsumoto, 1979). A typical time rate infiltration curve is given in Figure 4. This curve has three distinct phases. During Phase 1, breakdown of both the cohesive forces which hold the soil together, and the slaking of the soil caused by the affinity of the internal soil surface, produce an initial loss of the infiltration rate. Phase 2 shows an increase in the infiltration rate due to dissolution of entrapped air. After a maximum infiltration rate is achieved, there is a steady decline (Phase 3) primarily due to biological factors (Cotteral et.al., 1969).

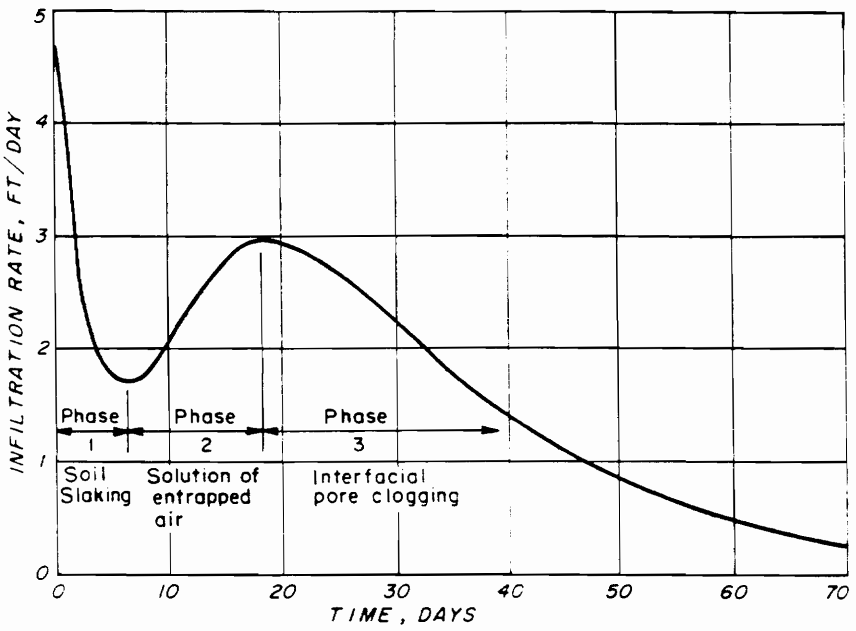



Figure 4. Typical time-rate infiltration curve (Cotteral and Norris, 1969).

This induced biological reduction of the infiltrative capacity is due to the formation of an organic mat, or biomat, at the liquid-soil interface. The thickness of this biomat is approximately 5 mm, and can extend on average 2 cm into the soil. It is within the biomat where the major biological activity occurs: deposition of suspended materials, bacterial build-up, biodegradation of organic material, and filtration of pathogenic parasites, bacteria and viruses (Canter and Knox, 1985; Anderson et al., 1981; Tyler et.al., 1977). The development of the biomat can be enhanced in anaerobic soil by the growth of slimes and deposition of ferrous sulfide (Cotteral et.al., 1969; Anderson et.al., 1981).

Even though the biomat becomes a barrier to the flow and creates unsaturated conditions below the mat, it can cause a "back-up" and spill over to the surface of the ground. To avoid the two main types of system failure, failure to treat and failure to accept the daily flow, clogging control methods have been developed (Tyler et.al., 1977; Otis et.al., 1977). The most common methods are: matching hydraulic loadings with soil material, infiltrative surface geometry and depth, wastewater application, pretreatment, and resting (Otis, 1984). Clogging control by hydraulic loadings is based in loading the system at certain rates which assure an infiltrative equilibrium through the clogged zone. Surface geometry and depth considers the drainfield design for clogging control. Intermittent discharge or dosing has been shown to decrease clogging. Pretreatment deals with improving the wastewater before drainfield application. Finally, resting consists of letting the drainfield stand idle for a specific amount of time (Otis, 1984; Otis et.al., 1977; Winneberger, Vol.I 1984).

## 2.2 GROUND WATER HYDROLOGY IN COASTAL AREAS

### 2.2.1 Basic Ground water Hydrology

The world's water balance shows saline water in the oceans accounts for 97.2% of the total and inland water accounts for the remaining 2.8%. Ground water accounts for 0.61% of the water balance, which translates to 95 to 98% of the available fresh water for human consumption (Fetter, 1994; Freeze and Cherry, 1979).

Ground water is found in two zones: the vadose or unsaturated zone, and the saturated zone. The vadose zone is located between the ground level and the saturated zone. In the vadose zone, the soil voids are partially or totally filled with air. Here the water is held to the soil particles by capillary forces; thus, the water is below atmospheric pressure. In the saturated zone, the voids are filled with water at pressures above atmospheric. The boundary between the saturated and the vadose zone is called the capillary fringe. The fluid surface were the water pressure is equal to atmospheric pressure is called the water table, which coincides with the boundary between the saturated zone and the capillary fringe (Freeze and Cherry, 1979).

The main geologic units from the standpoint of ground water are aquifers and confining beds. An aquifer is a geologic formation sufficiently permeable to transmit and yield sufficient amounts of water. Confining beds can be classified as aquicludes or aquitards. An aquiclude is a confining layer which is essentially impermeable. However, an aquitard is permeable enough to transmit water vertically but not laterally. An aquifer can be confined or unconfined. A confining aquifer is defined as an aquifer bounded by aquicludes. On the other hand, an aquifer bounded by one or two aquitards is called an unconfining aquifer (Bouwer, 1978; Freeze and Cherry, 1979; Heath, 1983).

The three basic hydrologic parameters that describe and quantify the physical properties occurring in aquifers and the vadose zone are: hydraulic head, hydraulic conductivity and intrinsic permeability (Bouwer, 1978; Freeze and Cherry, 1979; Heath, 1983; Price, 1985).

Fluid flow through porous media is a mechanical process. Among the different types of mechanical energies, the most important for ground water movement are kinetic, gravitational potential, and fluid pressure energies. A unit mass of fluid has a fluid potential $\phi$ (usually known as the hydraulic head (h)), defined as the sum of its kinetic, gravitational, and fluid pressure energy.

$$\phi = gz + \frac{v^2}{2} + \int_{p_o}^{p} \frac{dp}{p}$$

where $\phi$ = fluid potential (the mechanical energy per unit mass)

$gz$ = gravitational potential energy

$v^2/2$ = kinetic energy

$\int dp/p$ = fluid pressure energy

Considering that in porous media flow, the kinetic energy is negligible and, working in gage pressures ($p_o$=0) and measuring the gravitational potential from a datum level arbitrarily selected as z=0, the hydraulic head can be expressed by:

$$h = z + \frac{p}{\rho g}$$

where: h = hydraulic head

z = elevation head

$p/\rho g$ = pressure head

The elevation head is measured as the elevation of the point in reference to a datum level. The pressure head is equal to the weight of the overlying water per unit cross-sectional area, measured as the height of the water column above the point. Consequently, the pressure head is positive in the saturated zone, zero at the water table and negative in the vadose zone. The hydraulic head is usually expressed as the length [L] of water. The dimensions of the three heads are those of length [L], i.e., meters.

The hydraulic conductivity (K, [length/time]) of a medium is its ability to transmit a specific fluid. The hydraulic conductivity depends on the medium (i.e., soil) and the dynamic characteristics of the fluid. The specific or intrinsic permeability (k, [length$^2$] or [darcy]) is only a function of the medium. The intrinsic permeability is related to the hydraulic conductivity K by ($k=K\times\nu/g$), where $\nu$= kinematic viscosity.

Ground water movement in aquifers is calculated by Darcy's law, which states that ground-water moves in the direction of decreasing hydraulic heads.

$$Q = -KA\frac{dh}{dl}$$

where

$Q$ = volumetric discharge

$K$ = hydraulic conductivity

$A$ = cross-sectional area

$dh/dl$ = gradient of hydraulic head

### 2.2.2 Ground water Hydrology in Coastal Areas

Ground water hydrology near coastal areas is influenced by the effects introduced by ocean waters, density differences, and tidal effects. When fresh ground-water comes into contact with, or discharges into the ocean, (or any other body of water with different physical characteristics) the process is controlled by the difference in densities. Near coastal areas, there is a ground water flow of both fresh and saline water. Given that the density of fresh water ($\rho_f$) is lower than salt water ($\rho_s$), fresh ground water flows on top of the salt water and creates a zone of variable density that is usually called the saltwater interface (Fetter, 1994; Freeze and Cherry, 1979). The saltwater interface can be described by the Ghyben-Herzberg relation

$$z(x,y) = \frac{\rho_f}{\rho_s - \rho_f} h(x,y)$$

where

$z(x,y)$ = depth of the saltwater interface below sea level at location $(x,y)$

$h(x,y)$ = elevation of the water table above sea level at point $(x,y)$

$\rho_f$ = density of fresh water

$\rho_s$ = density of salt water

The Ghyben-Herzberg relation considers hydrostatic conditions and a sharp boundary interface. In reality, a zone of diffusion exists around the interface and the Ghyben-Herzberg relation underestimates the depth of the interface. A more realistic approach is given in Hubbert (1940); and different mathematical solutions can be found in Fetter (1972); Polubarinova-Kochina (1962); Cooper et al. (1964); and Glover (1964).

Ground water hydraulic gradients are influenced by variable densities. Ground water with constant density, or vertical density variations, or surface of equal density coincident with surfaces of equal pressure, have a potential (Hubbert, 1957). Thus, hydraulic gradient can be determined from well data when one of these three conditions is met (Hickey, 1989). On the other hand, Hubbert (1957) additionally stated that if vertical and lateral density variations exists, and/or equal density surfaces do not coincide with equal pressure surfaces, then the ground water does not have a potential. Thus, in these situations hydraulic gradients cannot be determined from well data (Hickey, 1989). Very few researchers have addressed the relationship between ground water potential and density in coastal areas. Jorgensen et al. (1982) discussed three methods actually used by researchers to overcome this problem, and provided an alternative method. The three methods used are: equivalent freshwater head, Hubbert method, and Darcy method.

The equivalent freshwater head method determines a freshwater hydraulic head from the actual hydraulic head by the relationship between the densities of the actual water and the fresh water. The equivalent freshwater head is defined as $h_f = (\rho_w / \rho_f) \times h_w$ , where $\rho_f$ =fresh water density and $\rho_w$ = actual water density. Then the fresh hydraulic head are used to determine ground-water flow directions. The weakness of this method is that the distance which defines the hydraulic head $h_w$ (aquifer depth) must be determined accurately.

Hubberts method determines ground water flow directions by computing the potential $\Phi = g \times h$, where h is the hydraulic head previously defined. The flow direction is determined by the slope of a tangent drawn to the curve obtained by plotting the potentials versus distances between wells. The weakness of this method are; the wells must be in a line, densities must vary in a vertical direction, and there must not be any intervening sources or sinks. Flow directions are determined by the Darcy's method by the calculation of flow vectors between the wells. Given that the calculation of the Darcy's velocities are dependent in the aquifer parameters, i.e. intrinsic permeability, pressure, density, and fluid viscosity, this is dependent in the availability of these data.

The alternative method reported by Jorgensen et al. (1982) is based on the determination of existent hydrostatic conditions. The method considers each pair of wells as a U-tube manometer. The pressure of a well is determined by the pressure of the second well plus the pressure between the wells. The pressure between the wells is a function of the difference in densities of the two wells. Flow would exist if these pressures were different. The method's weakness is that it assumes hydrostatic conditions between the two points with no external sources or sinks.

Aquifers near coastal areas are subject to pressure waves caused by tidal oscillations. These pressure waves induce hydraulic gradients and cause ground water levels to fluctuate as they travel inland (Jacob, 1940; Ferris, 1951). Water level measurements cannot be used to accurately characterize ground water flow if pressure waves are not considered. Ground water level fluctuation by tidal effects have been studied by Ferris (1951); Gregg (1966); Vacher (1978); Serfes (1987), (1991), and others.

Ground water flow directions determined by a single set of hydraulic head measurements would give erroneous results. In order to accurately determine ground water flow direction in an environment affected by tidal fluctuations, mean hydraulic gradients must be determined by monitoring hydraulic gradients over several tidal cycles, and computing the mean value of each hydraulic gradient (Serfes, 1991).

## 2.3 TRANSPORT AND FATE OF FECAL-COLIFORM BACTERIA UNDER SEPTIC TANK FIELDS

The volume, composition, and consistency of feces and urine depends on diet, climate, and health. The average fecal weight in Europe and the United States is 150 grams/day while in developing countries it is 350 grams/day. The average urine weight per day produced by an adult is between 1.0 and 1.3 kilograms (Feachem et al, 1983).

Human excreta has four main groups of pathogens: bacteria, viruses, protozoa, and worms. Tables 4 through 6 show the most important pathogenic elements for each particular group. For additional information on the fate and transport of viruses in ground water the following references are suggested: Kator, 1992; Farrah and Bitton, 1990; Feachem et al., 1983; and Goddard and Butler, 1981. For further information on protozoa and worms in human excreta, the reader is referred to Feachem et al., 1983.

Human feces contain a great number of different bacterial species. The most common bacteria are enterobacteria, enterococci, lactobacilli, clostridia, *Bacteroides*, bifidobacteria, and eubacteria. In contrast, human urine is generally a sterile and harmless substance. Schistosomiasis,

Table 4. Pathogens Excreted in Feces (a) Viruses, (b) Protozoal.
(Feachem et al., 1983)

(a)

| Virus Group | Disease |
|---|---|
| **Adenoviruses** | Respiratory disease, eye infections |
| **Enteroviruses** | |
| **Polioviruses** | Poliomyelitis, meningitis, fever |
| **Echoviruses** | Meningitis, encephalitis, respiratory disease, acute hemorrhagic conjunctivitis, fever |
| **Coxsackie viruses** | Meningitis, herpangina, myocarditis, disease, pleurodynia, diarrhea, rash, fever |
| **Hepatitis A virus** | Infectious hepatitis |
| **Reoviruses** | Not clearly established |
| **Rotaviruses, Norwalk agent and other viruses** | Vomiting, diarrhea |

(b)

| Protozoan | Disease |
|---|---|
| ***Balantidium coli*** | Diarrhea, dysentery and colonic ulceration |
| ***Entamoeba histolytica*** | Colonic ulceration, amebic dysentery, and liver abscess |
| ***Giardia lamblia*** | Diarrhea and malabsorption |

Table 5. Bacterial Pathogens Excreted in Feces
(Feachem et al., 1983)

| **Bacterium** | **Disease** |
|---|---|
| ***Campylobacter fetus ssp. jejuni*** | Diarrhea |
| **Pathogenic *Escherichia coli*[a]** | Diarrhea |
| ***Salmonella spp.*** | |
| ***S. typhi*** | Typhoid fever |
| ***S. paratyphi*** | Paratyphoid fever |
| **Other salmonellae** | Food poisoning and other salmonelloses |
| ***Shigella spp.*** | Bacillary dysentery |
| ***Vibrio spp.*** | |
| ***V. cholerae*** | Cholera |
| **Other vibrios** | Diarrhea |
| ***Yersinia enterocolitica*** | Diarrhea and septicemia |

a. Includes enterotoxigenic, enteroinvasive, and enteropathogenic *E. coli*

Table 6 . Helminthic Pathogens Excreted in Feces.
(Feachem et al., 1983)

| Helminth | Common Name | Disease |
|---|---|---|
| ***Ancylostoma duodenale*** | Hookworm | Hookworm |
| ***Ascaris lumbricoides*** | Round worm | Ascariasis |
| ***Clonorchis sinensis*** | Chinese liver fluke | Clonorchiasis |
| ***Diphyllobothrium latum*** | Fish tapeworm | Diphyllobothriasis |
| ***Enterobius vermicularis*** | Pinworm | Enterobiasis |
| ***Fasciola hepatica*** | Sheep liver fluke | Fascioliasis |
| ***Fasciolopsis buski*** | Giant intestinal fluke | Fasciolopsiasis |
| ***Gastrodiscoides hominis*** | n.a. | Gastrodiscoidiasis |
| ***Heterophyes heterophyes*** | n.a. | Heterophyiasis |
| ***Hymenolepis nana*** | Dwarf tapeworm | Hymenolepiasis |
| ***Metagonimus yokogawai*** | n.a. | Metagonimiasis |
| ***Necator americanus*** | Hookworm | Hookworm |
| ***Opisthorchis felineus*** | Cat liver fluke | Opisthorchiasis |
| ***O. viverrini*** | n.a. | |
| ***Paragonimus westermani*** | Lung fluke | Paragonimiasis |
| ***Schistosoma haematobium*** | Schistosome | Schistosomiasis; bilharziasis |
| ***Strongyloides stercoralis*** | Threadworm | Strongyloidiasis |
| ***Taenia saginata*** | Beef tapeworm | Taeniasis |
| ***T. solium*** | Pork tapeworm | Taeniasis |
| ***Trichuris trichiura*** | Whipworm | Trichuriasis |

typhoid, and leptospirosis are the principal urine infectious diseases that lead to appearance of pathogens in the urine (Feachem et al., 1983).

Certain types of fecal bacteria are used as indicators of fecal contamination. These include total and fecal coliforms, fecal streptococci and the anaerobic bacterium *Clostridium perfringens*. Table 7 shows the concentration of these indicator bacteria in human feces. Fecal coliform group may contain bacteria genera of non fecal origen such as, *Klebsiella*, *Citrobacter*, and *Enterobacter* as well as *Escherichia coli*. Among these three genera, *E. coli* is the bacterium generally used as an indicator for fecal pollution (Feachem et al, 1983; Tchobanoglous and Burton, 1991; Kator, 1992).

### 2.3.1 Bacterial Survival in the Soil System

Soil is one of the most complex microbial habitats. Whereas the abiotic soil components have been relatively well explained, the microbiotic soil components, as well as, the interaction between abiotic and microbiotic components are not clearly defined (Stotzky, 1985). Bacterial survival is affected by both environmental (abiotic) and biological (microbiotic) factors in the soil environment. The most important environmental factors are soil moisture, temperature, pH, and organic matter.

Soil moisture appears to be one the most important environmental factors governing bacterial survival (Gerba et al.,1975; Reddy et al., 1981). Morita (1992) stated that soil moisture provides the adequate system for the solute (nutrients) to be in contact with the organisms. As the soil moisture decreases, the availability of nutrients for bacterial growth decreases. Bacterial movement becomes negligible in soils drained to matric potential

Table 7. Number of Bacteria Commonly Found in Human Feces.
(Feachem et al., 1983)

| **Bacteria** | **Cells per gram of feces (wet weight)** |
|---|---|
| ***Bacteroides spp.*** | $10^{7}$-$10^{11}$ |
| ***Bifidobacterium spp.*** | $10^{7}$-$10^{11}$ |
| ***Clostridium perfringens*** | $10^{3}$-$10^{10}$ |
| **Coliforms** | |
| **Fecal** | $10^{6}$-$10^{9}$ |
| **Nonfecal** | $10^{7}$-$10^{9}$ |
| **Fecal streptococci** | $10^{5}$-$10^{8}$ |

between -0.2 and 1 bar. When the rate of substrate supply becomes less than the substrate requirements, the organisms become energy starved. Therefore, as the soil moisture increases so does the availability of nutrients in the bulk soil solution.

Gerba et al. (1975) stated that low temperatures favor bacterial survival. Reddy et al., (1981) concluded from data presented by several workers that die-off rates double for every 10°C in increase temperature. Kator (1992) cited some works where positive (Anderson et al., 1983; Rhodes and Kator, 1988) and negative (Faust et al., 1975; Vasconcelos and Swartz, 1976; Lessard and Sieburth, 1983) correlations of temperature and survival were found for *E. coli* in surface waters. Kator (1992) explained that temperature has positive, as well as negative effects on bacterial survival. He stated that under suitable circumstances warmer temperatures can have a positive effect on bacterial survival, alternatively low temperatures can affect bacteria in a negative way through sublethal stress.

Acid soils have an adverse effect on bacterial survival. The pH range 5.8 to 8.4 seems to be the most acceptable for bacterial survival (Lambert, 1974; McFeters and Stuart, 1972; Cuthbert et al., 1950). Organic matter or nutrient availability is the most important environmental factor for the regrowth of any bacteria (Morita, 1992). Bacterial survival and the possibility of regrowth increases as the organic matter increases (Gerba et al., 1975). Finally, bacterial survival is affected by biological factors, for example, competition for nutrient availability and/or predation (Kator, 1992; Morita, 1992)

Given that the study site is located in a coastal area, bacterial survival will be affected by salinity. Evaluation of the influence of salinity on *E. coli* has been made in marine or estuarine environments (Anderson et al., 1979; Vasconcelos and Swartz, 1976; Faust et al., 1975; Orlob, 1956). Salinity has a negative effect on *E. coli*, due to sublethal stress associated with increasing salinity.

### 2.3.2 Bacterial Transport Through Soils

Two primary mechanisms reduce bacterial concentration as they travel through the soil in the percolating water. These are the physical process of straining, and the chemical process of adsorption (Canter and Knox, 1985, Hagedorn, 1984; Hagedorn et al., 1981; Gerba et al., 1975; Reneau et al., 1989). Hagedorn et al. (1981), concluded based on several studies, microorganisms move only a few dozen of centimeters in unsaturated soils, bacterial retention is inversely proportional to the particle size of the soil, physical straining is the main travel limitation, and adsorption is more effective with increasing soil clay content.

Given that physical straining is inversely proportional to particle size, the largest suspended particles and bacteria in the effluent water become strained in the soil particles, and become themselves, part of the filter process. Gerba et al. (1975) stated that the greatest bacterial removal occurs at the soil surface where the biological mat occurs (top 2 to 6 mm). Kristiansen (1981) concluded that the production of polysaccharides by bacteria in the biological mat served as a bind or cement to other cellular material. As part of the work done by University of Wisconsin (1978), Ziebell et al. (1975b) measured fecal streptococci, fecal and total coliform concentrations in the soil of an absorption field (Plainfield loamy sand). The fecal coliform concentrations as MPN per 100 grams of soil were: at the liquid on top of the clogged zone, $1.9 \times 10^6$; in the biomat, $4 \times 10^6$; and 30 cm below the biomat, <200. This is in agreement with column tests (Ziebell et al, 1975a; McCoy and Ziebell, 1975; Ziebell, 1975) where biomat formation decreased the bacteria counts in the column effluents.

Adsorption is the other process by which bacteria are removed by the soil. Little is known about direct surface interaction between soil components and microbes *in situ* (Stotzky, 1985). No single mechanism can be attributed for the adsorption phenomenon. On the contrary, adsorption appears to be a function of the physiochemical characteristics of both the soil and the microbes. Hattori and Hattori (1976) offer an excellent summary of the principal physical interactions between microbes and soil. In their work, they cite Pethica (1961), who stated that the possible forces of sorption between cells and surfaces are chemical bonds between the opposed surfaces, ion-pair and ion-triplet formation, forces due to charge fluctuations, charge mosaics on surfaces of like or opposite overall charge, charge attraction of opposite signs, electrostatic attractions between surfaces of like charge, electrostatic attraction due to image forces, surface tension or surface energy, van der Waal's forces, charge repulsion between surfaces of like charge, van der Waal's forces of repulsion, and hindrance to attraction due to steric barriers such as inert capsules and solvated layer.

Stotzky (1985) in his extensive analysis of microbial adhesion to soils emphasized that "the major inorganic particulates that affect microbial events in soil are within the clay-sized fraction and consist primarily of clay minerals and polymeric hydrous oxides of mainly $Fe^{3+}$, $Al^{3+}$, and $Mn^{4+}$". One of the most important soil characteristics in relation to bacterial adhesion is the CEC:AEC (Cation Exchange Capacity:Anion Exchange Capacity) ratio. The cation and anion exchange capacity of a soil is primarily due to the clay and organic matter content. The CEC:AEC ratio determines the ability for a soil particle to repel particles of negative charge. As the ratio increases, this ability is reduced (Lipson and Stotzky, 1987). Clay, as well as organic particles and biological entities under most soil pH values, have a net negative charge (Stotzky, 1985). Given the complexity of the soil environment, these entities can adhere to each other by any of the previous processes mentioned by Pethica (1961).

Bacteria can be desorbed once adsorbed to a soil particle. The most important factors in the soil environment which can affect desorbtion are pH and ionic strength (Stotzky, 1985). Increases in fecal indicator bacterial concentrations have been measured during or after rainstorms (Reneau et al., 1975; Gerba et al., 1975; Hendricks et al., 1979; DeWalle et al.,1980).

## 2.4 TRANSPORT AND FATE OF NUTRIENTS UNDER SEPTIC TANK FIELDS

The chemical composition of human feces and urine is given in Table 8. The C:N ratio is approximately 8.0 for feces and less than 1.0 for urine. About 3.5 grams per capita daily of toilet paper is used in United States (Laak, 1974). Walker et al.(1973b) state the average nitrogen input per person is 8.0 kilograms per year in wastewater. Bernhard (1975) estimated an average phosphorus input of 3.0 kilograms per person per year in wastewater. The Water Pollution Control Federation (1983) stated the per capita phosphorus generation (kg/yr) is reported to be: total, 0.8-1.8; organic, 0.3-0.6; and inorganic, 0.5-1.2. Nitrogen generation is reported to be (kg/per-capita/yr): total, 3.4-5.0; organic, 0.7-1.0, and free ammonia/ammonium, 2.1-4.0.

Carbon, potassium and calcium are not considered to be dangerous to humans or the environment at the concentrations introduced by home sewage waters. Conversely, nitrogen and phosphorous can enhance eutrophication and $NO_3$-N can cause methemoglobinemia, or blue baby

Table 8. Composition of Human Feces and Urine.

(Feachem et al., 1983)

| | Approximate composition (percent of dry weight) | |
|---|---|---|
| **Constituent** | Feces | Urine |
| **Calcium (CaO)** | 4.5 | 4.5-6.0 |
| **Carbon** | 44-55 | 11-17 |
| **Nitrogen** | 5.0-7.0 | 15-19 |
| **Organic Matter** | 88-97 | 65-85 |
| **Phosphorus ($P_2O_5$)** | 3.0-5.4 | 2.5-5.0 |
| **Potassium ($K_2O$)** | 1.0-2.5 | 3.0-4.5 |

syndrome, at concentrations higher than 10 mg/l (Kroehler, 1990).

### 2.4.1 Nitrogen

Nitrogen undergoes a number of transformations in the absorption system (Figure 5). The basic nitrogen forms entering the drain field are: ammonium-N ($NH_4^+$) and organic-N ($R\text{-}NH_2$), on average rates of 75 and 25 percent, respectively (Canter and Knox, 1985). Once organic nitrogen is in the soil, it is eventually mineralized to ammonium. The mineralization of organic nitrogen to ammonium (ammonification) is optimum at 50 to 75 percent of the soil water-holding capacity, it is an aerobic or anaerobic process, and its rate increases as the soil is subject to drying and wetting cycles (Alexander, 1977).

Given the low C:N ratios in the wastewater (approx. 10:1, Sikora and Corey, 1976), the immobilization of nitrogen by biomass assimilation is minimal. Thus, little nitrogen immobilization occurs in the biomat (critical C:N ratios are 20 to 30:1. (Alexander, 1977). Once the ammonium reaches the soil environment, it is quickly sorbed to soil particles by cation exchange processes. This sorption process will continue until an equilibrium is reached with the cations in the effluent. From that point on, ammonium can leach to the ground water (Sikora and Corey, 1976).

Ammonium is oxidized by autotrophic bacteria to nitrate (nitrification) under aerobic conditions. Nitrification rates are affected by the soil environment, principally by acidity. Soils with pH below 6 have very little nitrification rates and these rates are negligible at pH below 5. Optimal pH values are between 6.6 to 8 or higher (Alexander, 1977). Oxygen is an obligate requirement for nitrification to occur. Minimum levels at which nitrification occurs on solid media is 0.08 mg $O_2$/L, and on liquid media

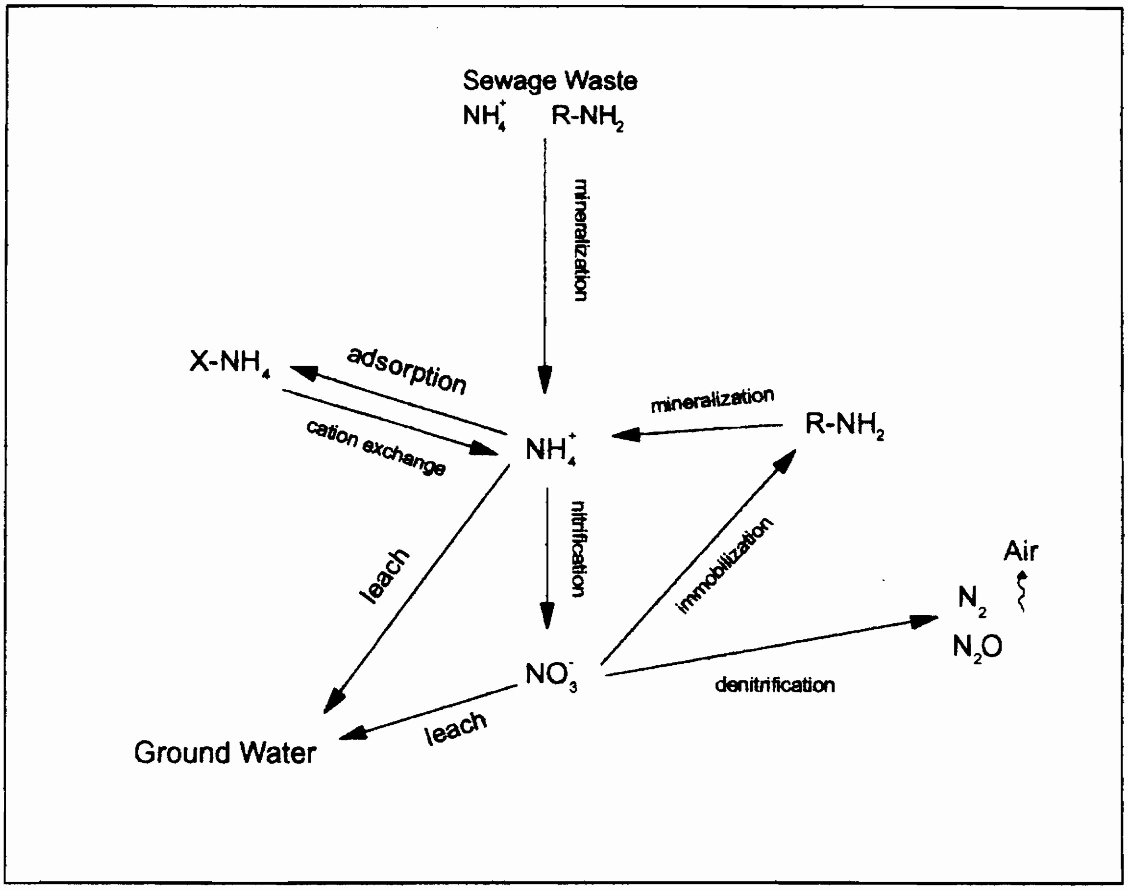



Figure 5. Nitrogen transformations in the absorption system (Adapted from Freeze and Cherry, 1979).

0.10 mg $O_2$/L (Kaplan, 1983).

The three main mechanisms that remove nitrogen from the soil environment are: denitrification, volatilization of free ammonia, and plant assimilation. The transformation of aqueous ammonia to ammonium ion is about half complete at a pH approximately of 9.24 (Hem, 1985) ($pK_b$=4.7 at 25°C); thus, at pH's generally found under absorption field (Sikora and Corey, 1976), the ammonia nitrogen in solution would have the form $NH_4^+$. Volatilization of free ammonia is enhanced in alkaline conditions and at warm temperatures. Ammonification can raise the pH to levels where volatilization occurs (Black, 1984; Alexander, 1977). Plant assimilation is heavily dependent on the plant type. A detailed and comprehensive description of plant assimilation of nitrogen is given in Black (1984); Alexander (1977); and Bartholomew and Clark (1965).

The major mechanism for nitrogen reduction is denitrification which is a reduction of nitrate to nitrogen gas and nitrogen oxides. Denitrification is the gaseous loss of nitrogen to the atmosphere by either biological or chemical mechanisms (exclusive of ammonia volatilization). Denitrifying bacteria are generally facultative aerobes that will use nitrate as an electron acceptor when oxygen supply is low. The following pathway lists the intermediate compounds, along with their respective oxidation states, for denitrification.

$$\begin{array}{c} NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \\ (+5) \rightarrow (+3) \rightarrow (+2) \rightarrow (+1) \rightarrow (0) \end{array}$$

In order for denitrification to occur, a quasi anaerobic environment and adequate carbon source must coexist (Alexander, 1977; Broadbent and Clark, 1965).

For denitrification to occur in an absorption system, an aerobic reaction must first occur followed by an anaerobic one (Lance, 1972). Cogger and Carlile (1984) concluded that after continuously saturating the soil system in order to provide an anaerobic environment, dilution and not dinitrification was the main factor of nitrogen reduction. Dilution was also found to be the primary cause of nitrogen reduction by Walker et al., (1973b); Dudley and Stephenson, (1973); and Pruel, (1967). Sikoro and Keeney (1974) and Stewart et al. (1979) concluded that the availability of an adequate energy source was the limiting factor for dinitrification to occur. Reynolds et al.(1979) found denitrification was enhanced by the placement of an impermeable shield below the drain field to generate an anaerobic environment and by the usage of methanol as an carbon source. Eastburn and Ritter (1984) reported that in a conventional system denitrification can be enhanced by dosing procedures. For mound systems, Harkin et al. (1979) found that denitrification removed on average of 44 % of the nitrate formed in a study of thirty-three mound systems.

### 2.4.2 Phosphorus

Most of the phosphorus in a septic tank effluent is in the soluble orthophosphate form (Canter and Knox, 1985; Sikora and Corey, 1976). From Table 2 the typical total phosphorus concentration in a septic tank effluent is 13 mg/l. Phosphate ions, once in the soil environment, are readily chemisorbed to mineral surfaces. In acid to neutral soils the chemisorption is mainly to Fe and Al, in neutral to alkaline soil, the chemisorption occurs to Ca minerals. Soil beneath absorption systems are exposed to high concentrations of phosphorus and many precipitates may be formed (University of Wisconsin, 1978; Sikora and Corey, 1976; Lindsay and Moreno, 1960).

The amount of phosphorus that can be retained by a soil will depend on the characteristics of the soil. Sawhney and Hill (1975) measured 90 mg/g phosphorus sorption capacity for Merrimac soil and 290 mg/g for a Paxton soil. University of Wisconsin (1978) cites studies in which the phosphorus retained by a sandy loam soil was, 100 to 300 mg/g and 121 mg/g for a sand column. The concluded that with these data, the phosphorus penetration per year for a sandy soil would be 50 cm and for a finer textured soil as low as 10 cm. Sikora and Corey (1976) concluded that in an extreme case, a drainfield in a very sandy soil, considering average input phosphorus values (Otis et al., 1975), and using a Langmuir adsorption isotherm with a maximum of 90 mg/g, phosphorus will saturate the soil at a rate of 104 cm/year. Sawhney and Starr (1977) measured concentrations of 0.5 mg/l of soluble inorganic phosphorus at 60 cm depth below the drain field trench in a six year old system. In conclusion, the capacity of phosphorus retention by soils will depend on the site characteristics. Nevertheless, typical soils effectively reduce phosphorus concentrations and generally very low concentrations are introduced to the ground water. Phosphorus contamination of groundwater is possible when the sorptive capacity of the soil is exceeded, hydraulic loadings and the percolation rates are high, and/or there are elevated water tables (Canter and Knox, 1985; Sawhney and Starr, 1977; Sikora and Corey, 1976). Details and comprehensive descriptions of phosphorus in soils are given in Black, 1984; and of phosphate compounds in soil in Lindsay and Moreno, 1960.

# 3.0 STUDY SITE DESCRIPTION

Research was conducted in Assateague Island National Seashore Park located in the northern portion of the Delmarva peninsula (Figure 6). The unconfined ground-water system is designated as the Columbia aquifer, which is underlain by the Yorktown-Eastover confined aquifer approximately 45 meters below sea level. The upper Yorktown-Eastover confining unit is at 11.9 meters below sea level (USGS Well 68M4, Meng and Harsh, 1988). The site is characterized by coastal beach soil (Stevens, 1920). A discontinuous thin peat layer 0.1-0.3 meters thick was located approximately 0.5-0.7 meters below ground level. Mean tidal range was 1.1 meters with spring ranges of 1.3 meters (NOAA, 1990). The ground water table was on the order of 1.3 meters below ground-level.

Research focused on the Parking Lot #1 and #2 comfort stations, located at the southern part of the island (37°51'N, 75°20'W) (Figure 7). Comfort station Parking Lot #2 was located approximately 3200 meters from the island's southern most tip. The comfort station had the Atlantic Ocean to the southeast, and Little Toms Cove (a tidal flat marsh) to the northwest (USGS, 1980). The island width at that point was approximately 200 meters. Comfort station Parking Lot #1 was located approximately 500 meters to the north-northeast from comfort station Parking Lot #2. The comfort station had the Atlantic ocean to the southeast, and the east portion of the Black Duck marsh lagoon to the northwest (USGS, 1980). The island width at that point was approximately 130 meters.

The two comfort stations had similar construction design; equal number of commodes (7), urine latrines (2), shower facilities (8), and lavatories (4). The wastewater of both stations was treated by OSWDS. The difference between the two OSWDS was the type of absorption system.

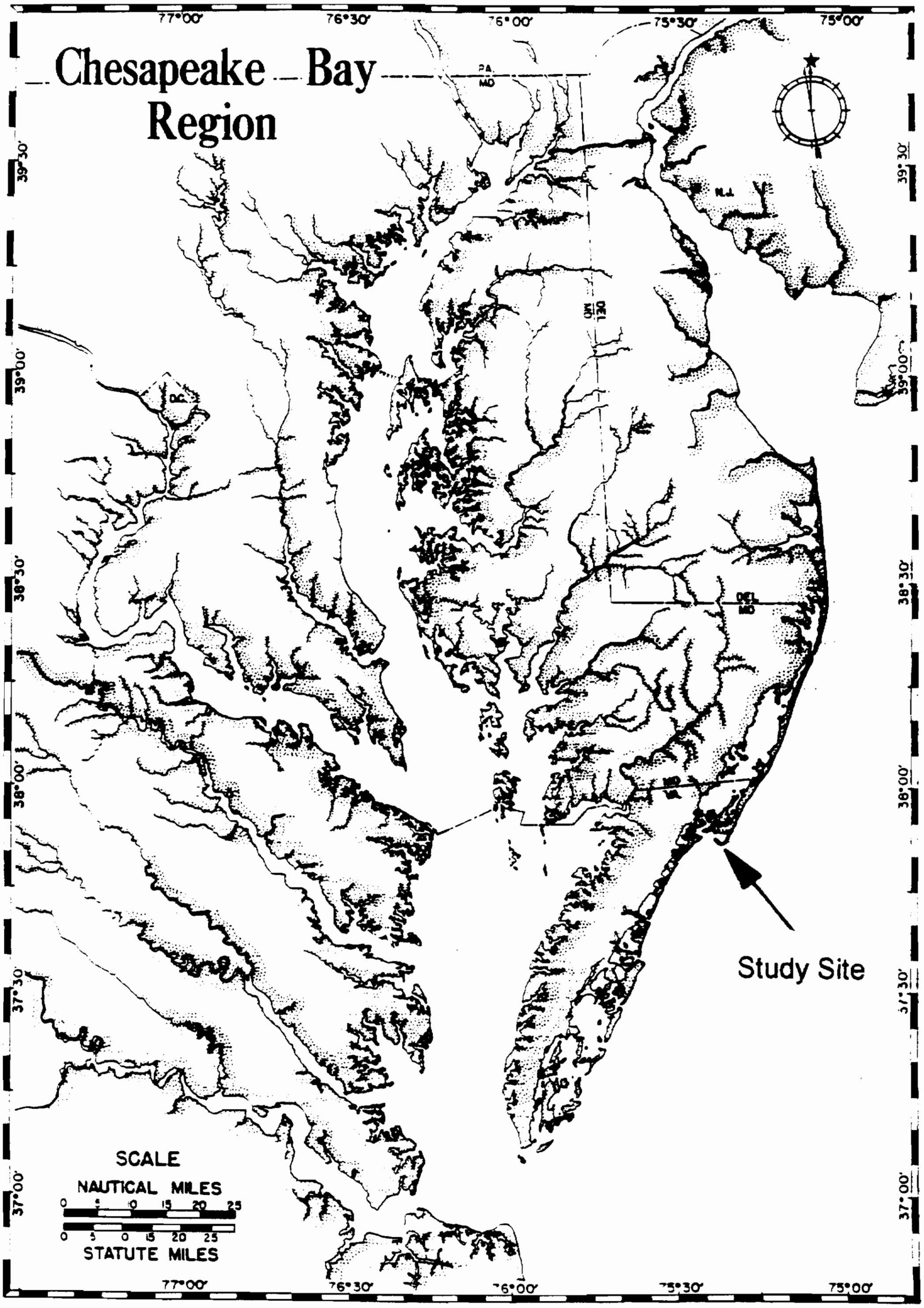



Figure 6. Study site. Chesapeake Bay region.

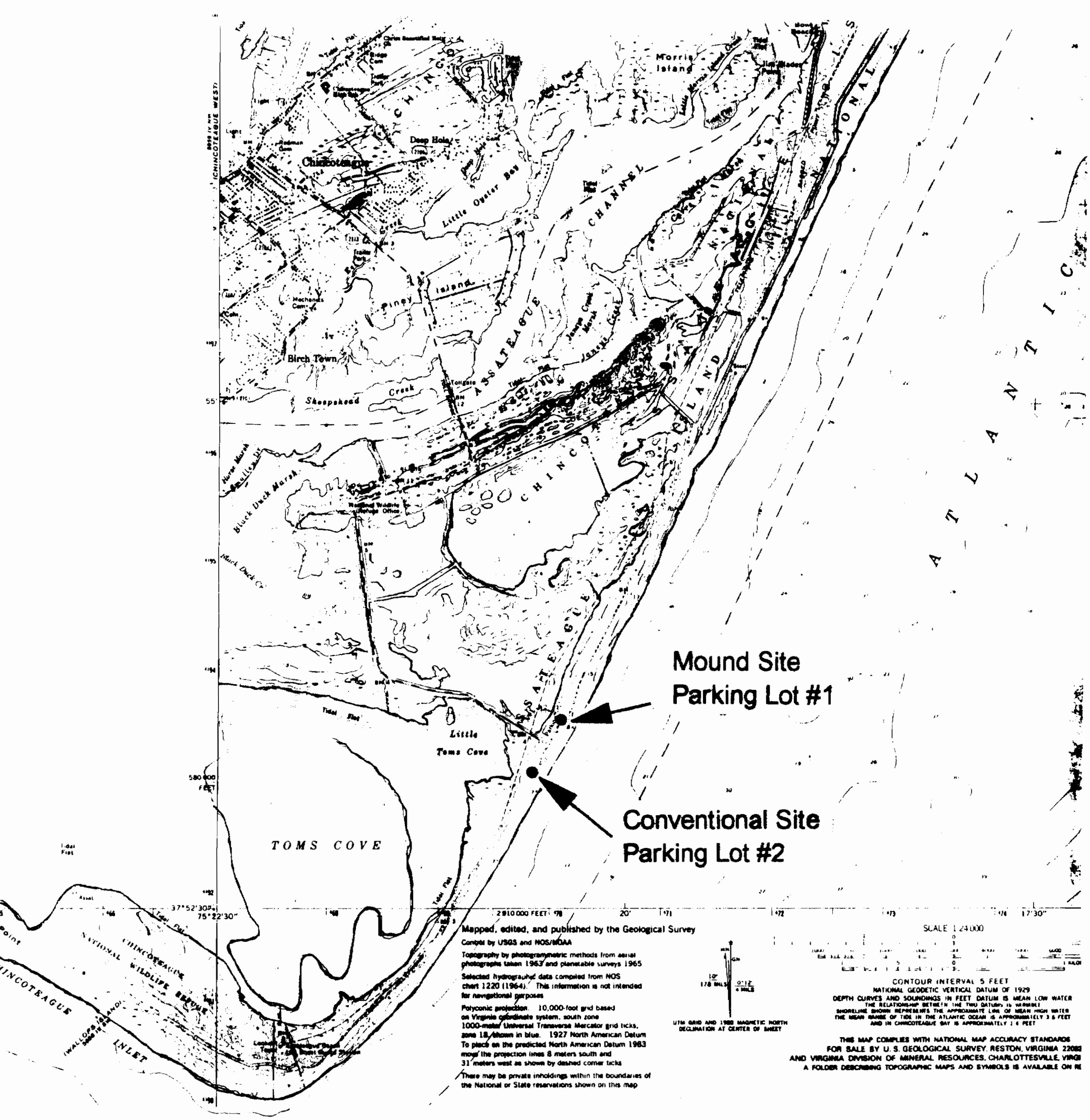



Figure 7. Study site. Assateague Island National Seashore Park.

While comfort station Parking Lot #1 had a mound system, comfort station Parking Lot #2 had a conventional absorption system. Two and three septic tanks, with a capacity of 10.6 $m^3$ each one were connected in series in Parking Lot #1 and #2, respectively.

The conventional absorption field (Parking Lot #2) was a disposal bed type (length = 18.3 m; width = 9.2 m ). The conventional absorption field was constructed during 1982. It contained five, 0.038 m ∅ (diameter), PVC distribution pipes. Spacing between the pipes was 1.83 m. Pipes were laid on top and covered by Virginia gravel # 57, with a minimum gravel backfill width of 0.3 meters.

The mound absorption field (Parking Lot #1) was a bed type with a wastewater treatment capacity of 11.36 $m^3$ (3000 gal) daily. The mound drainfield was constructed during May 1994. The basal area was 36 by 16 meters with a slope of 0.7%. The distribution network consists of a mainfold and laterals. A T to T construction with two sets of 13.4 m long, 0.038 m ∅ , PVC pipes, connected to the lift station pipe by a non-perforated mainfold 0.1 m ∅ PVC pipe. Spacing between pipes was 2.1 m. The fill material was sand spec C-33 with a minimum depth of 0.6 m. The distribution pipes were laid at a 0.3 m depth, on a Virginia #57 gravel bed. The minimum height of the mound was 1.37 m.

# 4.0 GEOHYDROLOGIC CHARACTERISTICS

## 4.1 Introduction

A basic knowledge of ground water hydrology is fundamental to the understanding of wastewater transport and fate at the research site. To acquire this basic knowledge, well transects were established to monitor water table fluctuations, sediment structure and hydraulic conductivities were determined.

## 4.2 Methodology

Monitoring wells were established around the perimeter of the conventional and mound drainfields. Three types of well devices were used in the monitoring process. Permanent wells were installed in the uplands near the vicinity of the drainfields. Temporary wells were employed to monitor water tables in the beach. These temporary wells were installed and later removed following each monitoring period. Finally, piezometers were used to monitor ground water hydrodynamics at specific depths.

Permanent and temporary wells were constructed with PVC piping (diameter ∅ = 3.8 cm permanent wells; ∅ = 5.1 cm temporary wells) with 40 cm (length) of a ∅ 5.1 cm PVC screen (0.025 cm slot width). For permanent wells, well screens were gravel packed and sealed with bentonite to eliminate vertical infiltration. No packing or sealing was used on temporary well screens. All wells were hand-augured (8.25 cm o.d.) and penetrated approximately 0.5-0.7 meters into the Columbia aquifer. Wells were logged and surveyed to a common reference point.

Three permanent wells were constructed of 2.54 cm PVC pipe with a 45.7 cm stainless steel well point. These wells were driven down at the conventional drainfield site. Two of these wells penetrated approximately 2.5 meters into the Columbia aquifer. The third well penetrated approximately 1.5 meters into the Columbia aquifer.

In order to study ground water quality changes on a small scale, a set of five piezometers were installed immediately adjacent to the conventional drainfield at 0.5 and 1.0 meters below water the table. Piezometers were constructed of 1.2 cm plastic tubes and followed the basic design of Lee and Cherry (1978). A manometer was used to measure hydraulic head differences between the piezometers. The manometer followed the basic design of Winter et al. (1988).

The following notation was employed for well differentiation; permanent wells were noted by a suffix W followed by a number from 1 to 13 at each site. Temporary wells were noted by a suffix B followed by a number or a letter and a number. Piezometers were noted by a suffix PZ followed by a number or a letter and a number. Driven wells are noted by a suffix D. The permanent well transects at the mound and conventional drainfield sites are shown in Figures 8 and 9. The piezometer set is shown in Figure 10. The temporary well transects for the mound and conventional drainfield sites are shown in Figures 11-12, and Figures 13-15, respectively. In order to facilitate the presentation of the results and discussions, a certain notation was used to distinguished the upland wells. Wells immediately adjacent to the drainfield were referred as first tier wells. The wells not immediately adjacent to the drainfield were designated second tier wells. The term "upland wells" was used to refer to both first and second tier wells. The first tier wells were further subdivided into two sections at the conventional drainfield. The distal half section of the drainfield consisting of

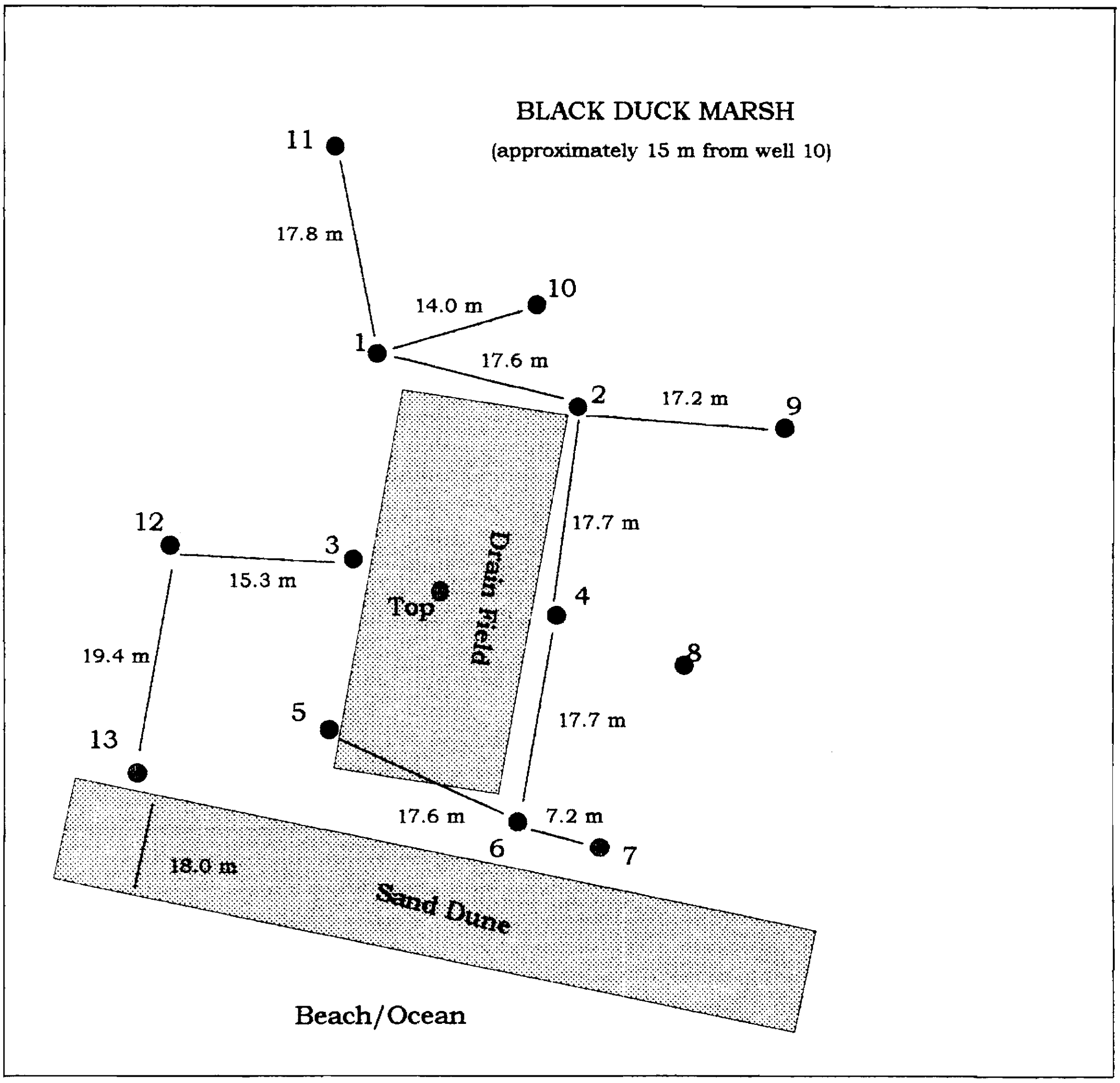



Figure 8. Permanent well transect for the Mound drainfield site. Distances are given in meters. The wells locations are drawn to scale (1cm = 6.1 m). The draindfield and the sand dune are not drawn to scale.

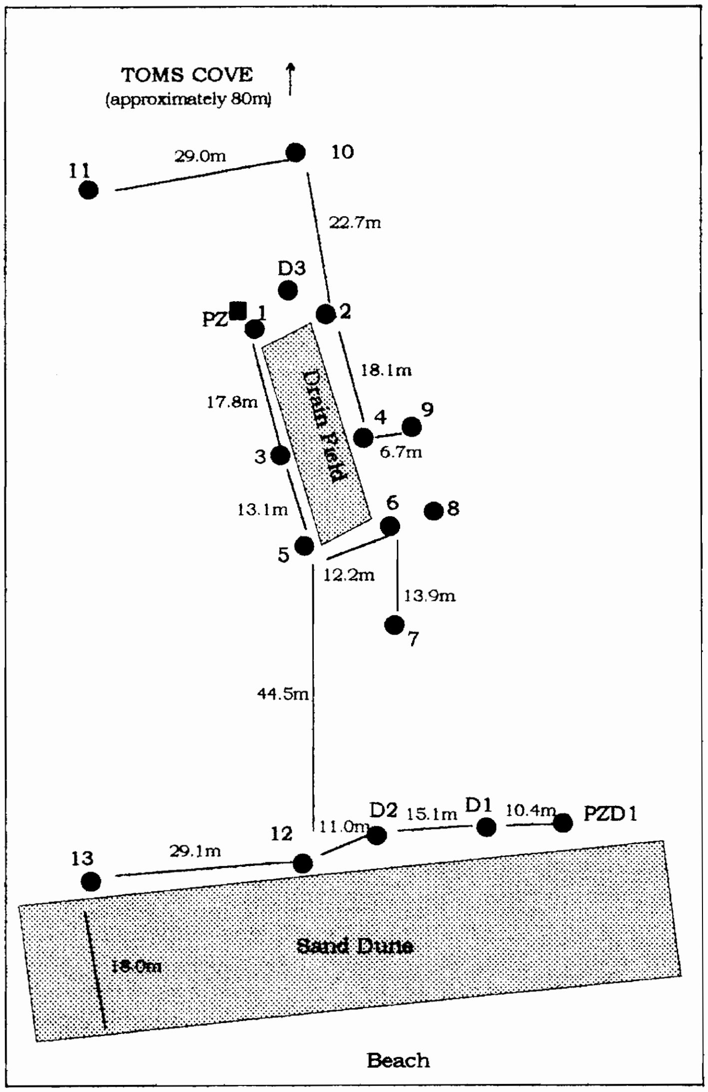



Figure 9. Permanent well transect at the Conventional site. Distance between wells is given in meters. The map is drawn to scale (1 cm = 8.6 m). The drainfield is not drawn to scale. The piezometer set is located at PZ, near well 1.

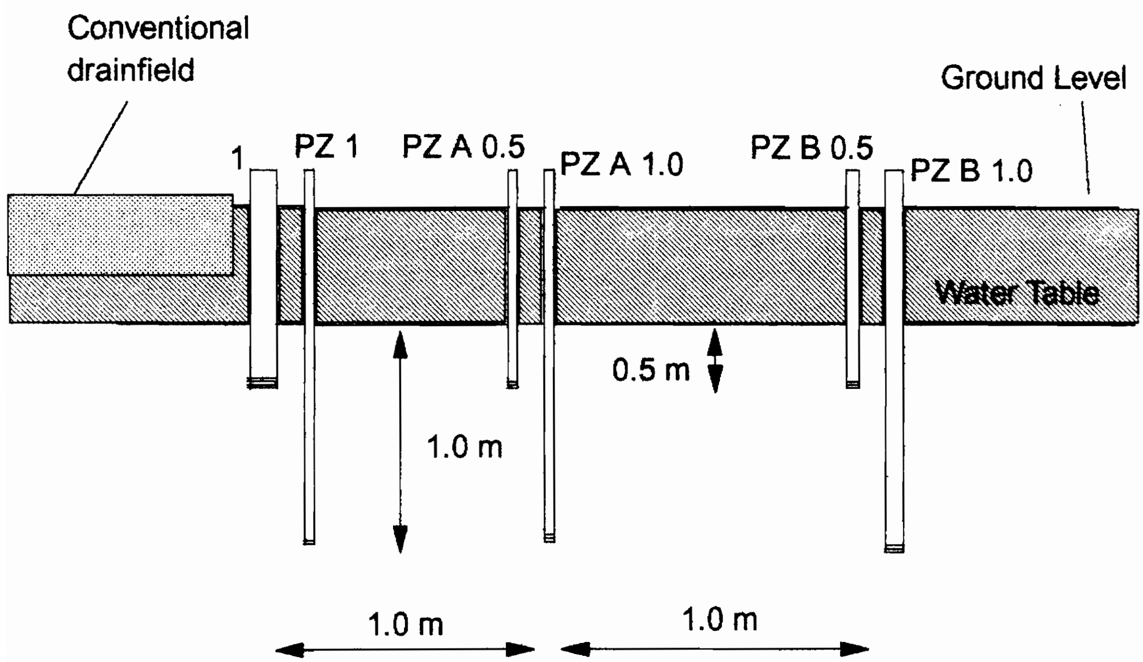



Figure 10. Piezometer set at the conventional site. Three sets of two piezometers each. Each set with piezometers at 0.5 and 1.0 meters below water table. Piezometer set was located at PZ (Figure 9).

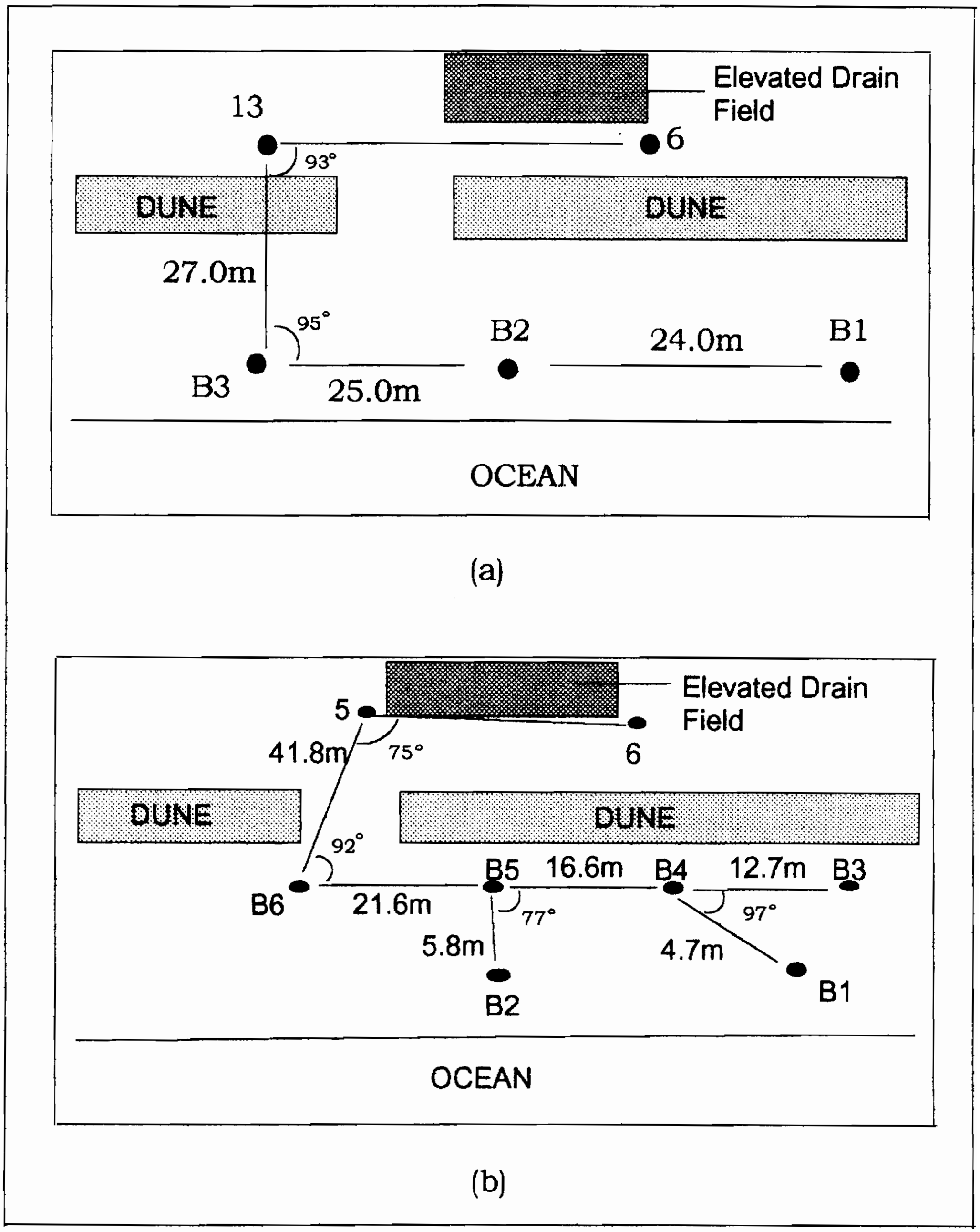



Figure 11. Temporary well transects for the Mound site (a) July; (b) August. The maps are not drawn to scale.

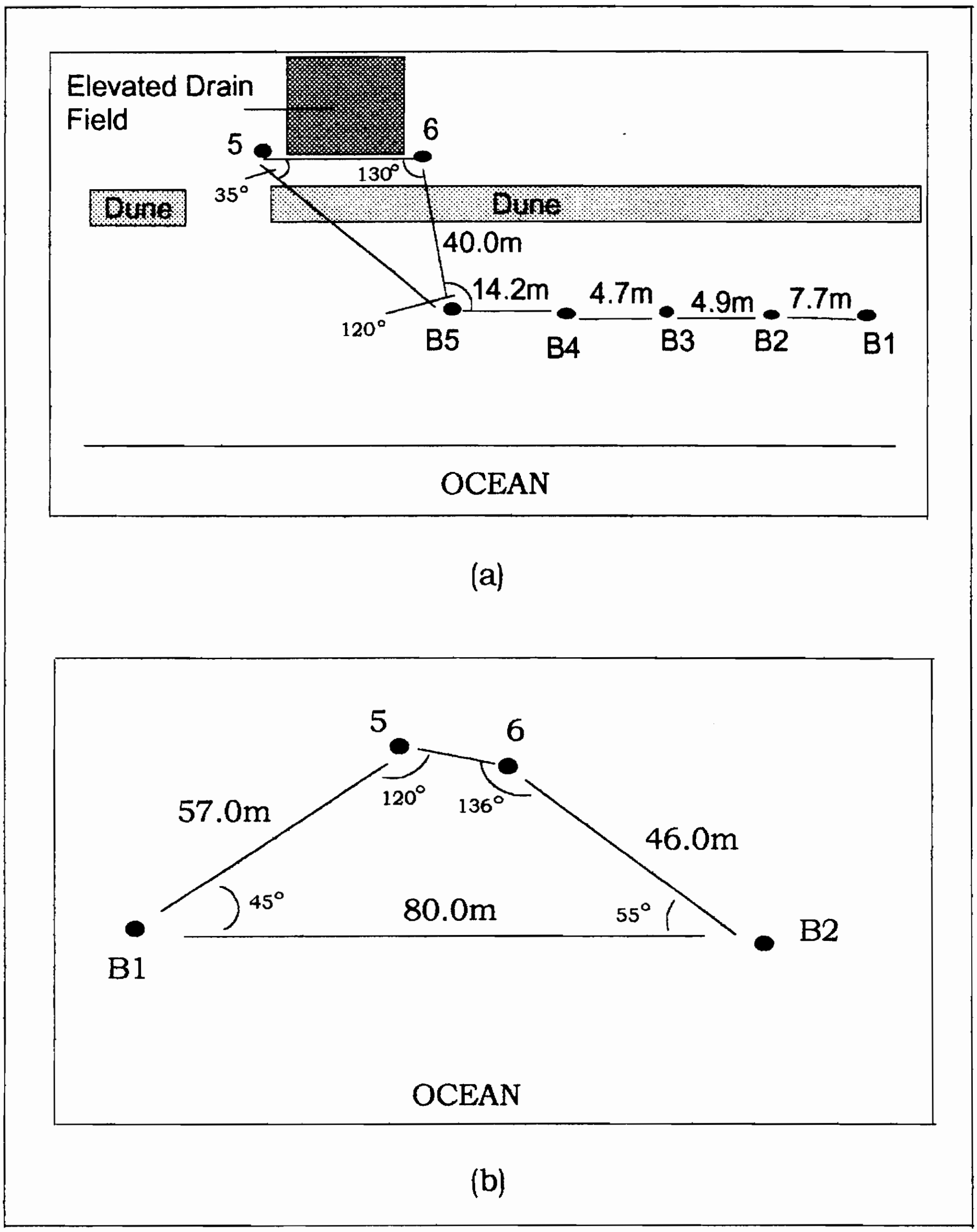



Figure 12. Temporary well transects for the Mound site (a) September; (b) October. The maps are not drawn to scale.

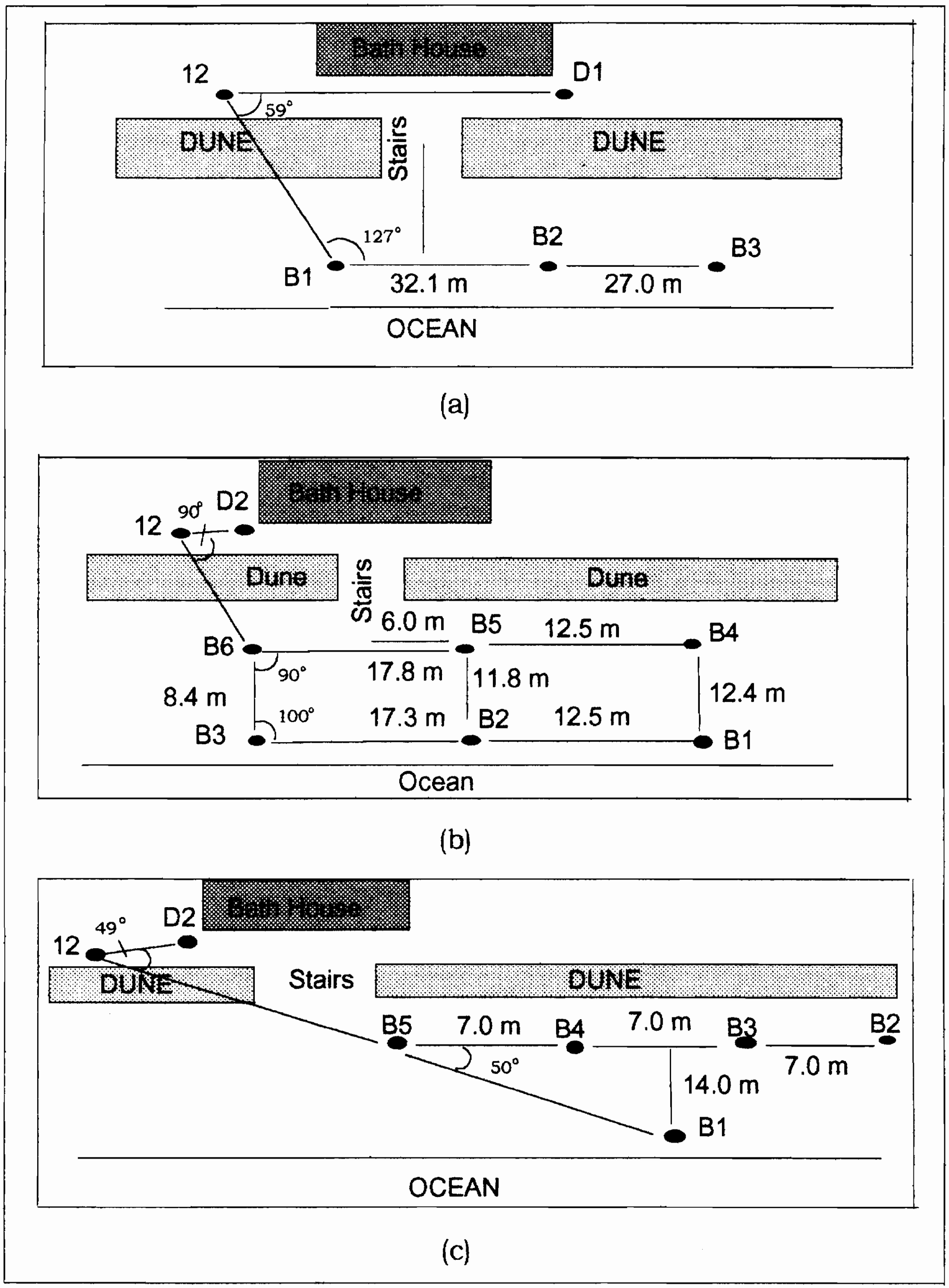



Figure 13. Temporary well transects for the Conventional site (a) July (b) August 5; (c) August 12. Maps are not drawn to scale.

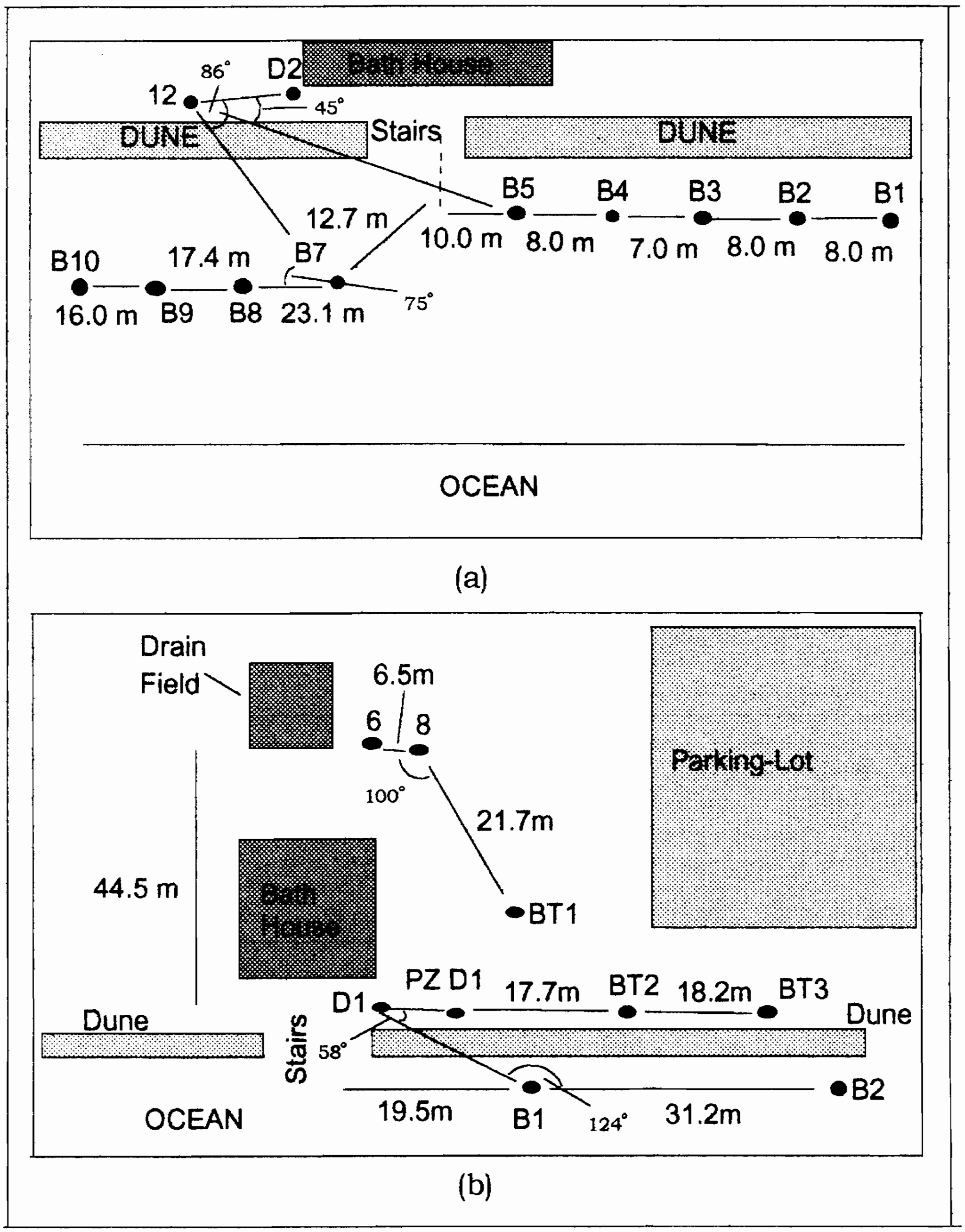
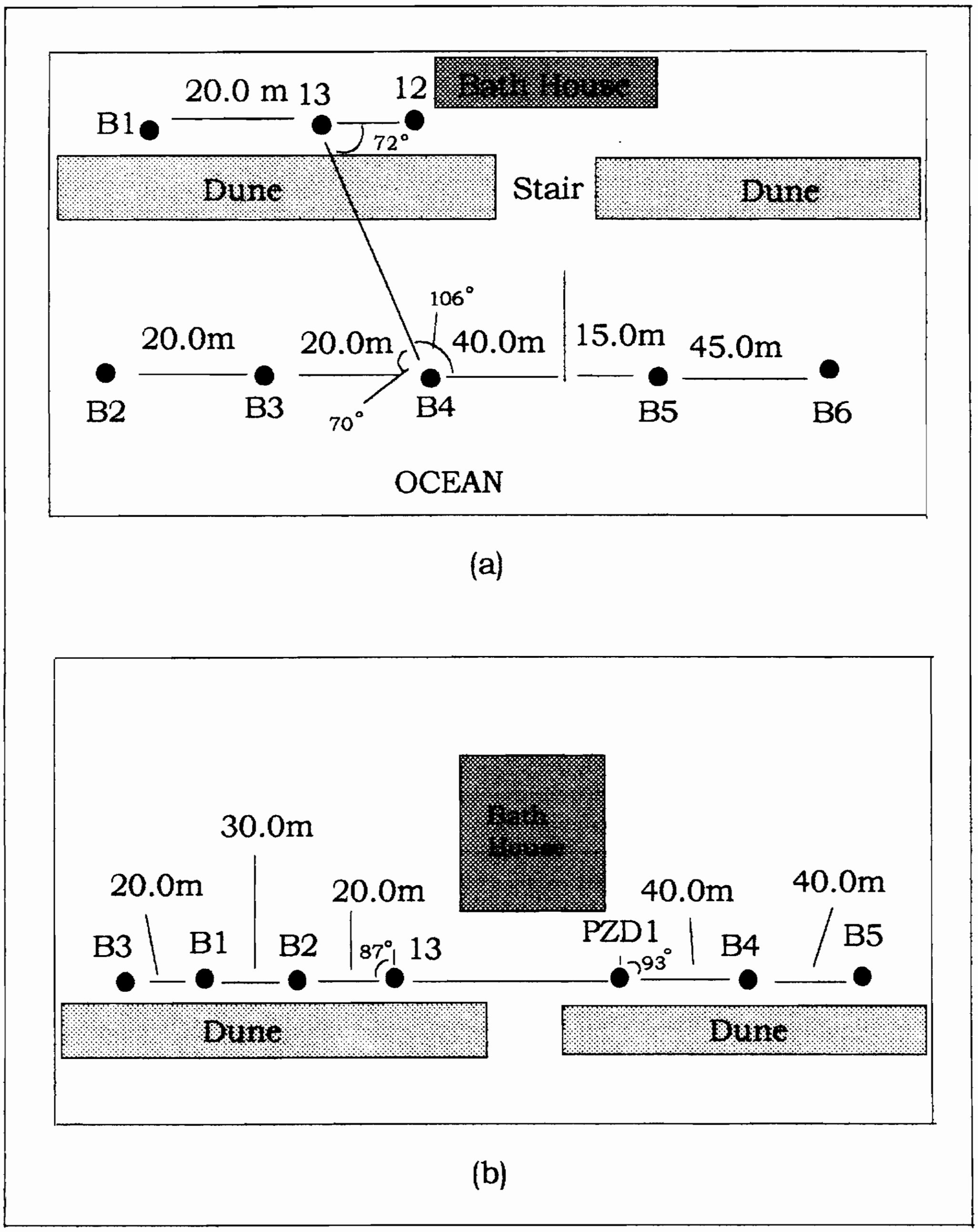



Figure 14. Temporary well transects for the Conventional site
(a) September 16; (b) October 2. Maps are not drawn to scale.




(a)



(b)

Figure 15. Temporary well transects for the Conventional site
(a) October 28; (b) November 19.
The maps are not drawn to scale.

wells 1 through 4, and the proximal half section of the drainfield consisting of wells 5 and 6 (Figure 9).

Water table elevations were determined by an electrical probe (Solinst Water Level Meter). Ground water flow directions were determined using equipotential contours assuming flow to be perpendicular to the contour in the direction of decreasing hydraulic head (Heath, 1983). Given the coastal characteristic of the site, variable density conditions existed within the unconfined aquifer. Simplifying assumptions were made in order to make the utilization of hydraulic head contour maps valid for delineation of ground water flow direction. First, given that the upper confining unit of the Yorktown-Eastover aquifer was approximately 12 meters below sea level, it was assumed that this distance was small for the saltwater interface to penetrate a great distance inland (Sherif et al., 1990). Second, the salt-fresh water interface was assumed to be a well defined boundary. Third, the density variations were considered to occur only in the horizontal direction. With these assumptions, a barotropic field would exist as the surfaces of constant pressure were perpendicular to the surfaces of equal density.

As mentioned in the literature review, the most common approaches to deal with the determination of ground water flow in environments of variable density are: equivalent freshwater head, Hubbert method, Darcy's method, and the Hydrostatic method. All of these methods have disadvantages for their application in this study. The accuracy of the equivalent freshwater head is directly related to the precision of the salt-water level measurement (distance from the datum (upper confining unit) to the measuring point). Hubbert's and the hydrostatic methods present problems with intervening sources or sinks. The accuracy of Darcy's method is dependent on aquifer and fluid parameters.

The equivalent freshwater head method was considered to provide the smallest errors associated with hydraulic head gradient calculations given the site characteristics. Hydraulic gradient contour maps were used to determine gross tidal effects in the ground water flow, and to determine the ground water flow directions at specific times. For each of the contour maps, sensitivity analyses were performed to study variations in hydraulic heads with the salt-water levels.

To estimate ground water travel time and determine potential preferential flow paths, a dye study was conducted at the conventional drainfield site from September through November, 1993. A 20% aqueous solution of Rhodamine A was used as the dye. One and a half liters was placed in the last septic tank of the conventional system. Samples for dye measurement were taken after a minimum of three volumes of water were removed from wells in order to allow recharge with fresh ground water. A peristaltic pump with Nalgene 180 tubing for each well was used for sample collection. Samples were collected in nalgene bottles and stored in the dark until analyzed. Samples were filtered with 0.45 mm membrane filters and read in a Turner Fluorometer (Model 10). Unique features of the fluorometer and QA/QC procedures used are given in Appendix B.

In order to provide insight into surficial upland sediment structure and water transmission characteristics, physical properties were determined on vertical sediment profiles from selected wells. Grain size mass ratios were determined by wet sieving and pipette analysis (Folk, 1980). Sediment organic matter was determined by combusting dried samples at 500°C for five hours followed by reweighing (Dean, 1974). Organic matter was expressed as a percentage weight loss from combustion of the dried sample.

To determine the hydraulic conductivity of sediments and the peat layer, two consecutive undisturbed soil core samples were collected from the upper soil portion of the peat layer close to the drainfields. Vertical hydraulic conductivities were determined using the falling head permeameter and methodology described by Klute (1965).

## 4.3 Results

Surficial (upper 1.5 meters) upland sediments were characteristic of inorganic sandy sediments for both sites. A Mann-Whitney U test (Systat, 1992) showed there were no significant differences in sediment characteristics between the two drainfield sites ($p=0.3$ for sand; $p=0.9$ for silt; $p=0.6$ clay; $N=50$). Grain size analysis for selected wells are presented in Tables A1 and A2 (Appendix A). Sediments were dominated by sand size particles, which represented 97% of the sediment on a dry weight basis. There was a significant difference ($p=0.001$) in percent organic matter between the two sites. The average value for the mound site was 0.5 percent, and 0.2 percent for the conventional site. A discontinuous thin peat layer, on the order of 0.1 to 0.3 meters thick, was located approximately 0.5 to 0.7 meters below ground level at both sites. Percent organic matter for the peat layer was 30.0 percent.

Two consecutive, undisturbed soil cores, each 26 cm in length, were obtained in neighborhood of the mound drainfield for sediment and peat hydraulic conductivity calculations. The peat layer was located at approximately 0.75 meters below ground level. The first core was characterized by inorganic sandy sediments. The second core was also characterized by inorganic sandy sediment, but contained 2 cm of peat. The hydraulic conductivity of the first core measured $10^{-1.66}$cm/s, typical of a sandy soil were hydraulic conductivities range between $10^{-3}$ to $10^{-1}$ cm/s (Fetter, 1994). The hydraulic conductivity of the second core measured

$10^{-4.97}$cm/s, from which, the peat hydraulic conductivity was calculated to be $10^{-6.09}$cm/s. The peat hydraulic conductivity is characteristic of silt, sandy silts, and clayey sands with hydraulic conductivities ranging between $10^{-4}$ to $10^{-6}$ cm/s. Also the peat could behave as clay soils with a hydraulic conductivity ranging between $10^{-6}$ to $10^{-9}$ cm/s (Fetter, 1994). The peat layer acted as a confining unit due to its low hydraulic conductivity. The peat layer was not found immediately adjacent to the conventional drainfield. On the other hand, a peat layer was found in three of the six wells immediately adjacent to the mound drainfield.

In order to evaluate hydraulic gradient fluctuations due to tidal effects, water level elevations were measured hourly for most of a complete tidal cycle during May 1993, at the conventional drainfield site. During the tidal experiment, the bath house was not in operation. Hydraulic gradient contours and general flow patterns are presented in Figures 16 and 17. Contours maps were generated with the Surfer Graphics (1984) program. The Kriging estimation procedure was used to interpolate the values of the surface at any unsampled location. Ground water flow at high tide (9:00 am, Figure 16) was primarily from the east eastsouth (ocean) toward the west (Toms cove). The hydraulic gradient at the northwest of the drainfield was ~0.0013 $m \cdot m^{-1}$, and ~0.001 $m \cdot m^{-1}$ at the southeast of the drainfield, both towards Toms cove. At 12:00 pm (Figure 16), two hours before low tide, the hydraulic gradient at the northwest of the drainfield increased to ~0.002 $m \cdot m^{-1}$, and at the southeast approached zero. One hour after low tide, 15:00 hrs, (Figure 17), ground water flow patterns at the southeast of the drainfield exhibited preferential flow toward the ocean, with a hydraulic gradient of ~0.001 $m \cdot m^{-1}$. Thus, an inversion of flow occurred at the southeast of the drainfield. At the northwest of the drainfield, ground water flow patterns continued to exhibit preferential flow toward Toms cove with

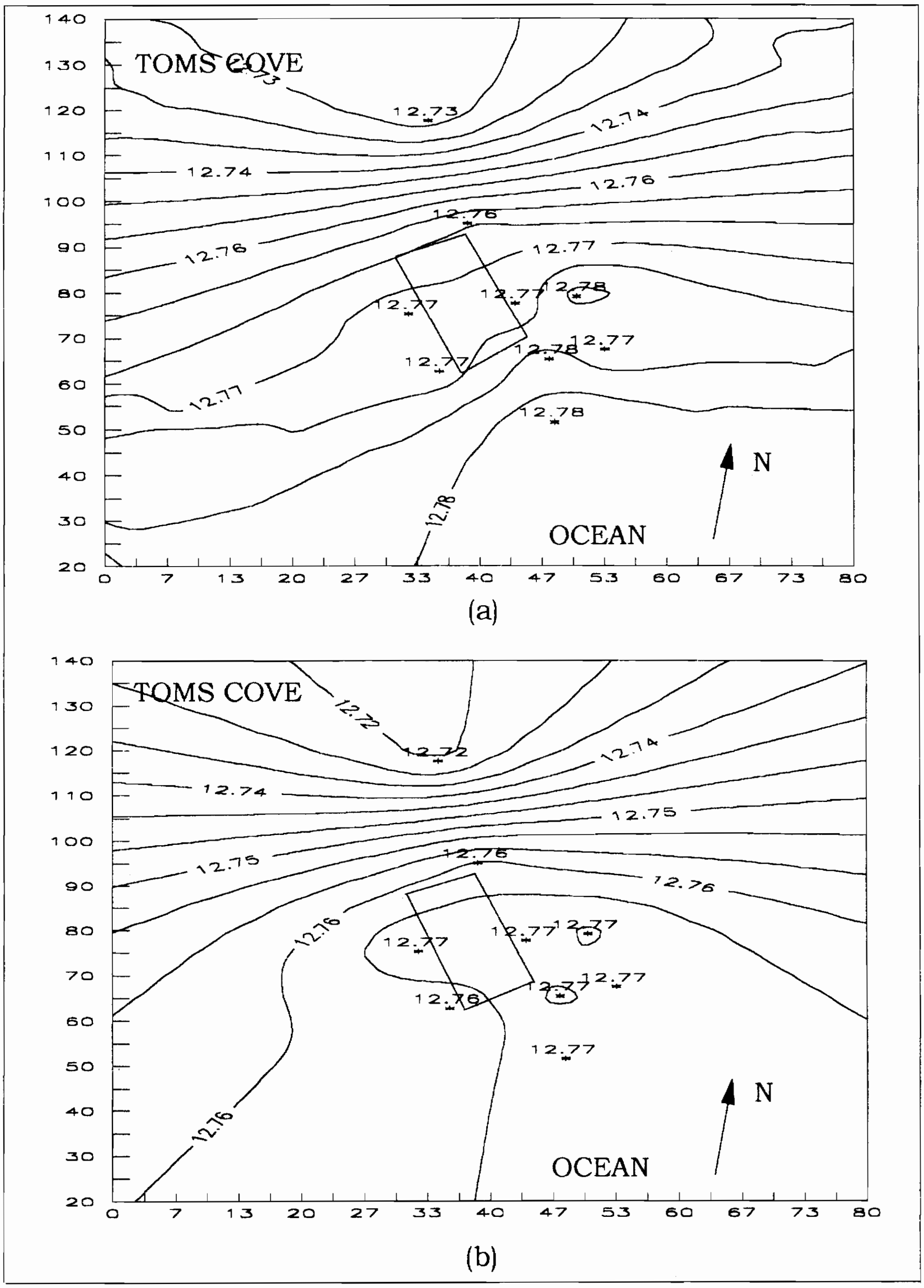



Figure16. Tide Study, (a) 9:00 am; (b) 12:00 pm. High tide 8:00 am; Low tide 14:00 pm. Axes are given in meters. Hydraulic heads are given in meters.May 6, 1993.

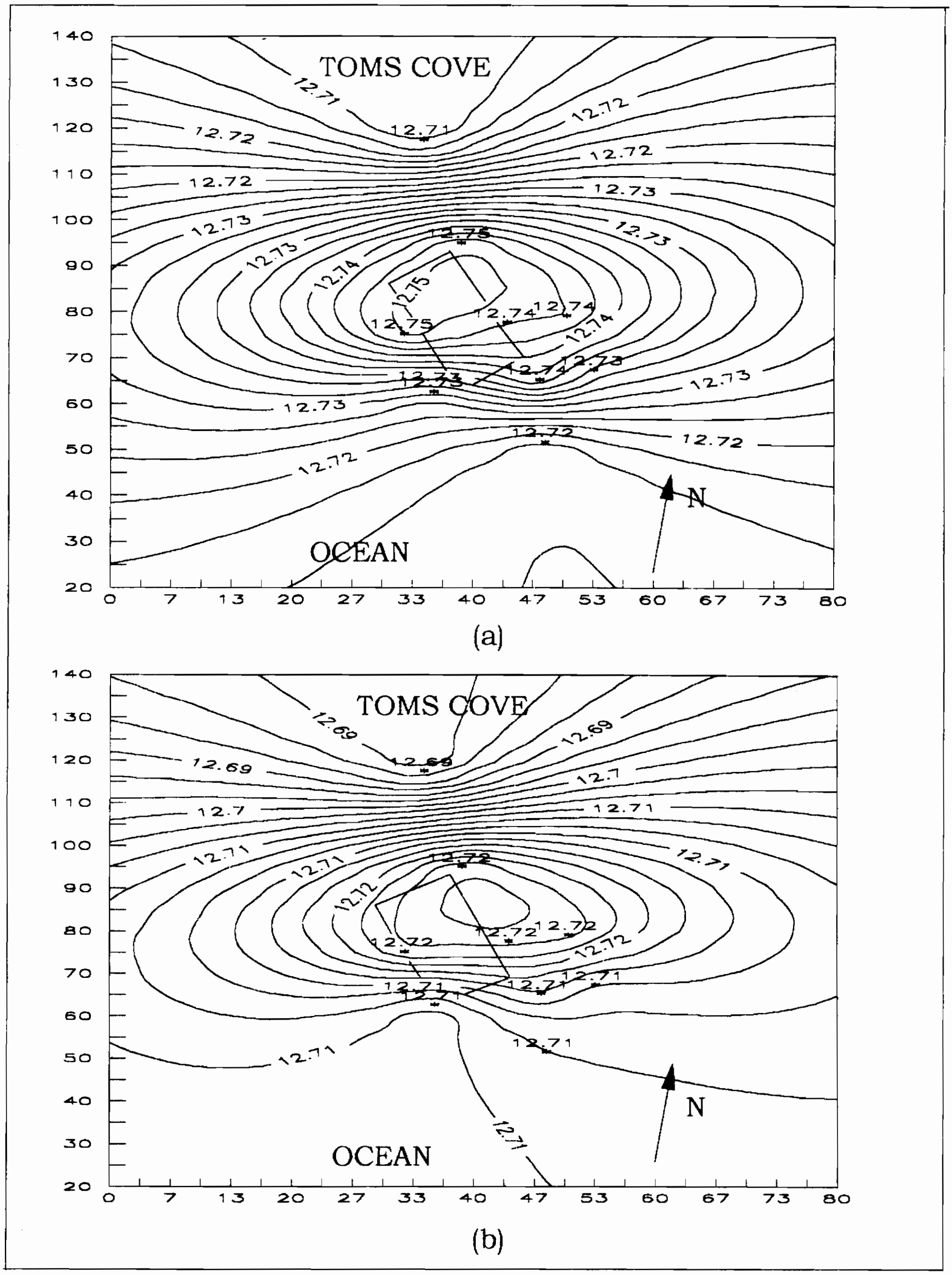



Figure 17. Tide Study, (a) 15:00 pm; (b) 18:00 pm. Low tide 14:00 pm, High tide 20:30 pm. Axes are given in meters. Hydraulic heads are given in meters. May 6, 1993.

no change in hydraulic gradient. Two and a half hours before high tide, (18:00, Figure 17), hydraulic gradient approached zero again, at the southeast of the drainfield. Thus, the point of flow direction reversal was reached. At the northwest, flow patterns continued toward Toms cove, with a hydraulic gradient of ~0.001 $m \cdot m^{-1}$.

The above analysis was based on the equivalent freshwater head method. This requires accurate aquifer depth. Since this value was only known approximately a sensitivity analysis was performed to evaluate flow pattern changes with measured water table elevations. Nine wells were used during the tidal experiment. The maximum error considered possible in the aquifer depth was estimated at one third of its thickness. Considering this variation to be ± 4m in each of the measured water heads, only the hydraulic heads of three wells showed appreciable differences with the computed freshwater head, measuring ±1.1, ±0.2, and ±0.5 cm. A Mann-Whitney U test showed there was no significant difference ($p$=0.894) between hydraulic heads (freshwater heads) used to estimate the contour maps and the hydraulic heads calculated in the sensitivity analysis. In addition, no appreciable differences were found between the contour maps drawn with hydraulic heads obtained form the sensitivity analysis and the contour maps used for the tide study.

In order to evaluate ground water flow patterns once the bath houses were operational, hydraulic gradients were measured during the sampling days. A single set of water-level measurements cannot be used to accurately characterize ground water flow affected by tidal fluctuations (Serfes, 1991). Given the time constraints, there was no alternative other than to measure hydraulic gradients at specific times during the sampling days. Therefore, the hydraulic gradient contours mapped for each sampling day give an indication of the flow patterns only at the time when the sampling was performed.

The hydraulic gradients were measured during bath house operation. During this time, the drainfields acted as a continuous water source to the ground water system. The average daily wastewater input to each of the drainfields was estimated to be 10,800 L. The average loading values were, for the conventional system 65 $L/m^2/day$ (1.60 $gal/ft^2/day$), and for the mound system, 47 $L/m^2/day$ (1.16 $gal/ft^2/day$).

Four different flow patterns for the conventional system were selected and mapped, two during ebb tidal stages (Figure 18), and two during the flood tidal stages (Figure 19). Figure 18 shows ebb flow patterns on June 8 at 15:00. Flow patterns showed ground water movement southeast toward the ocean and northwest toward Toms cove. Ebb tide ground water flow patterns on October 28 (Figure 18) exhibited preferential flow west toward Toms cove. Flood tide hydraulic heads were measured on July 9 at 11:00 am, and on August 5 at 9:30 am (Figure 19). Similar hydraulic gradient contour maps were obtained for the two sampling days. While at the northwest part of the drainfield, preferential ground water flow was toward Toms cove, at the southeast, flow patterns exhibited preferential flow toward the south southwest and southwest. Appreciable differences were observed between flood flow patterns measured during the tidal experiment and during the time of bath house operation.

Similarly to the conventional system, four different flow patterns were selected and mapped for the mound system, three were during flood tide (Figures 20 and 21) and one at ebb tide (Figure 21). Hydraulic heads for Figure 20 were measured during bath house usage. The two contour maps show a reversal of flow at the beach zone. Near the drainfield, ground water flow exhibited flow patterns southeast toward the ocean. In contrast, ground

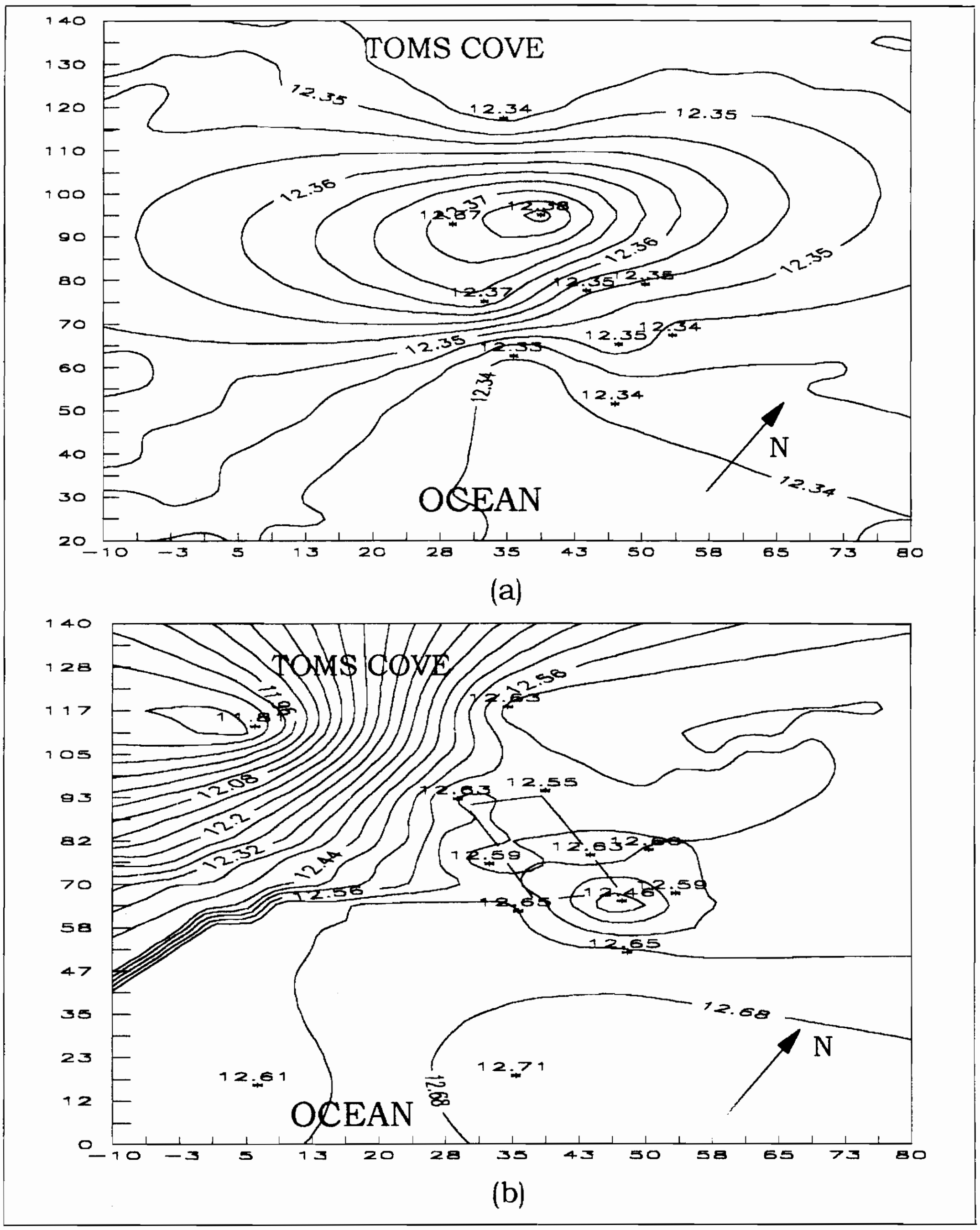



Figure 18. Ebb flow patterns at the conventional site. (a) Flow patterns for June 8. Measured time 15:00. Low tide 17:30. (b) Flow patterns for October 28. Measured time 12:00 pm. Low tide 12:30 pm. Axis scale and hydraulic heads are given in meters. The maps are not drawn to scale.

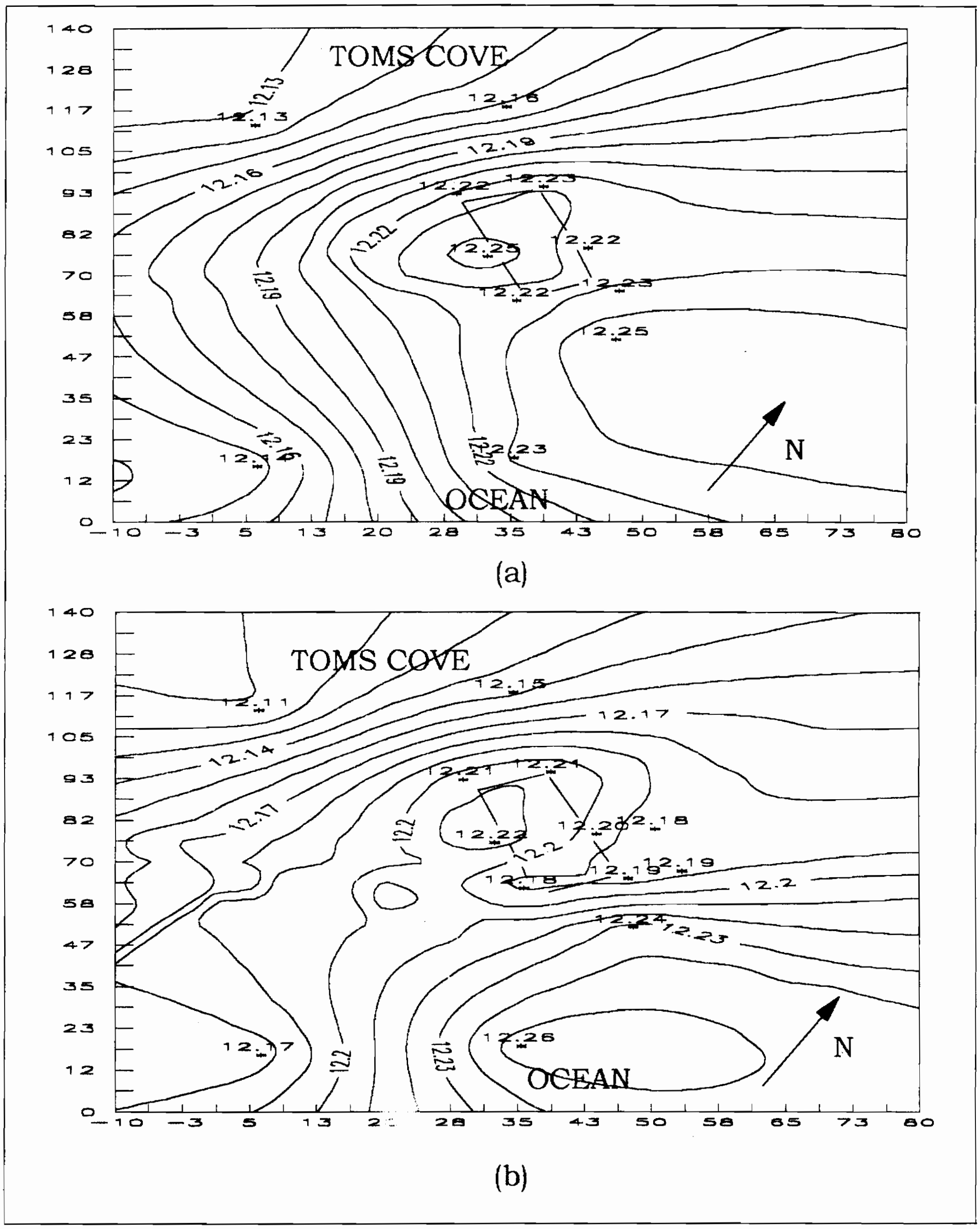



Figure 19. Flood flow patterns at the conventional site. (a) Flow patterns for July 9. Measured time 11:00 am. High tide 11:20 am. (b) Flow patterns for August 5. Measured time 9:30 am. High tide 10:20 am. Axis scale and hydraulic heads are given in meters. The maps are not drawn to scale.

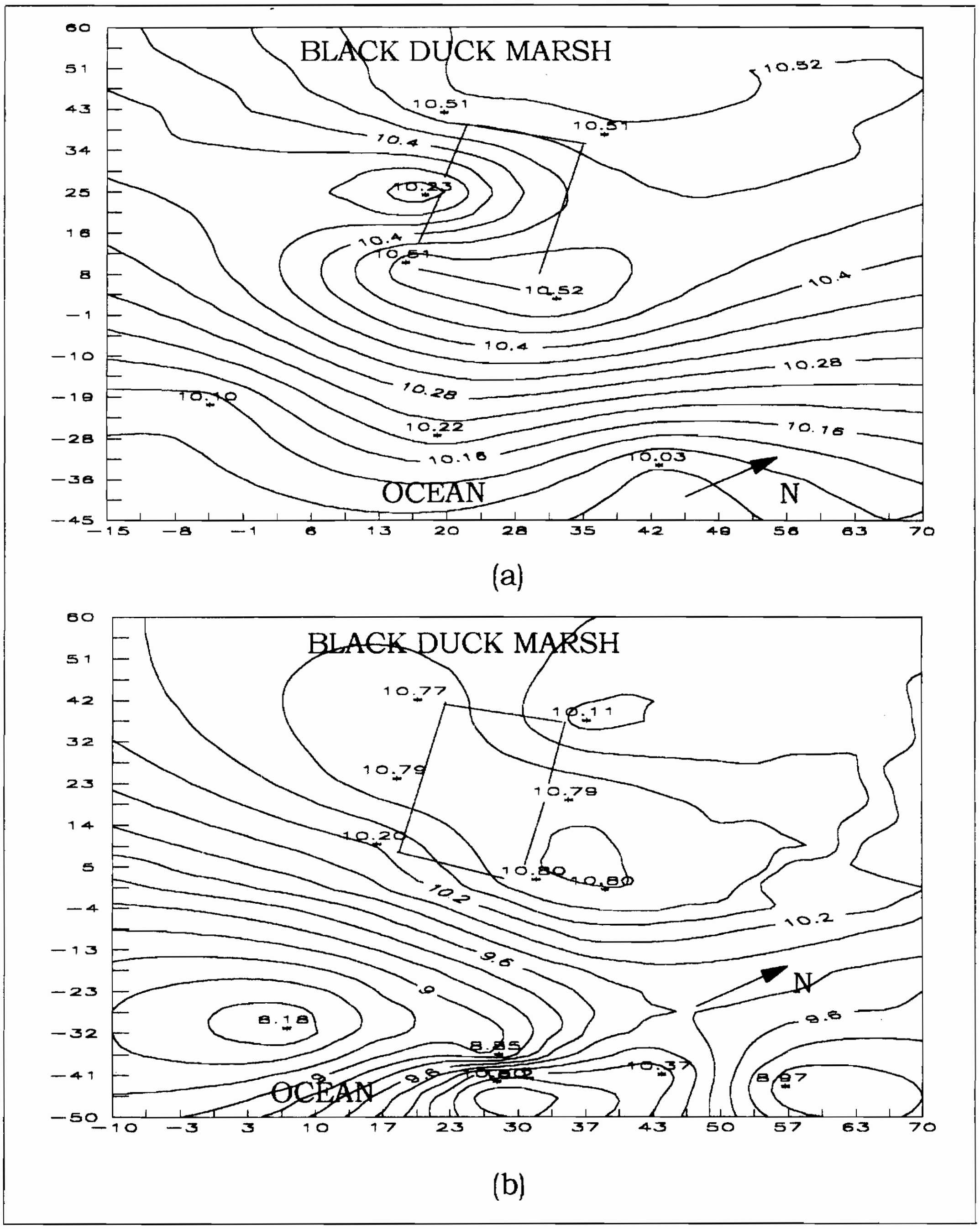



Figure 20. Flood flow patterns at the mound site. (a) Flow patterns for July 9. Measured time 9:00 am. High tide 12:30 pm. (b) Flow patterns for August 5. Measured time 7:30 am. High tide 10:30 am. Axis scale and hydraulic heads are given in meters. The maps are not drawn to scale.

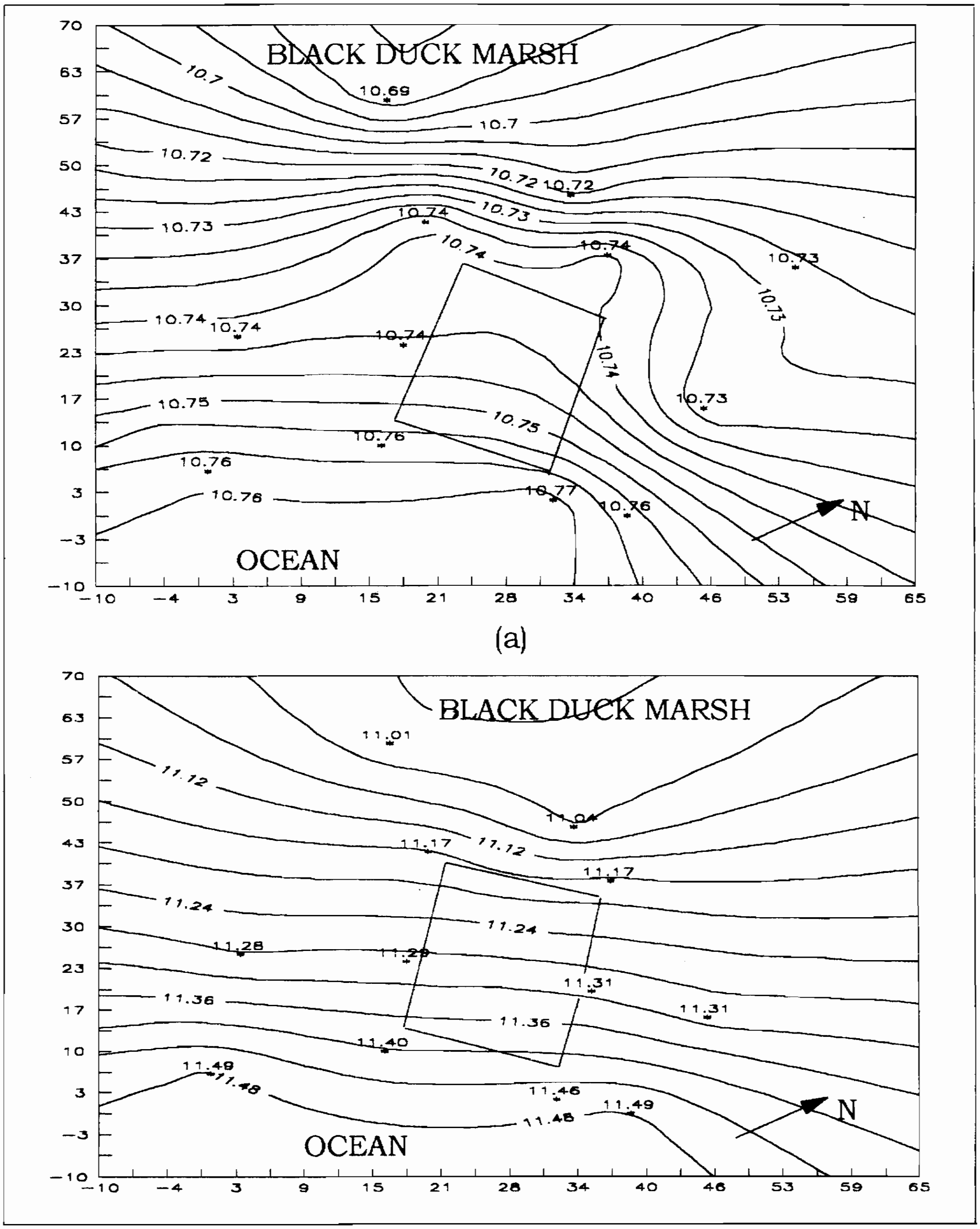



Figure 21. Flow patterns at the mound site. (a) Flood flow patterns for September 9. Measured time 18:30. High tide 20:30. (b) Ebb flow patterns for October 28. Measured time 8:30 am. Low tide 12:30 pm. Axis scale and hydraulic heads are given in meters. The maps are not drawn to scale.

water flow at the beach zone exhibited flow patterns northwest, inland. Figure 21 shows ground water flow patterns during ebb and flood tide stages. Similar flow patterns occurred during these sampling periods with flow patterns exhibiting preferential flow to the northwest toward Black Duck mash.

For each contour map a sensitivity analysis was run, and contour maps with maximum freshwater hydraulic head changes were drawn. Fresh-water hydraulic head changes as high as 10 cm were found at the beach wells. Even though these were large differences, the flow contour maps did not show appreciable differences in ground water flow patterns, mainly because the measurements were taken during flood tide stages.

Hydraulic head gradients were measured on the set of piezometers to determine ground water flow immediately adjacent to the drainfield during September 3. Well 1 was considered to be the reference hydraulic head. Hydraulic gradients from Well 1 to the first set of two piezometers was 0.003 $m \cdot m^{-1}$, for the 0.5 piezometer (PZB 0.5) of the second set the gradient was 0.006 $m \cdot m^{-1}$, and for the 1.0 piezometer (PZB 1.0) the gradient was 0.002 $m \cdot m^{-1}$.

The dye study began on September 3. One and a half liters of a 20% solution of Rhodamine WT was placed in the last septic tank of the conventional system. Dye concentrations for different sampling days are shown in Table A3 (Appendix A). Preferential flows under the conventional drainfield were determined to occur on the distal half of the drainfield. A Mann-Whitney U test showed there was significant differences ($p=0.026$) in the dye concentrations in Wells 1, 2, 3, and 4, and the piezometer set, between September 5 (reference) and the rest of the sampling days. Even though dye concentrations continued to increase in some of the wells and piezometers, there were no significant differences ($p \gg 0.6$) in dye concentrations after September 16. For the rest of the wells and piezometers, there were no significant differences (p=0.958) in dye concentrations between September 5 and the remainder of the sampling days. There were no significant differences (p=0.575) between dye concentrations in the piezometers at 0.5 and 1.0 meters below water table.

## 4.4 Discussion

Upland surficial sediments at both sites (upper 1.5 meters) can be described as inorganic sandy sediments, representative of coastal beach soils. Sand size particles represented 97% of the sediment on a dry weight basis for the two sites. Average percent organic matter at the mound site was 0.5, and 0.2 at the conventional site. These soil characteristics are indicative of moderately high ability to transmit water, and moderate filtration and adsorption abilities.

A soil absorption system must have the capability to absorb and adequately treat the effluent wastewater from the septic tank. For a soil to be suitable for absorption systems it must have an adequate percolation rate for the amount of wastewater loadings. The perlocative capacity for sandy soils is approximately 1.524 meters per hour, or 39.37 min/m (Winneberger, 1984 vol. 1). The percolation rate limit for a soil to be used in an absorption system is 2362 min/m (Winneberger, 1984 vol.1). Thus, the sandy soils of the site are well suited in terms of perlocative capacity for an absorption system.

The Environmental Protection Agency (EPA) tabulated recommended sewage loading rates in terms of soil percolation rates. In order to calculate an estimated value for the percolation rate at the site, the following equation was used (Winneberger, 1984 vol. 1):

$$\log k = -4.76 + 1.55 \log p$$

where k = Darcy's coefficient of permeability (cm/s)

p = percolation rate (in./hr)

Using the hydraulic conductivity measured in the laboratory ($10^{-1.66}$ cm/s), the percolation rate for the soils at both sites was 2.5 m/hr or 24 min/m, characteristic of a coarse to medium sand. The recommended loading rate for this percolation rate is 52.6 L/m$^2$/day (1.29 gal/ft$^2$/day) (Winneberger, 1984 vol. 1). This value is in agreement with the loading rate recommended by Bouma (University of Wisconsin, 1978) of 48.9 L/m$^2$/day (1.2 gal/ft$^2$/day) for sandy soils. Using a conservative approach, the conventional drainfield was on average over-loaded by a factor of 1.33 or 33 percent, and the mound was on average under-loaded by approximately 4 percent.

In order for the soil absorption system to be capable of adequately purifying the wastewater, it has to be able to filter the pathogenic organisms, and have an adequate microenvironment for aerobic biodegradation of the wastewater components. To achieve, or enhance these requirements, the wastewater has to leach through the soil environment a minimum time before reaching the ground water. Distances of 60-90 cm of unsaturated soil are capable of removing nearly all bacteria and viral contamination (University of Wisconsin, 1978; Tyler et al., 1977). Recommended separations between the bottom of the gravel pack and the maximum seasonal water table are 0.6-1.2 m (EPA, 1980), a minimum of 0.9 m (Bouma, 1980), and a minimum of 1.2 m (Canter and Knox, 1985).

Considering the recommended loading rate of 48.9 L/m²/day for sandy soils, and the unsaturated depth criteria set by EPA (1980), and assuming a direct relationship between loading rate and unsaturated depth, the unsaturated depth for the conventional system would have to be between 0.8-1.6 m for the actual average loading rates. Similarly, for the mound system the unsaturated depth would have to be between 0.58-1.15 m for the actual average loading rates. The average of the ten peak loading rates was 122 $L/m^2/day$ (2.99 $gal/ft^2/day$) for the conventional drainfield, and 88 $L/m^2/day$ (2.16 $gal/ft^2/day$) for the mound drainfield. Considering these loading rates for a worst case scenario, the minimum distance would have to be 1.5-3.0 m meters for the conventional drainfield, and 1.1-2.2 meters for the mound drainfield.

Water table elevation was, on average, one meter below ground level, and ranged between 0.8-1.2 m below ground level. Considering a 0.3 m gravel pack, the distance from the bottom of the gravel pack to maximum water table elevation for the conventional drainfield was 0.5 m, and 1.4 m for the mound. Thus, the conventional drainfield failed to provide the required unsaturated zone for the adequate wastewater purification. On the other hand, the mound drainfield provided the adequate unsaturated soil depth to comply with loading rates between average to 65% of a worst case scenario.

There are two basic solute transport mechanisms in ground water: diffusion and advection. Advection is the process by which solutes are transported by the motion of the flowing ground water (Fetter, 1994). Coastal beach soils have a high ability to transmit water, thus advective transport would be the predominant transport mechanisms in sandy soils (Reay et al., 1993). The velocity of flowing ground water can be determined from Darcy's law. Given that the site was influenced by tides, hydraulic gradients fluctuated continuously. In order to accurately determine ground water velocities, hydraulic gradient fluctuations have to be taken into account. The mean hydraulic gradient can be calculated and used to determine the net ground water flow during the tidal cycle (Serfes, 1991). This approach was beyond the scope of this project. Thus, the estimated ground water velocities were only valid for the specific time at which measurements were taken.

If the soil system is considered to be homogeneous and isotropic, the horizontal hydraulic conductivity was $10^{-1.66}$ cm/s at the conventional site. The minimum and maximum hydraulic gradient measured during the tidal experiment toward Toms cove were ~0.0013 $m \cdot m^{-1}$ and ~0.0017 $m \cdot m^{-1}$, respectively. Maximum hydraulic heads toward the southwest (toward the ocean) were ~0.0014 $m \cdot m^{-1}$ measured at ebb tidal stage (15:00). Thus, estimated ground water velocities ranged between 0.025-0.032 m/day toward Toms cove, and 0.000-0.026 m/day toward the ocean. Assuming these velocities were representative of the real net velocities, it would have taken the ground water under the drainfield 72-92 months to reach Toms cove, and 89 months to reach the ocean waters.

Once the bath house was operational, 10,800 L/day were added, on average, to the ground water system at each site. This additional water modified the hydraulic gradients, and the natural ground water flow patterns. For the conventional drainfield, maximum and minimum hydraulic gradients toward Toms cove were ~0.003 $m \cdot m^{-1}$ (July) and ~0.001 $m \cdot m^{-1}$ (June). The maximum hydraulic gradient toward the ocean was ~0.003 $m \cdot m^{-1}$ (June). Thus, ground water velocities would be expected to range between 0.02-0.07 m/day toward Toms cove, and 0.00-0.06 m/day toward the ocean. Assuming these velocities were representative of the real net velocities, with the additional input of water to the ground water system, it would have taken the ground water under the drainfield 35-94 months to reach Toms cove, and at least 40 months to reach the ocean waters. Hence, a reduction of 22-54% in the ground water travel time occured when the bath house was on used based on data from the tide experiment.

During July and August at the mound site, inversion of flow was measured at the beach. The hydraulic gradients near the drainfield toward the ocean water were ~0.012 $m \cdot m^{-1}$. The hydraulic gradients between the uplands and the beach were ~0.013 $m \cdot m^{-1}$ (July) and ~0.064 $m \cdot m^{-1}$ (August). Ground water velocities ranged for these two sampling periods between 0.215-1.183 m/day. If these velocities were representative of the real net ground water velocities, it would have taken the ground water beneath the mound drainfield between 1.3 to 7.7 months to reach ocean waters. September and October hydraulic head contours showed ground water moving inland, toward the Black Duck marsh. Hydraulic gradients were ~0.002 $m \cdot m^{-1}$ (September) and ~0.008 $m \cdot m^{-1}$ (October). Ground water velocities ranged between 0.037-0.148 m/day. Thus, it would have taken the ground water under the mound drainfield 4.4-17.7 months to reach the Black Duck marsh.

It must be emphasized that these ground water velocities can only give an idea of the behavior of the ground water at the time the measurements were taken. In order to accurately determine ground water velocities, pressure transducers or other devices used to measure water table elevations, would have to be employed to measure water table fluctuations over several tidal cycles. In addition, a conductivity meter to estimate water density would have to be set with each pressure transducer to determine fluctuations of densities over time. With these measurements, an average hydraulic gradient could be determined and the net ground water flow patterns established.

Homogeneity of the soil system simplifies the calculations of Darcy velocities. However the soil system is more likely to be heterogeneous. During the past twenty years, the shoreline has, and continues to be altered by ocean erosion. Additionally, the top one to two meters of the soil horizon has been transformed over the years by man and weather. The discontinuous peat layer found approximately 0.75 meters below ground level used to be, some years ago, the soil surface. Furthermore, old bushes exist under the dune and possibly under the soil surface. All of these distinctive characteristics makes the possibility of preferential flows very likely. Thus, under preferential flows, the travel time for the ground water beneath the drainfields to reach surface waters would decrease considerably and possibly be in the range of weeks to several days.

The dye study was conducted to determine the possibility of preferential flows and estimate ground water travel time. The weakness of the study was that it was not started at the beginning of the season and sampling could not be done in a continuous fashion, thus ground water travel times were unable to be determined with this study. Preferential flows under the conventional drainfield were determined to occur on the distal half of the drainfield given that only in this section of the drainfield appreciable differences in dye concentrations, compared with reference values, were measured. Significant differences in dye concentrations were first found almost two weeks after dye addition. All piezometers at 0.5 m below WT (Figure 10) showed high differences in dye concentrations compared to the reference values (between 6 and 18 ppb). Only one piezometer at 1.0 meters below WT (PZ1) had high dye concentration (18 ppb). Thus, anisotropy existed in the soil. Assuming hydraulic gradients between the piezometers measured during September 3 were still valid, Darcy's velocity from Well 1 to the first set of two piezometer (PZA) was

0.057 m/day. Thus, it would have taken 17 days for the dye to travel from Well 1 to PZA 0.5, a distance of 1 meter. Considering this ground water velocity it would have taken the ground water under the conventional drainfield 47 months to reach Toms Cove, and around 6 month to reach the ocean.

In summary, two main flow patterns were determined to exist, at the conventional site, before and during drainfield operation. One flow pattern was toward Toms Cove, and the other flow pattern was toward the ocean. Ground water velocities toward Toms Cove increased from 0.025-0.032 m/day before the season to 0.025-0.066 m/day during the season. Likewise, ground water velocities toward the ocean increased from 0.000-0.026 m/day to 0.000-0.060 mg/day. The mound site showed two main flow patterns during bath house operation. One of the flow patterns was from the drainfield toward the ocean, with ground water velocities of 0.215-1.183 m/day. The second ground water flow pattern was toward the Black Duck marsh, with velocities ranging between 0.037-0.148 m/day. The dye study established the existence of preferential wastewater flows under the conventional drainfield, primarily, at the distal section of the drainfield under wells 1 to 3.

# 5.0 PHYSICAL AND CHEMICAL CHARACTERISTICS OF GROUND WATER

## 5.1 Introduction

On-site wastewater disposal systems (OSWDS) are potential sources of fecal and nutrient contamination to ground and surface waters. The study of physical and chemical characteristics of ground water can indicate the possibility of fecal and nutrient contamination. Chemical characteristics analyzed for ground and surface waters were ammonium, nitrate, nitrite, dissolved inorganic phosphorus, dissolved oxygen, pH, salinity. The physical characteristic measured was temperature.

## 5.2 Methodology

Samples for physical and chemical analyses were taken after a minimum of three volumes of water were removed from wells in order to allow recharge with fresh ground water. A peristaltic pump with sterilized Nalgene 180 tubing for each well was used for sample collection. Sampling collection was performed monthly or bimonthly. Single samples were collected per well with a 10% field replicates. Samples were stored on ice and returned to the Virginia Tech Coastal Groundwater Research Laboratory (Eastern Shore Station). Water samples for inorganic nutrient analysis were filtered with a pre-rinsed 0.45 μm membrane filter. The analytical chemistry, except for nitrate, began within eight hours of collection. At least 10% of the samples were duplicated or triplicated in the laboratory. At least 10% of the samples were spiked. Nitrate samples, if not analyzed during the same period, were frozen and analyzed within two days of collection. Ammonium ($NH_4^+$) was determined by a modified phenate method (Stickland and Parsons, 1972). Nitrate ($NO_3^-$) was determined by the cadmium reduction technique (U.S.E.P.A., 1983; Method 353.3). Nitrite ($NO_2^-$) was determined by the diazonation of sulfanilamide and coupling with N-(1-naphthyl)-ethylenediamine dihydrochloride to form an azo dye (U.S.E.P.A., 1983; Method 354.1). Dissolved inorganic nitrogen (DIN) was calculated as the sum of nitrate, nitrite, and ammonium. Dissolved inorganic phosphorus (DIP) was determined by the single reagent ascorbic acid method (U.S.E.P.A., 1983; Method 365.2). An azide modification of the standard iodometric method was used to determine dissolved oxygen levels (APHA, 1989; Method 4500-0). The pH of water samples was determined with a Fisher Scientific Accumet 910 pH meter and microprobe combination electrode. Salinity was determined by a Beckman Industrial induction salinometer (Model RS-10). Temperature was determined by a calibrated digital or field thermometer.

Quality assurance guidelines were provided by procedures outlined by Standard Operating Procedure (SOP) for the Coastal Ground Water Research Program (Miles et al., 1992). The quality assurance include guidelines for sample collection and handling, sample preservation, control chart analyses, and safety considerations.

## 5.3 Results

### 5.3.1 Conventional OSWDS

Ground water and surface water chemical and physical characteristics by sampling period for upland wells are shown in Tables A4 to A11, for the beach wells in Tables A15 to A16, and for the piezometer set in Table A13, Appendix A.

In order to facilitate the presentation of the results and discussions, a certain notation was used to distinguished the upland wells. Wells immediately adjacent to the drainfield were referred to as "first tier wells". The wells not immediately adjacent to the drainfield were designated as "second tier wells". The term "upland wells" was used to refer to both, first and second tier wells. The first tier wells were further subdivided into two sections. The "distal half section" of the drainfield consisting of wells 1 through 4, and the "proximal half section" of the drainfield consisting of wells 5 and 6.

Ground water quality characteristics measured during January and March were considered as reference values because they were collected prior to the start of the season. Temporal and spatial variations were observed for DIN species, both in terms of quantitative amounts and relative percent of each species. Once the bath house was operational, DIN concentrations increased in the distal half section of the drainfield. First tier wells, wells 1, 2, 3, and 4 (Figure 9), showed an increase of two orders of magnitude in DIN concentrations compared with reference values. Average dissolved inorganic nitrogen concentrations were 0.76 mg/L ($\sigma$=0.42 mg/L; N=4) for January and increased to 80.78 mg/L ($\sigma$=32.33 mg/L; N=4) by August. The DIN concentrations were, on average, three orders of magnitude higher than the average DIN of the ocean waters, which measured 0.05 mg/L ($\sigma$=0.03 mg/L; N=9). During the same period, ammonium values increased from 0.02 to 60 mg/L, nitrate from 1.10 to 24 mg/L, and nitrite values from below detection limits to 2.4 mg/L. Temporal DIN variations with relative percent of each species are shown in Figure 22.

Average ground water ammonium and nitrate concentrations, and statistical comparisons between the two for the distal half section of the drainfield are shown in Table 9. Once the bath house was operational, the ammonium and nitrate concentrations at the distal half section of the drainfield increased. Mann-Whitney U tests showed these increments were significant differencet ($p<0.005$; N=24; the symbol < denotes that all the test had p values less than 0.005) from reference values. Mann-Whitney U tests showed there were no significant differences ($p>0.916$; N=32) in ammonium and nitrate concentrations between the values measured during bath operation and after fall closed down.

The second tier wells did not show, in general, a consistent variation with respect to DIN concentration during the study. Significant differences ($p\leq0.001$; N=50) were found between ammonium and nitrate concentrations in the second tier wells. Nitrate was the major inorganic nitrogen species and represented approximately 90% of the DIN. During June-August average nitrate was 1.38 mg/L ($\sigma$=1.24 mg/L; N=21), and increased to 2.77 mg/L ($\sigma$=5.16 mg/L; N=14) during October-November. Even though this was an appreciable change in the average nitrate concentrations, no significant differences ($p=0.173$; N=35) were found between the two periods. The maximum DIN concentrations in the second tier wells were measured in Well 10 (7.02 mg/L) during October and in Well 12 (19.27 mg/L) during November. During, and after the

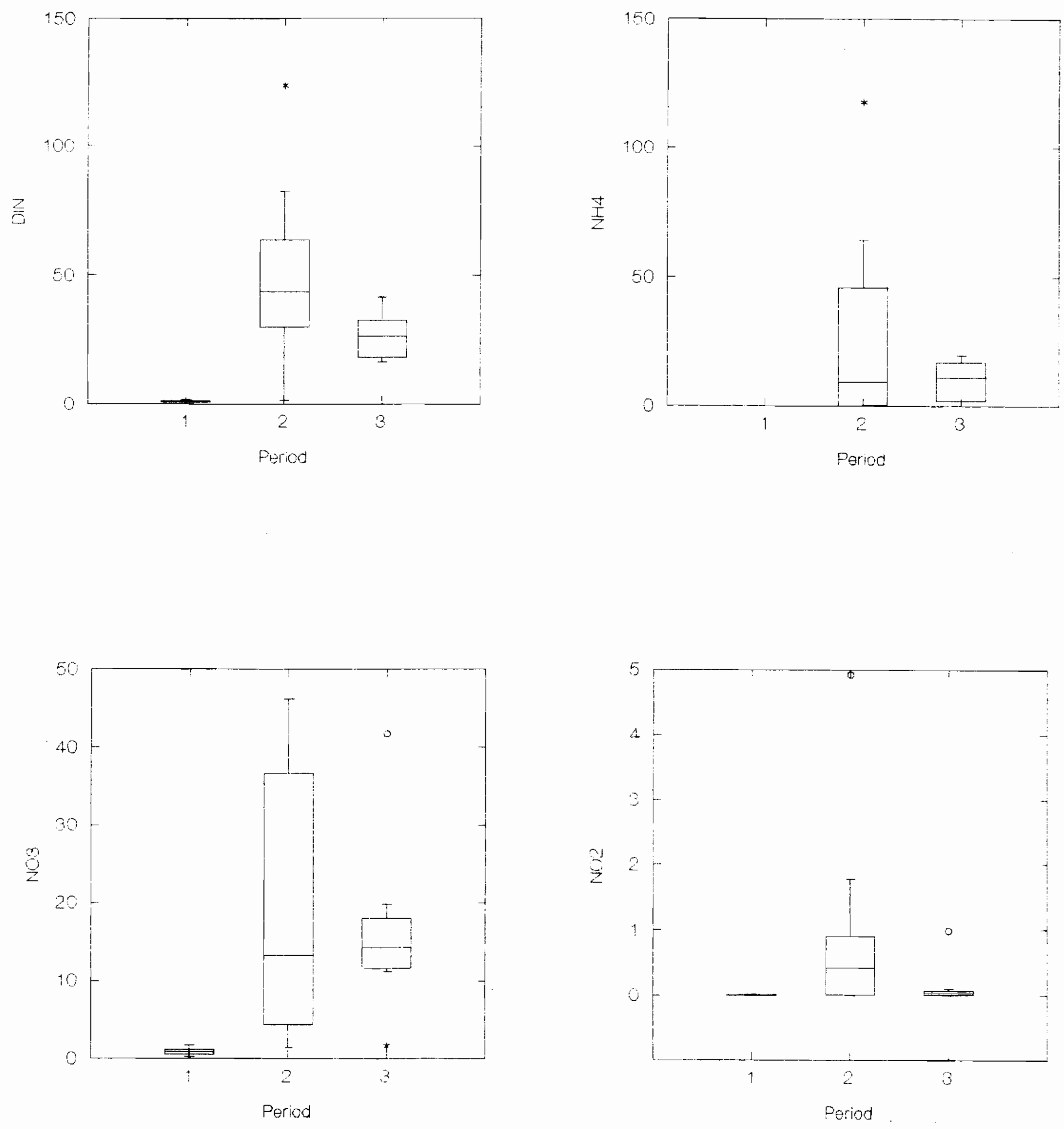



Figure 22. Temporal variation in dissolved inorganic nitrogen (DIN) and inorganic nitrogen species at the distal section of the conventional drainfield (Wells 1-4). Period 1, 2 and 3 represent before, during, and after the season, respectively. Concentrations are given in mg per liter Number of samples used for period 1, N=8; period 2, N=16; period 3, N= 12.

Table 9. Average ground water ammonium and nitrate concentrations, and statistical comparisons between the two for the distal half section of the conventional drainfield. Mean concentrations and standard deviations are given in mg/L. Statistical comparisons were made using a Mann-Whitney U Test.

| | | P values | | Mean value | Standard Deviation |
|---|---|---|---|---|---|
| | | $NH_4$ | $NO_3$ | | |
| **Prior to season** (N=8) | $NH_4$ | - | 0.001 | 0.06 | 0.08 |
| | $NO_3$ | 0.001 | - | 0.88 | 0.48 |
| **During season** (N=16) | $NH_4$ | - | 0.773 | 27.29 | 36.36 |
| | $NO_3$ | 0.773 | - | 19.54 | 17.24 |
| **After fall close down** | $NH_4$ | - | 0.368 | 9.83 | 7.77 |
| (N=8) | $NO_3$ | 0.368 | - | 16.42 | 11.57 |

bath house was operational, significant differences ($p<0.003$; N=40) were found in ammonium and nitrate concentrations between the first and the second tier wells.

Dissolved oxygen (DO) concentrations decreased in all the upland wells once the bath house was operational (Table 10). The comparison of DO concentrations between the first and second tier wells showed significant differences ($p=0.022$; N=24) only during June-August.

Average dissolved inorganic nitrogen in the beach wells was 1.03 mg/L ($\sigma$=0.87 mg/L). The nitrate and ammonium were the major DIN species, and represented, on average, 60 and 40 percent, respectively. Nitrite levels were below detection limits. Even though these percent differences between nitrate and ammonium, no significant differences ($p=0.118$; N=36) were found between the two concentrations during the study. Appreciable differences were measured in DIN concentrations between some of the beach wells. During the sampling days, it was observed that those beach wells which had the highest DIN concentration also had the lowest salinity values. A Spearman rank correlation coefficient ($r_S$= -0.506) (Systat, 1992) was used to test the possible correlation between DIN and salinity in the beach wells. Rejection of the no correlation hypothesis was at $p<0.001$. Only during October-November were the beach wells significantly different ($p=0.030$; N=29) in nitrate concentrations when compared with the upland wells.

In the area immediately adjacent to the drainfield where the piezometer set was located, no significant differences were measured in the DIN concentrations between 1.0 and 0.5 m below water table (WT) during the study. Average DIN and nitrogen species concentrations are given in Table 11. During August to October ammonium was the major inorganic nitrogen species at 0.5 and 1.0 m below WT. No significant

Table 10. Dissolved oxygen concentration at the conventional upland wells. Values are given in mg/L.

| | Mean Value | Standard Deviation |
|---|---|---|
| **Distal section of the drainfield** | | |
| **Prior to season** (N=8) | 7.6 | 2.7 |
| **During season** (N=14) | 1.7 | 1.7 |
| **After fall close down** (N=8) | 1.8 | 1.7 |
| **Proximal section of the drainfield** | | |
| **Prior to season** (N=4) | 10.7 | 0.6 |
| **During season** (N=5) | 7.2 | 4.0 |
| **After fall close down** (N=4) | 4.2 | 2.2 |
| **Second tier wells** | | |
| **Prior to season** (N=4) | 7.9 | 3.2 |
| **During season** (N=18) | 4.8 | 2.6 |
| **After fall close down**(N=13) | 3.0 | 2.4 |

Table 11. Ammonium, nitrate, and DIN concentrations at 0.5 and 1.0 meters below water table at the piezometer set immediately adjacent to the conventional drainfield. Mean values and standard deviations (shown in parenthesis) are given in mg/L.

| | **0.5 m below WT** | **1.0 m below WT** |
|---|---|---|
| **August to October** | | |
| **DIN** (N=18) | 79.14 (29.95) | 98.23 (11.17) |
| **$NH_4$** (N=18) | 74.25 (35.58) | 97.97 (11.34) |
| **$NO_3$** (N=18) | 4.81 (6.90) | 0.28 (0.68) |
| **October and November** | | |
| **DIN** (N=12) | 21.87 (12.48) | 22.60 (7.74) |
| **$NH_4$** (N=12) | 5.83 (6.83) | 12.10 (10.99) |
| **$NO_3$** (N=12) | 15.36 (8.58) | 9.12 (9.17) |

differences in ammonium (p=0.268; N=18) and nitrate (p=0.459; N=18) concentrations were measured between 0.5 and 1.0 m below WT. During October and November nitrate was the major inorganic nitrogen species at 0.5 m below WT. At 1.0 m below WT, ammonium and nitrate represented approximately the same percent of the inorganic nitrogen species. No significant differences were measured between 0.5 and 1.0 m below WT in ammonium (p=0.423; N=12) and nitrate (p=0.150; N=12) concentrations.

The collection of DO at the piezometer set was restricted by problems with air and dissolved gases introduced during pumping. On average, DO was higher at 0.5 m below WT than at 1.0 m below WT. Anaerobic conditions were only measured in Well 1 and at 1.0 m below the WT at PZA and PZB (Figure 10). Low dissolved oxygen conditions in the piezometers were related to high ammonium and very low nitrate concentrations.

Temporal and spatial variations were observed for dissolved inorganic phosphorus (DIP) at the distal half section of the drainfield (Table 12). Wells 5 and 6 did not show any appreciable differences in DIP concentrations during the study. Their average DIP concentration was 0.05 mg/L ($\sigma$=0.02 mg/L; N=16). The only second tier well which showed DIP concentrations higher than 1 mg/L was well D3, located approximately four meters from wells 1 and 2. In the second tier wells, as well as, the beach wells, no significant DIP differences (p=0.801) were measured between the sampling periods. In addition, no significance differences (p=0.299; N=70) were measured between the upland wells and the beach wells. During the study, the average DIP concentration was 0.07 mg/L ($\sigma$=0.05 mg/L; N=40) for the second tier wells, and 0.09 mg/L ($\sigma$=0.08 mg/L; N=30) for the beach wells. The average DIP

Table 12. Dissolved inorganic phosphorus for the distal half section of the drainfield and for the piezometer at the conventional site. Values are given in mg/L. A Mann-Whitney U test was used to calculate the P values. Notation for p values: prior to season (1), during season (2), and after fall close down (3).

| | Mean | STD | P values | | |
|---|---|---|---|---|---|
| **Distal section** | | | 1 | 2 | 3 |
| **Prior to season** | 1.20 | 1.37 | - | 0.034 (N=32) | 0.143 (N=16) |
| **During season** | 7.36 | 6.60 | 0.034 | - | |
| **After fall close down** | 1.86 | 1.17 | 0.143 | | - |
| **Piezometer set** | | | | | |
| **August to October** | | | | | |
| **0.5** | 23.04 | 6.84 | | | |
| **1.0** | 17.48 | 8.60 | | | |
| **October and November** | | | | | |
| **0.5** | 0.74 | 0.51 | | | |
| **1.0** | 0.53 | 0.23 | | | |

concentration in the ocean waters was 0.01 mg/L ($\sigma$=0.02 mg/L; N=10). The beach wells and the second tier wells showed significance differences (p=0.001; N=104) in DIP concentrations with the wells in the distal section of the drainfield during the study.

In the piezometer set, no significant DIP differences (p>0.211; N=30) were measured between 0.5 and 1.0 m below WT from August to November. Dissolved inorganic phosphorus and DIN concentrations at 0.5 and 1.0 meters below WT are shown in Figure 23.

Ground water pH did not show temporal or spatial variations. The average upland ground water pH was 7.09 (range=6.36-8.31), the average beach ground water pH was 7.26 (range=6.95-7.62), and the average ocean pH was 8.07 (Range=7.57-8.13). Ground water temperature showed seasonal but not spatial variation. Measurements showed a minimum during January-March of 8°C and a maximum during August of 23°C.

Temporal and spatial variations were observed for salinity at the distal half section of the drainfield. No appreciable salinity differences were observed at the second tier wells during the study. Average salinity values are shown in Table 13.

### 5.3.2 Mound OSWDS

Ground water and surface water chemical and physical characteristics for the upland wells at the mound site are shown in Tables A17 to A20, and for the beach well in Table A22, in Appendix A.

In order to facilitate the presentation of the results and discussions a certain notation was used to distinguished the upland wells. Wells immediately adjacent to the drainfield were referred to as "first tier

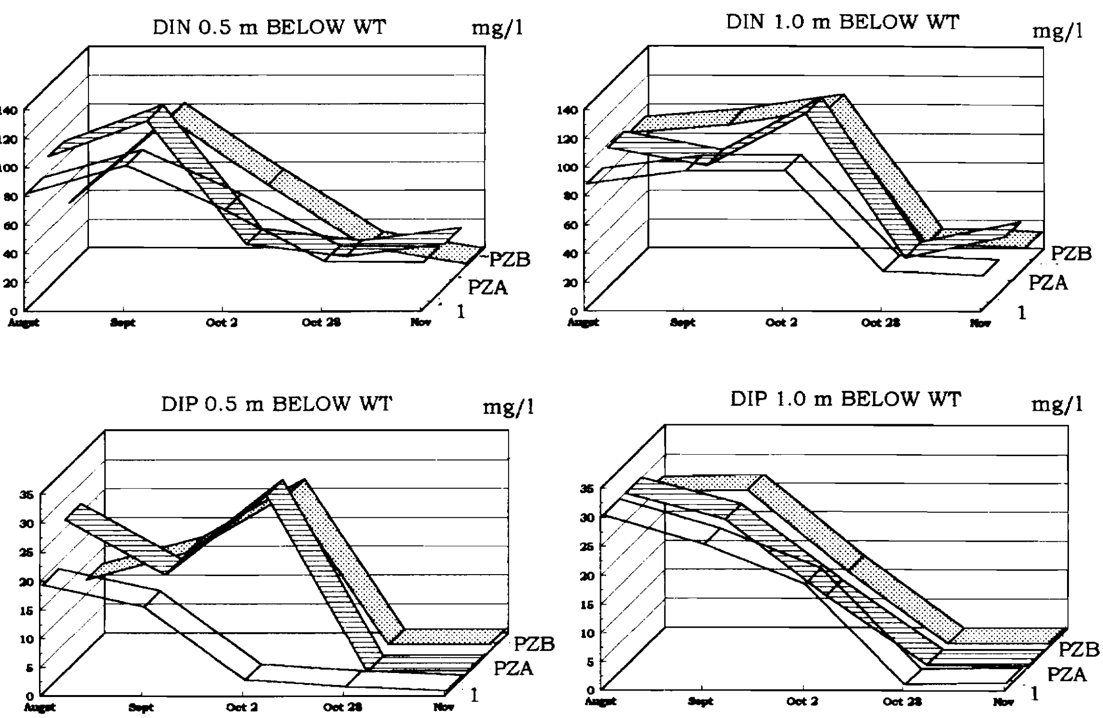



Figure 23. Dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) by sampling date for the piezometer set (Figure 10) at the conventional drainfield. Concentrations are expressed in mg per liter.

Table 13. Salinity values at the conventional site. Salinity is given in parts per thousand. Standard deviation is noted as STD.

| | | Mean | STD |
|---|---|---|---|
| **Distal half section** | | | |
| **Prior to season** | (N=8) | 0.232 | 0.145 |
| **During season** | (N=15) | 3.116 | 0.159 |
| **After fall closed down** | (N=8) | 3.247 | 1.282 |
| **Proximal half section** | | | |
| **Prior to season** | (N=4) | 0.166 | 0.033 |
| **During season** | (N=8) | 0.115 | 0.030 |
| **After fall closed down** | (N=4) | 0.165 | 0.052 |
| | | | |
| **Second tier wells** | (N=32) | 0.651 | 0.274 |
| **Water used in the bath house** | | 4.072 | 0.075 |
| **Beach wells** | (N=30) | 23.210 | 8.643 |
| **Ocean waters** | (N=10) | 31.354 | 0.426 |

wells". Wells not immediately adjacent to the drainfield were designated as the "second tier wells". The term "upland wells" was used to refer to both, first and second tier wells.

Ground water quality characteristics measured during May were considered as reference values. Temporal and spatial variations were observed for DIN species, both in terms of quantitative amounts and relative percent of each species. Concentrations of DIN in the drainfield region were 1.01 mg/L ($\sigma$=0.81 mg/L; N=6) for May and increased to 39.95 mg/L ($\sigma$=14.64 mg/L; N=6) by September. The DIN concentrations were on the range of three orders of magnitude higher than average DIN ocean waters concentration (0.05 mg/L ($\sigma$=0.03 mg/L; N=6)). Temporal DIN variations with relative percent of each species are shown in Figure 24.

The second tier wells showed maximum DIN values during October and November, with an average value of 8.77 mg/L ($\sigma$=15.02 mg/L; N=14). Ammonium represented approximately 90% of the inorganic nitrogen. No significant differences (p=0.438; N=23) were found between DIN and ammonium concentrations for these wells during the study. Ammonium and nitrate concentrations for the upland wells are shown in Table 14.

Reference DO concentrations were between 0.3 and 0.6 mg/L. Once the mound was operational, the first tier wells did not show a consistent pattern in DO variation. For June, average DO concentration were 0.4 mg/L ($\sigma$=0.4 mg/L; N=6), increased to 2.1 mg/L ($\sigma$=1.9 mg/L; N=6) during August, and decreased to 0.3 mg/L ($\sigma$=0.4 mg/L; N=6) during October. General anoxic conditions were found in the rest the second tier wells.

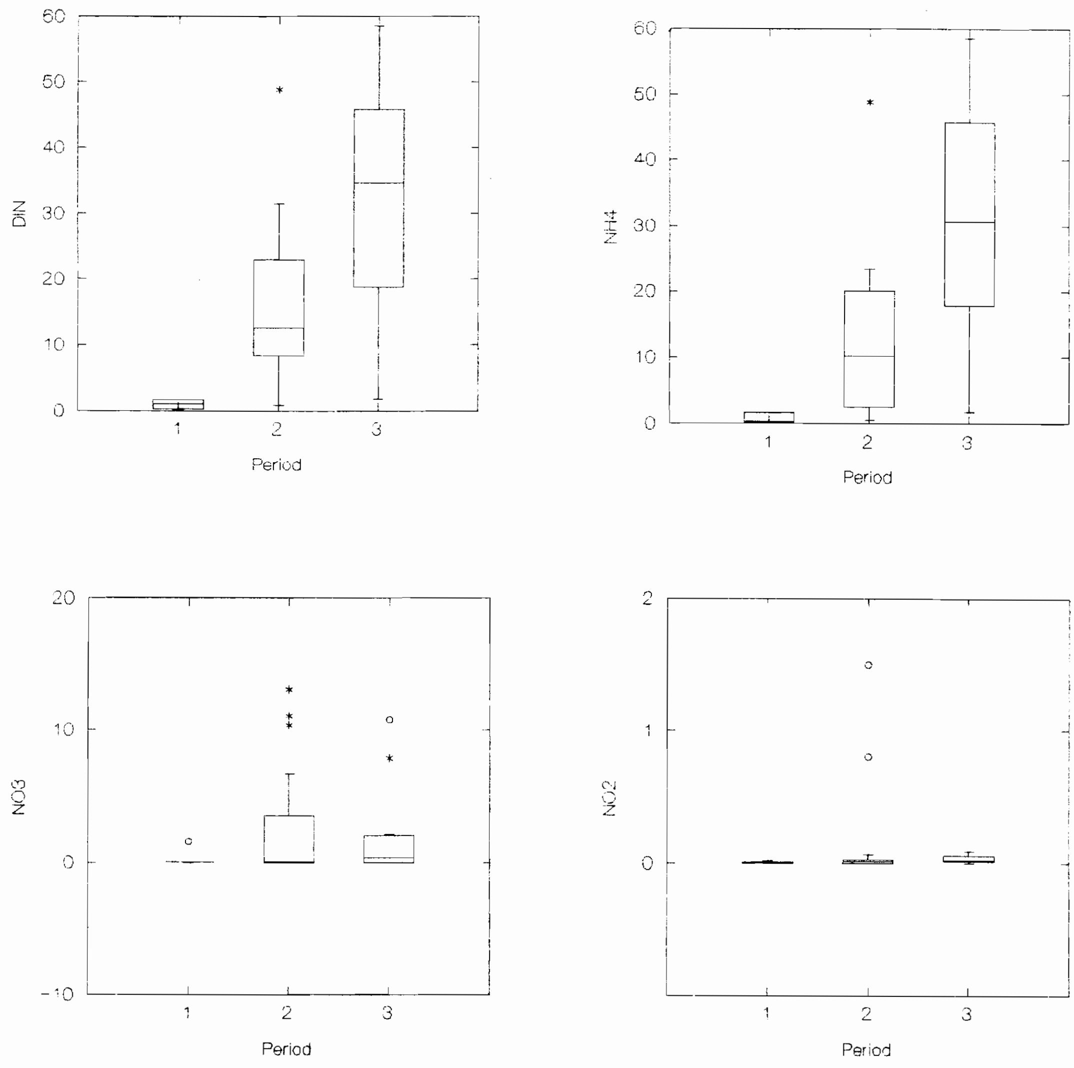



Figure 24. Temporal variation in dissolved inorganic nitrogen (DIN) and inorganic nitrogen species at the first tier wells in the mound drainfield. Periods 1, 2 and 3 represent before, during, and after the season, respectively. Concentrations are given in mg per liter. Numbers of samples used for period 1, N=6; period 2, N=29; period 3, N=12.

Table 14. Ammonium and nitrate concentrations and statistical comparisons for the upland wells at the mound site. Values are given in mg/L. Statistical comparisons were done using a Mann-Whitney U Test. Standard deviation is noted as STD.

| | | | $NH_4$ | | $NO_3$ | |
|---|---|---|---|---|---|---|
| | | | Mean | STD | Mean | STD |
| **Prior to season** | (N=6) | First tier | 0.72 | 0.80 | 0.29 | 0.67 |
| **During season** | (N=23) | First tier | 19.19 | 16.14 | 3.88 | 9.38 |
| | (N=9) | Second tier | 3.90 | 6.33 | 0.08 | 0.16 |
| **After fall close down** | | First tier | 30.16 | 18.17 | 2.04 | 3.55 |
| | (N=14) | Second tier | 8.72 | 15.03 | 0.04 | 0.14 |
| | | Statistical comparisons | | | | |
| First tier wells | | $NH_4 \leftrightarrow NO_3$ | | | $p<0.020$ (N=36) | |
| First tier wells | | $NH_4$ | | | | |
| | Prior to season↔Rest of sampling periods | | | | $p= 0.006$ (N=42) | |
| | During season↔After fall close down | | | | $p> 0.136$ (N=36) | |
| $NH_4$ | | First tier↔Second tier wells | | | $p<0.032$ (N=63) | |

The average dissolved inorganic nitrogen in the beach wells was 0.25 mg/L ($\sigma$=0.24 mg/L; N=15). While no significant difference (p=0.097; N=15) was found between DIN and nitrate concentrations, significant differences were found between ammonium and DIN (p=0.001; N=15), and ammonium and nitrate (p=0.039; N=15). Nitrate represented, on average, approximately 84% of the inorganic nitrogen. Nitrite concentrations generally were below detection limits. In contrast with the correlation measured between high DIN and low salinity values in the conventional beach wells, the DIN concentrations at the mound beach wells were not related to their salinities. In addition, no appreciable differences were found between the beach wells and ocean salinities during the sampling periods.

The DIP did not show significant differences ($p>0.052$; N=42) from reference values during the study. The first tier wells showed maximum DIP of 0.34 mg/L ($\sigma$=0.23 mg/L; N=6) during May. By September, the DIP decreased to a minimum of 0.07 mg/L ($\sigma$=0.08 mg/L; N=6), and increased again to 0.18 mg/L ($\sigma$=0.35 mg/L; N=6) by November. No significant differences ($p>0.474$; N=75) were found between DIP concentrations in the first tier wells and the rest of the upland wells, as well as, the beach wells. The average DIP concentration for the upland wells was 0.11 mg/L ($\sigma$=0.16 mg/L; N=24), and 0.04 mg/L ($\sigma$=0.03 mg/L; N=15) for the beach wells.

Ground water pH did not show temporal or spatial variations. The average upland ground water pH value was 6.78 (range=5.25-7.29), the average beach ground water pH was 7.42 (range=5.83-7.83), and the average ocean pH was 8.06 (range=7.57-8.13). Ground water temperature showed seasonal but not spatial variation. The minimum temperature was during May of 14°C and the maximum of 23°C occurred during the first days of September. Temporal and spatial variations were observed for salinity at the drainfield region. Salinity values and statistical comparisons are shown in Table 15.

## 5.3 Discussion

Hypothetical calculations were made to determine nitrogen and phosphorus inputs to the aquifer during the bath house operation. The average daily wastewater produced by comfort station Parking Lot #1 (mound site) was 10,845 liters. If 13.3 liters were used per toilet flush, there was an average of 819 flushes per day (assuming the wastewater was only produced by the commode operation).

The total nitrogen to the soil system was between 1.8 and 2.8 kg per day, assuming 3.46 grams of nitrogen per fecal toilet flush and 2.23 grams per non-fecal flush (University of Wisconsin, 1978). In addition to the nitrogen input by human excreta, there was an additional nitrogen input from the background nitrogen in the water used in the comfort station. The average DIN input by the bath house water was measured to be 5 mg/l or 0.05 kg daily.

Similarly, considering the total phosphorus input per event of fecal toilet flush to be 0.62 grams, and 0.23 grams per event of non-fecal toilet flush (University of Wisconsin, 1978), the total phosphorus input to the soil system was 0.19 to 0.51 kilograms per day. There are certain parameters that were not considered for simplicity purposes, but they could affect the input nutrient values. These parameters were: 1) the water was also used for cleaning purposes; 2) some of the commodes used higher volumes for flushing purposes; and 3) great number of children used the commodes. In addition, it was assumed that the

Table 15. Salinity values and statistical comparisons at the mound site. Values are given in mg/L. Statistical comparisons were done using a Mann-Whitney U Test. Standard deviation is noted as STD.

| | | | Mean | STD |
|---|---|---|---|---|
| **Prior to season** | (N=6) | First tier wells | 1.919 | 0.508 |
| **During season** | (N=24) | First tier wells | 3.277 | 1.266 |
| | (N=9) | Second tier wells | 3.816 | 1.381 |
| **After fall close down** | | First tier wells | 3.842 | 0.865 |
| | (N=14) | Second tier wells | 4.078 | 1.781 |
| **Beach wells 15** | (N=6) | | 31.033 | 1.039 |
| **Bath house water** | | | 4.072 | 0.075 |
| **Ocean water** | (N=8) | | 31.354 | 0.426 |
| Statistical comparisons | | | | |
| Prior to season↔During the season | | | p= 0.039 (N=30) | |
| During the season↔After fall closed down | | | p= 0.099 (N=36) | |
| First tier↔Second tier | | | p= 0.229 (N=59) | |
| Beach wells↔Upland wells | | | p= 0.001 (N=74) | |

conventional OSWDS had the same total nitrogen and total phosphorus inputs to the aquifer.

### 5.3.1 Conventional drainfield

The conventional drainfield exhibited two very distinct sections in terms of DIN and DIP variations. The two sections were; 1) the distal half section, composed of wells 1-4 (farther from the septic tanks), showed temporal and spatial variations of DIN and DIP; and 2) the proximal half section composed of wells 5 and 6 (closer to the septic tanks), which exhibited DIN variations at least one order of magnitude smaller than those in wells 1-4, and did not exhibit appreciable changes in DIP concentrations.

The ground water nutrient behavior surrounding the conventional drainfield was in agreement with the dye study results. The dye study established the possible existence of preferential wastewater flows under the drainfield, primarily, at the distal section of the drainfield under wells 1-3. Even though dye variations were measured in the drainfield section at well 4, these variations were smaller, and occurred later, than variations measured at wells 1-3. These same flow patterns were measured during nutrient analysis. Wells 1-3 exhibited a consistent variation in DIN and DIP concentrations during the study. These three wells showed significant variations in every inorganic nitrogen species, as well as, in DIP concentrations. On the other hand, the DIN variations at well 4 were governed by nitrate. Nitrate at this well represented 98% of the inorganic nitrogen species. In addition, DIP and nitrate variations in well 4 were significant, mainly, after August. The nutrient variations measured in well 4 manifested the possibility that the wastewater in that sector of drainfield moved by a combination of diffusion and advective transport mechanisms, contrary to wells 1-3 where the wastewater probably moved mainly by advective forces. Even though there were differences between wells 1-3, and well 4, the four wells were considered to constitute the distal section of the drainfield for simplification purposes.

A good relationship was observed at the distal half section of the drainfield between bath house usage and DIN variations. Dissolved inorganic nitrogen concentration was on average 0.95 mg/L ($\sigma$=0.42 mg/L; N=8) during January-March, increased to 54.40 mg/L ($\sigma$=32.20 mg/L; N=16) during bath house operation June-September and decreased to 26.40 mg/L ($\sigma$=9.36 mg/L; N=8) after the bath house closed (October-November) (Figure 22).

As mentioned in the literature review, the basic nitrogen forms entering the drainfield are ammonium and organic nitrogen, with average ratios of 75 and 25 percent, respectively. The organic nitrogen is quickly mineralized to ammonium once it is in the soil system (Canter and Knox, 1985). Therefore, the DIN entering the soil under the drainfield was primarily ammonium. During the study, anoxic conditions were only measured at well 1, during August-September. In the remaining wells, aerobic conditions prevailed. The average pH at these wells was 7.13, and the temperature varied between 15-26 °C. With these soil conditions, the ammonium could have been absorbed, leached, or nitrified.

The DIN species, ammonium and nitrate, can be used to determine wastewater contamination and basic soil characteristics adjacent to drainfield areas. For example, if nitrate concentrations are insignificant compared with ammonium concentrations in the ground water, at a certain distance form the drainfield, the soil system is most likely saturated and anoxic, from the drainfield to the measuring point. The third inorganic nitrogen species, nitrite, generally does not accumulate in the soils. Nitrite accumulations in soils are the result of alkaline soils and high ammonium levels (inhibition of *Nitrobacter*), or the existence of high nitrification potentials (Alexander, 1977).

These relationships among the DIN species were used to determine the possible soil characteristics under the conventional drainfield. Nutrient measurements taken in the drainfield wells were indicative that nitrification most probably did occur under the drainfield. Only on two occasions at the distal drainfield wells did anoxic conditions occur. In these two occasions, it was most probable that the wastewater saturated the soil path between the discharge pipes and the ground water under the drainfield nearby well 1, producing anoxic conditions. Even though adverse conditions of high wastewater loadings and high water tables occurred, on average, the soil and ground water were not depleted of dissolved oxygen. One of the indications that nitrification probably occurred under the drainfield was the relative high nitrite concentrations measured on several occasions in the distal drainfield wells. Given that the site pH was neutral, no apparent inhibition of *Nitorbacter* activity would be expected. Thus the nitrite measured was due probably to nitrification.

Variations in the ground water flow patterns, and solute transport mechanisms, under the conventional drainfield were suggested by the DIN and DIP measurements in well 4. Given the point in time of the drainfield operation when appreciable nutrient concentration variations were measured, diffusion was probably the predominant component of solute transport at that region of the drainfield. Four months after the drainfield was operational, a significant increment in ammonium concentration was measured in well 4. This increase in ammonium was most probably due to the modification in the ground water flow patterns due to the exceed of the soil percolation capacity. The nitrite concentrations measured at the same time would indicate that, with this additional ammonium, nitrification was taking place.

The piezometer set was used to observe, among other things, variations in nitrogen species due to potential nitrification in a small soil section immediately adjacent to the conventional drainfield. During August, nitrification was observed to have probably occurred mainly at two meters from the drainfield. Immediately adjacent, and at one meter from the drainfield, ammonium represented 99.7% of the DIN. At two meters from the drainfield, nitrate concentrations increased by 8000% at 0.5 m below WT and by 920% at 1.0 m below WT. Despite these elevated nitrate values, ammonium remained the major inorganic nitrogen species and represented 62.8% (at 0.5), and 97.8% (at 1.0) of the DIN pool. During September, nitrate was undetected at 0.5 and 1.0 m below WT. By the first week of October, the average daily wastewater loading rate decreased to approximately 44% of the loading rate of the previous two sampling periods. This reduction in wastewater loading appears to have affected nitrification. The coastal beach soils have porosities between 20-30%, and high hydraulic conductivities (Fetter, 1994). Thus, the ground water had a great capacity to return to aerobic conditions once the input of wastewater was reduced. This capacity of oxygenation was measured only at 0.5 m below WT, during October. Nitrate accounted for a considerable percent of the DIN at 0.5 m below WT (20% in well 1, 27% at PZA, and 98% at PZB). On the other hand, at 1.0 m below WT, ammonium represented 100% of the DIN. During the final three sampling periods, aerobic conditions continued to extend into the soil.

Once there was no more wastewater input to the system, nitrate became the major inorganic nitrogen species at 0.5 m below WT. At 1.0 m below WT, the nitrate concentration increased to represent similar percentage as the ammonium concentration of the DIN species.

In ground water with high redox potentials, nitrate is the stable form of dissolved nitrogen. In this type of environment, nitrate is the inorganic nitrogen species with the greatest leaching capacity. Once in the ground water, it is transported with potentially little transformation or retardation. Ammonium could also leach in the ground water when low redox potential prevails, or very large concentrations exist. In Chapter 4.0, Geohydrologic Characteristics, the difficulty in determining the net ground water flow patterns due to the effects introduced by tidal fluctuations was emphasized. This difficulty can be overcome if solute flow patterns are used to give an idea of the net ground water flow patterns. In an advective system, the solute is principally transported by ground water movement. Thus, the solute flow patterns could be used to indicate net ground water flow patterns.

During bath house operation, the drainfield behaved as a continuous contaminant source introducing into the aquifer an average daily concentration of contaminant, $C_{eff}$. As ground water transport occurred, the contaminant was spread due to hydrodynamic dispersion. Because longitudinal dispersivity is generally greater than transverse dispersivity, a solute transported by ground water would be more strongly dispersed in the flow direction than in directions normal to the flow lines. Because of the hydrodynamic dispersion, the initial concentration $C_{eff}$ decreased with distance from the drainfield. A continuous contaminant source produces a plume of contaminant in the direction of the ground water movement. Therefore, a particular point in the ground water flow direction, at a certain distance form the source, one would experience an increase in contaminant concentration with time (Fetter, 1994; Freeze and Cherry, 1979). Once the bath house was closed, and contaminant input ceased, the remainding contaminant under the drainfield could be thought as an instantaneous point source input. Thus, the contaminant starting at that moment, traveled as a slug that grew as it was dispersed when transported down the ground water flow path.

To give an idea of the net ground water flow patterns at the conventional OSWDS site, contour maps for DIN were drawn for August, October and November (Figures 25-27). The DIN contour maps showed preferential solute flow patterns toward the ocean and toward Toms Cove. During August (Figure 25), the kriged value[1] of the center point of the distal section of the drainfield was 86 mg/L, one to two orders of magnitude higher than any other upland, or beach well. The highest DIN concentrations, away from the drainfield, were measured at well 12 (3.17 mg/L), approximately 60 m toward the ocean form the drainfield, and in beach well B6 (2.29 mg/L), approximately 15 m from well 12. It is interesting to note that these two wells seemed to be in the same ground water flow path toward the ocean. By October (Figure 26), the DIN at the center point of the distal half section of the drainfield decreased to 29 mg/L. During this month, two upland wells had similar DIN concentrations to those found in the distal section of the drainfield. Well 10, approximately 30 m from the drainfield toward Toms Cove, measured 7.02 mg/L. The second well was the temporary well TB1 (Figure 15a), approximately 90 m east (toward the ocean) from the drainfield,

[1] Kriging is an statistical estimation procedure which allows to estimate the value of any unsampled location once measurements have been made at scattered sampling points.

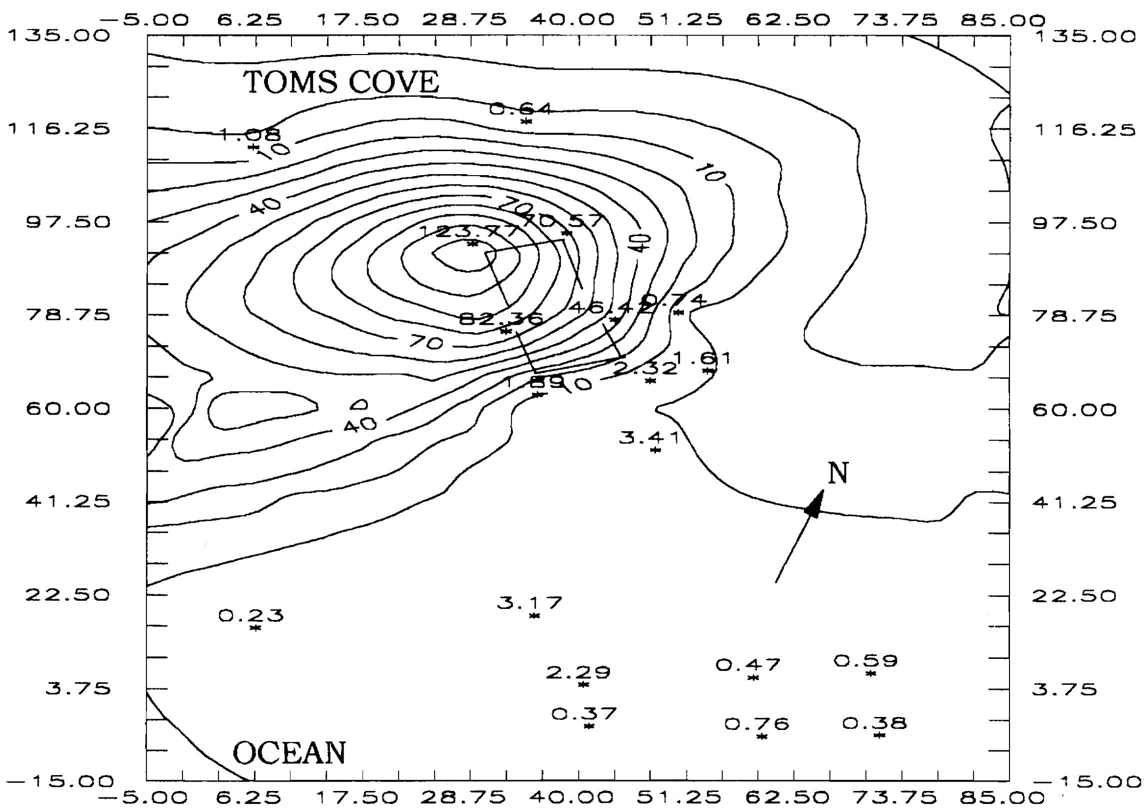



Figure 25. Dissolved inorganic nitrogen (DIN) contour map for August sampling period at the conventional site. Axis are in meters. DIN concentrations are expressed in mg per liter.

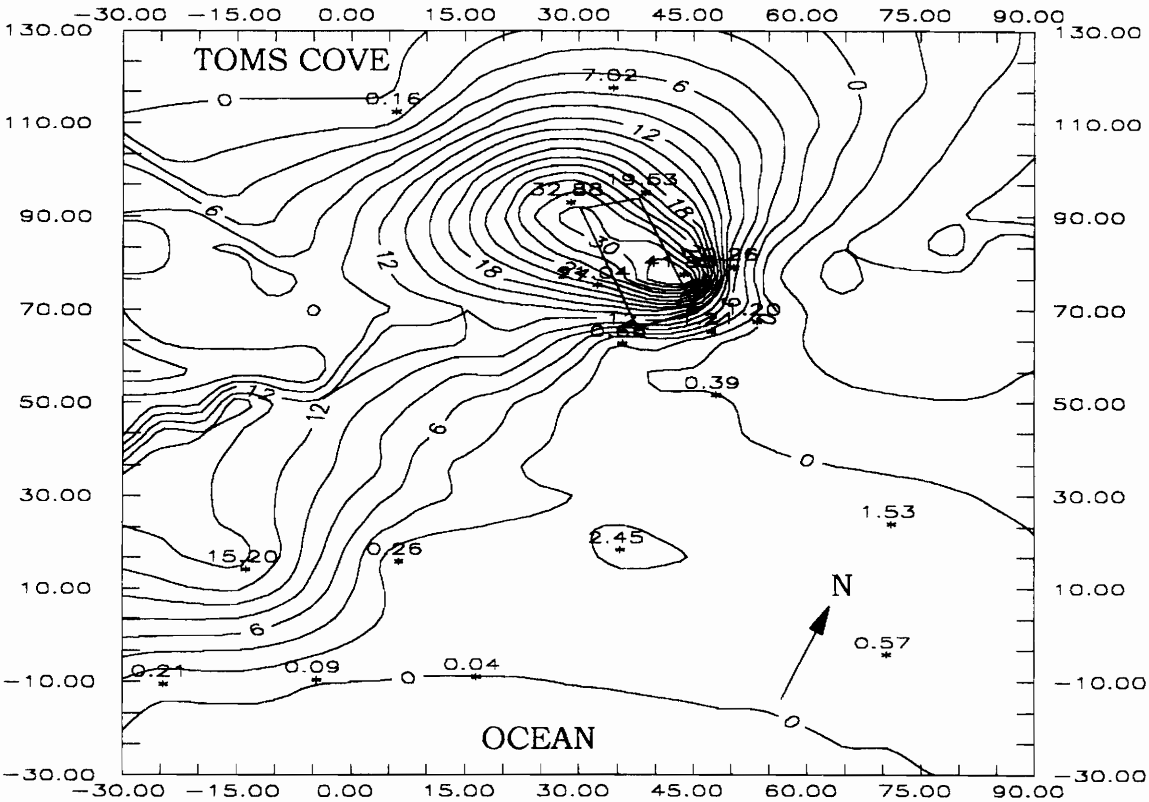



Figure 26. Dissolved inorganic nitrogen (DIN) conour map for October sampling period at the conventional site. Axis are in meters. DIN concentrations are expressed in mg per liter.

measured 15.2 mg/L. By November (Figure 27), the DIN concentration at well 10 decreased to 0.32 mg/L. Similarly, a DIN reduction was measured in the region of the well TB1 with a value of 6.09 mg/L. During November, wells 7, 12, and 13 had DIN increments of 1190%, 686%, and 704% compared with the October values. A specific ground water flow pattern was not detected by analysis of the contour plots. The solute was transported in two general directions, towards Toms Cove, and towards the ocean. Additionally, possible preferential flow patterns, from the drainfield to the surface waters bodies, became visible in the three contours. If preferential flow patterns existed, as i.e., channelized flows, the solute transport time from the drainfield to the surface water would have been reduced considerably.

The solute contour maps can be used to calculate ground water velocities. In order for a well, at a certain distance, d, from the drainfield, to measure at a given time, t, a specific solute concentration, Co, the ground water must transport the solute at a certain velocity, $v_x$. Analytical solutions of two advection-disperion equations were used to determine the average linear ground water velocities. The two equations were, Ogata and Banks (Fetter, 1993)), and Sauty (1980). Ogata and Banks (Fetter, 1993) considered the solute concentration injected into the aquifer to be constant in their solution.

$$C = \frac{C_0}{2}\left[erfc\left(\frac{L - v_x t}{2\sqrt{D_L t}}\right) + \exp\left(\frac{v_x L}{D_L}\right) erfc\left(\frac{L + v_x t}{2\sqrt{D_L t}}\right)\right]$$

where:

erfc = complementary error function

$C_0$ = initial concentration (mg/L)

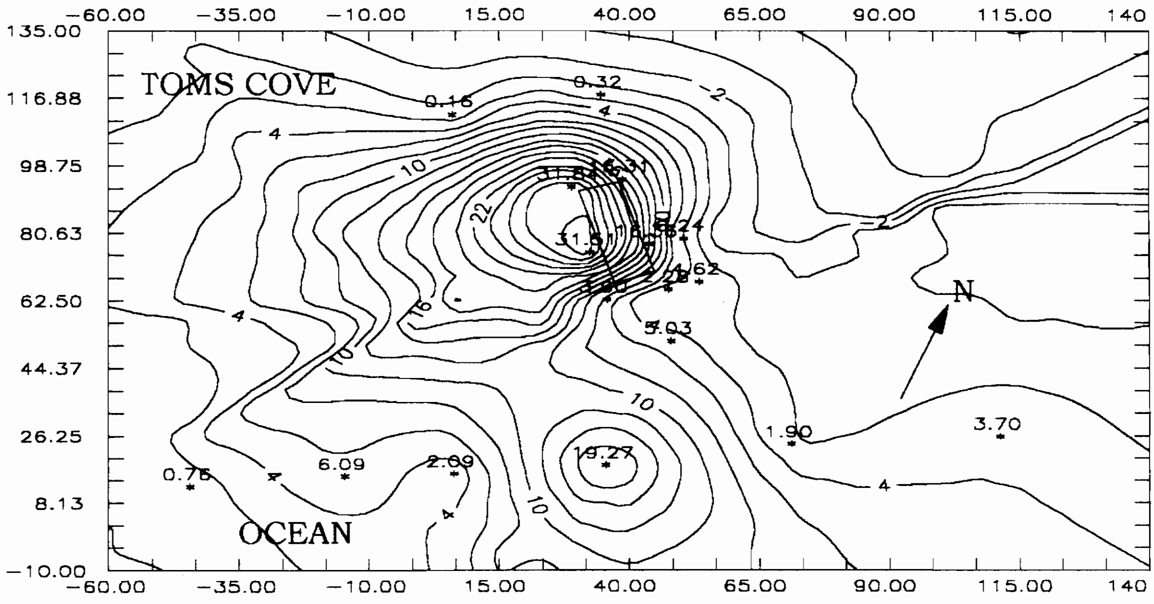



Figure 27. Dissolved inorganic nitrogen (DIN) conour map for November sampling period at the conventional site. Axis are in meters. DIN concentrations are expressed in mg per liter.

$C$ = tracer concentration at time t (mg/L)

$D_L$ = longitudinal hydrodynamic dispersion (m)

$v_x$ = average linear ground water velocity (m/day)

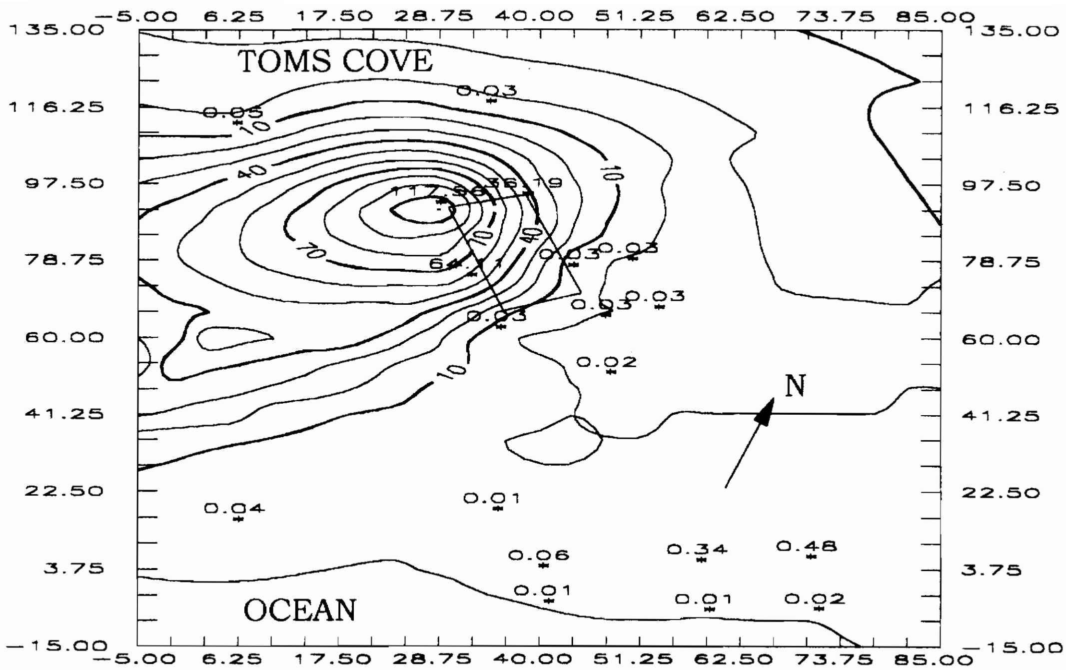
$L$ = flow path length (m)

On the other hand, Sauty (1980) considered the injected mass of the solute proportional to the duration of the injection.

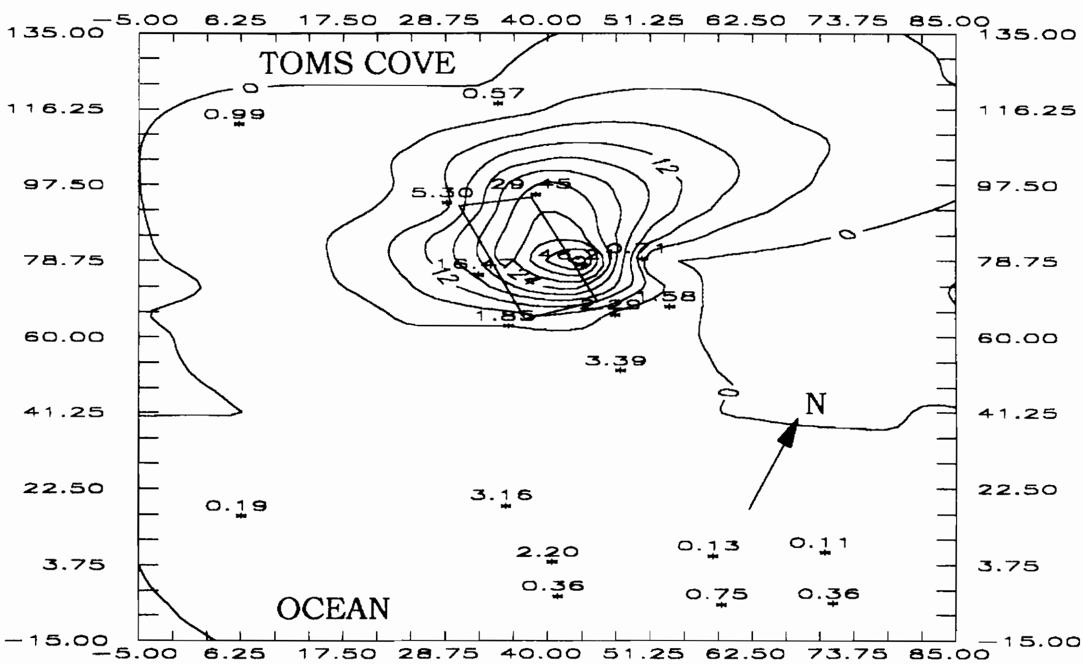
$$C = \frac{C_0}{2}\left[erfc\left(\frac{L - v_x t}{2\sqrt{D_L t}}\right) - \exp\left(\frac{v_x L}{D_L}\right) erfc\left(\frac{L + v_x t}{2\sqrt{D_L t}}\right)\right]$$

The longitudinal dispersion coefficient $D_L$ is equal to $a_L v_x + D^*$, where $D^*$ is the effective diffusion coefficient, and $a_L$ the longitudinal dynamic dispersivity. The longitudinal dynamic dispersivity was calculated using Neuman (Fetter, 1994) approximation; $a_L = 0.0175 \times L^{1.46}$. The effective diffusion coefficient was not considered in the advection-dispersion equation given that in a flow dominated by advective forces it is negligible compared with the longitudinal dynamic dispersivity. For simplification purposes, the source was considered to be continuous over the time period the calculations were made. Therefore, the advection equations were used to determine the average linear ground water velocities. Given that the velocities determined by the two equations were very similar, the two solutions are given as a range. For the DIN concentration to be 7.02 mg/L during October in well 10, the ground water velocity had to be between 0.412-0.417 m/day. Similarly, for the case of well TB1, the ground water velocity had to be between 1.23-1.31 m/day, and for well 12, the ground water velocity had to be between 0.195-0.205 m/day. With these velocities, it would have taken the effluent wastewater approximately between 6 months to reach Toms Cove, and approximately between 3 to 16 months to reach the Ocean. These times are smaller than the velocities calculated employing the fresh hydraulic heads (2.9-7.8 years to reach Toms cove and at least 3.36 years to reach the Ocean). Therefore, these results reaffirm the possibility of preferential flow path from the septic field to the surface waters.

In addition to the DIN contour maps, ammonium and nitrate contour maps were used to observed how the wastewater was nitrified close to the conventional drainfield as it was transported by the ground water (Figures 28-30). The ammonium contour maps were similar to the DIN contour maps for the region around the drainfield. In turn, the nitrate contour maps were similar to the DIN contour maps of the regions distant from the drainfield. These similarities and differences between the DIN and ammonium-nitrate contour maps were expected. As the ammonium was transported by the ground water away from the drainfield, oxygenated ground water was encountered and the nitrification process would have been enhanced. For this reason, the highest nitrate values were found at the regions near the ocean. As the rich oxygen ocean waters advanced inland due to the tides, redox potentials of the ground water system increased, thus potentially enhancing nitrification processes. During August, nitrification was most probably taking place slowly around the drainfield region (Figure 28). By October and November (Figures 29, 30) the wastewater traveled large distances from the drainfield encountering, in its way, aerobic ground water. Thus, high nitrate concentrations were measured at the upland wells. Once the bath house was closed, the aquifer had received a total of approximately 225-345 kg of nitrogen. This nitrogen was transported, spread, and diluted by the ground water, as the biota performed the

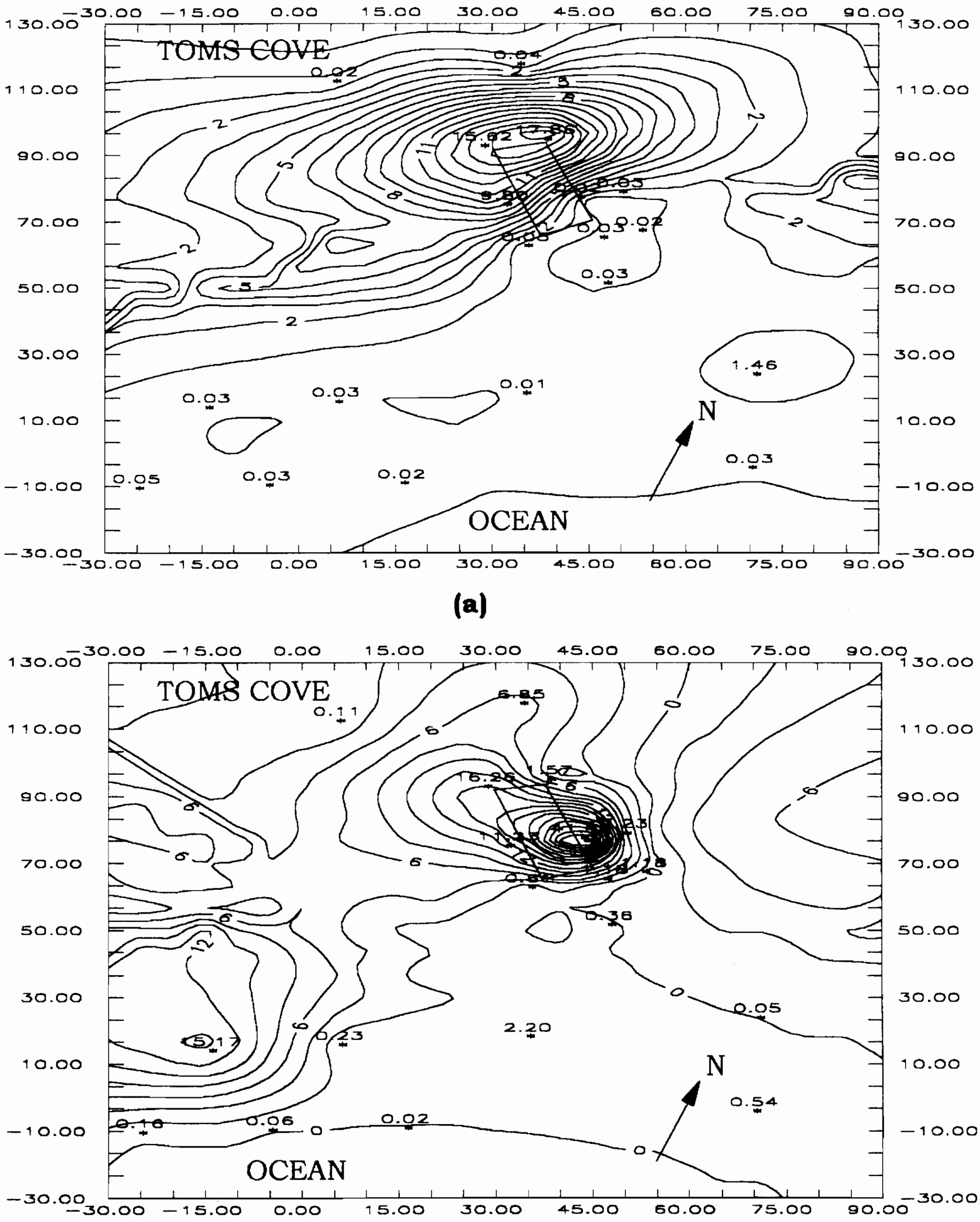



(a)




(b)

Figure 28. Ammonium (a) and nitrate (b) contour maps for August sampling period at the conventional site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.




(a)

(b)

Figure 29. Ammonium (a) and nitrate (b) contour maps for October sampling period at the conventional site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.

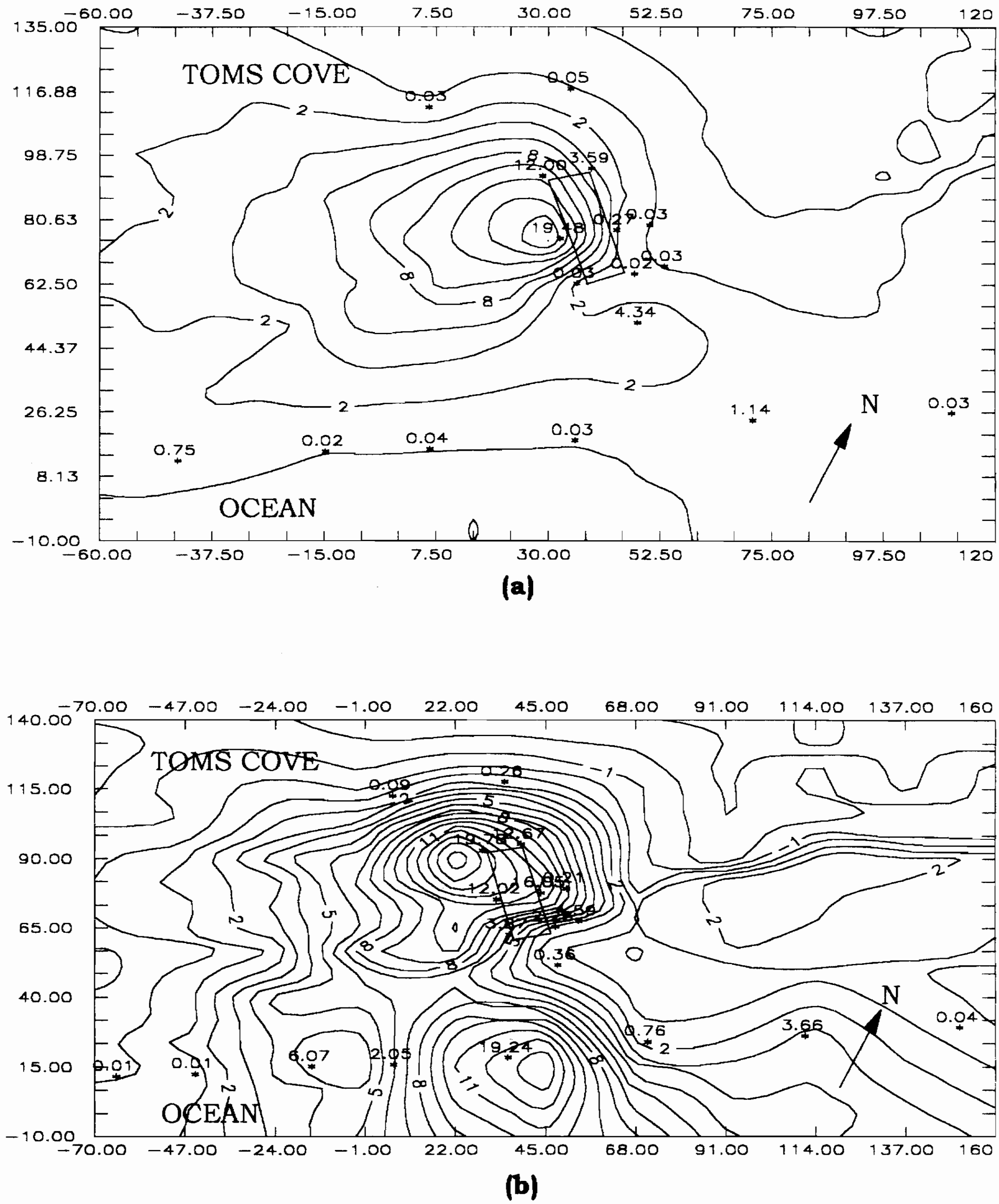



Figure 30. Ammonium (a) and nitrate (b) contour maps for November sampling period at the conventional site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.

nitrification process. These findings largely corroborate those reported in the literature. As Walker et al. (1973) concluded, the only active mechanism for lowering the nitrate, formed by nitrification from the significant quantities added by septic tank systems effluents in sand, is by dilution with uncontaminated ground water.

Temporal and spatial DIP variations were measured on the distal section of the conventional drainfield. As mentioned in the literature review, most of the phosphorus in the septic tank effluent is in the soluble orthophosphate form. Phosphate ions, once in the soil environment, are readily chemisorbed to mineral surfaces. In acid to neutral soils, the chemisorption is mainly to iron (Fe) and aluminum (Al). In neutral to alkaline soils, the chemisorption occurs to Ca minerals. The average pH at the conventional site was 7.2, which meant near neutrality conditions prevailed through the study. It is well known that DIP does not have a similar leaching capacity as nitrate. During the study, no significant DIP variations were measured in any of the upland wells except one. Thus, even though a total of approximately 23-61 kg of total phosphorus were discharged into the aquifer, the phosphorus did not leach long distances from the drainfield. The only upland well which measured comparable DIP to those found adjacent to the drainfield, was well D3. Well D3 was approximately 3 m from the drainfield.

To give an indication of the soil phosphorus adsorption capacity under the conventional drainfield the following assumptions were made. First, the soil volume through which the wastewater had to percolate was equal to the conventional bed area times 0.7 m (distance from the bottom of the gravel pack to the average WT elevation). Second, the soils at the site were considered to have a bulk density of 1.6 g/cubic centimeter. Third, the soil had a maximum Langmuir adsorption coefficient of 90 micrograms of phosphorus per gram of soil. Fourth, the input of total phosphorus to the aquifer was between 0.19 and 0.51 kg per day. With these assumptions, it would take approximately between one to three months for soil saturation.

Several laboratory experiments showed that phosphorus moves slowly through soil columns (Sawhney and Starr, 1977). In all of these experiments, the conclusion was the same, the movement of phosphorus through soil columns is minimal until the sorption sites were occupied. From that point on, the phosphorus was not adsorbed and leaching through the soil column occurred. Field measurements showed that orthophosphate concentrations were significant during August, in well D3, almost two months after the drainfield was operational. Thus, field measurement were in agreement with literature results and with the saturation time of phosphorus sorption sites calculated previously. If the travel time for the phosphorus to reach well D3 was equal to the time necessary for the phosphorus to saturate the sorption sites in the soil volume between the drainfield and well D3, then the travel time was approximately between one and four months.

### 5.3.2 Mound drainfield

Temporal and spatial variations were observed for DIN species in the first tier wells. The DIN did not show a good correlation with bath house usage. Dissolved inorganic nitrogen was on average 1.01 mg/L ($\sigma$=0.81 mg/L; N=6) during May, increased to 27.19 mg/L ($\sigma$=17.23 mg/L; N=18) during July-September, and continued to increased 32.24 mg/L ($\sigma$=17.36 mg/L; N=12) during October-November (Figure 31). It was assumed that under most of the mound drainfield a peat layer existed to

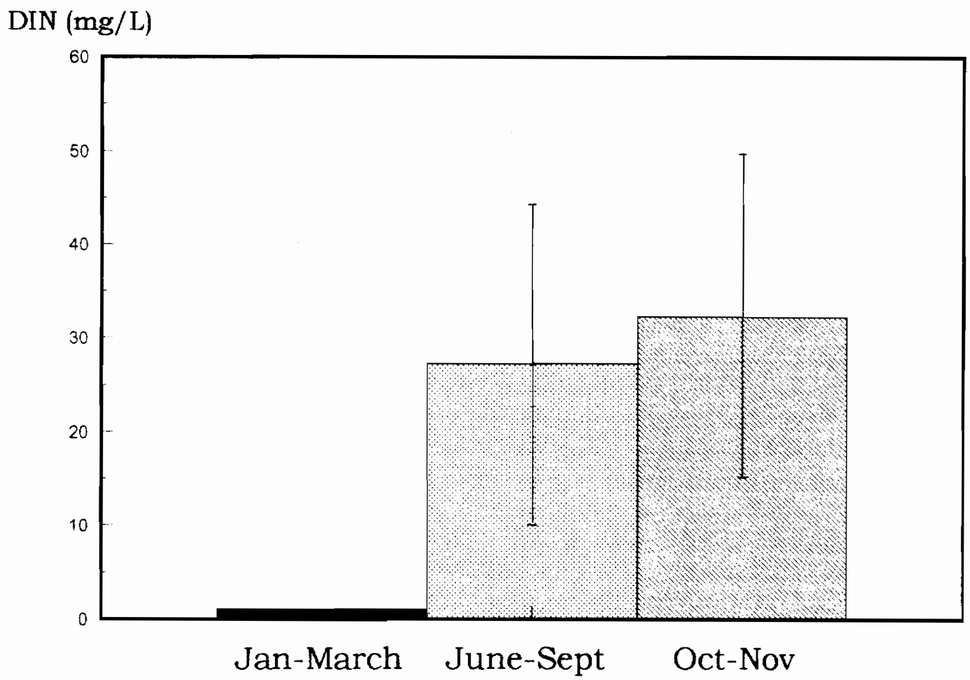


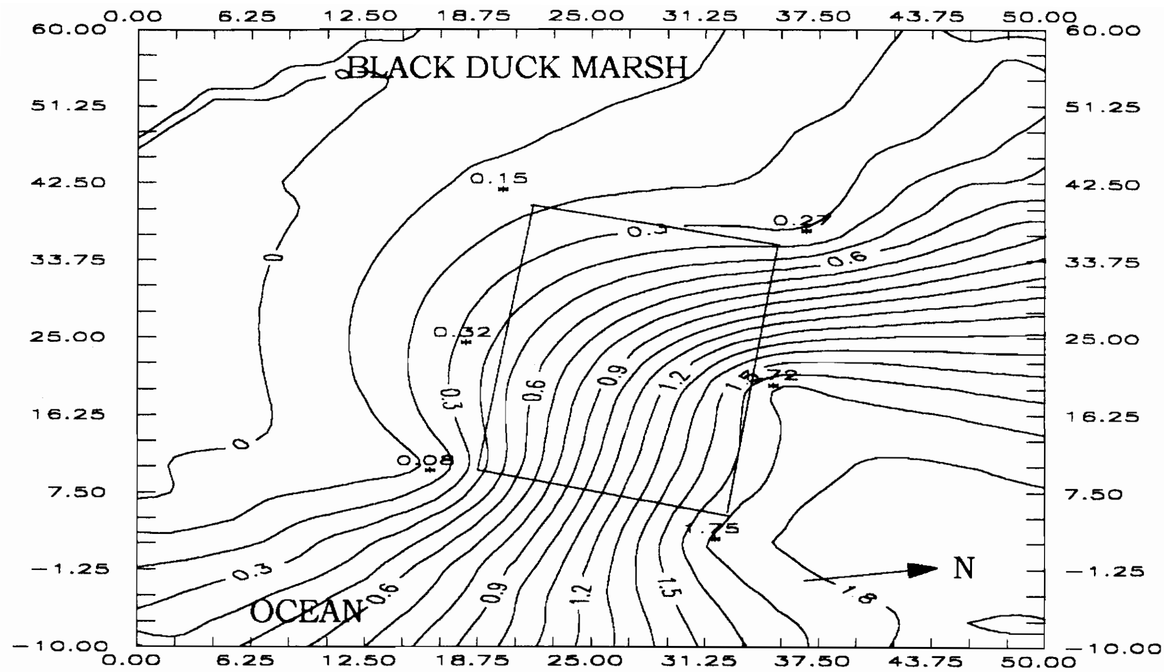

Figure 31. Relationship between bath house usage and DIN variations in the mound drainfield.

explain the absence of a DIN reduction in the ground water immediately adjacent to the drainfield once the bath house was closed. This peat layer acted as a confining layer causing ponded conditions and produced a perched water table. This reservoir of DIN continued to flow slowly through the sediments, and as a result, no appreciable decrease in DIN was measured. This hypothesis was supported by the ground water salinity measurements. The average salinity in the first tier wells increased from 1.921 psu during May to 3.656 psu during bath house operation (the water used in the bath house had an average salinity of 4.072 psu). Once the bath house closed in the fall, no appreciable decrease in salinity was measured, neither during October (4.048 psu), nor during November (3.635 psu). The indication that a reduction in DIN would have occurred in the adjacent wells to the drainfield was given by the well in the middle of the mound (Top well). The DIN in this well was almost constant during September and October, and decreased to half its value by November.

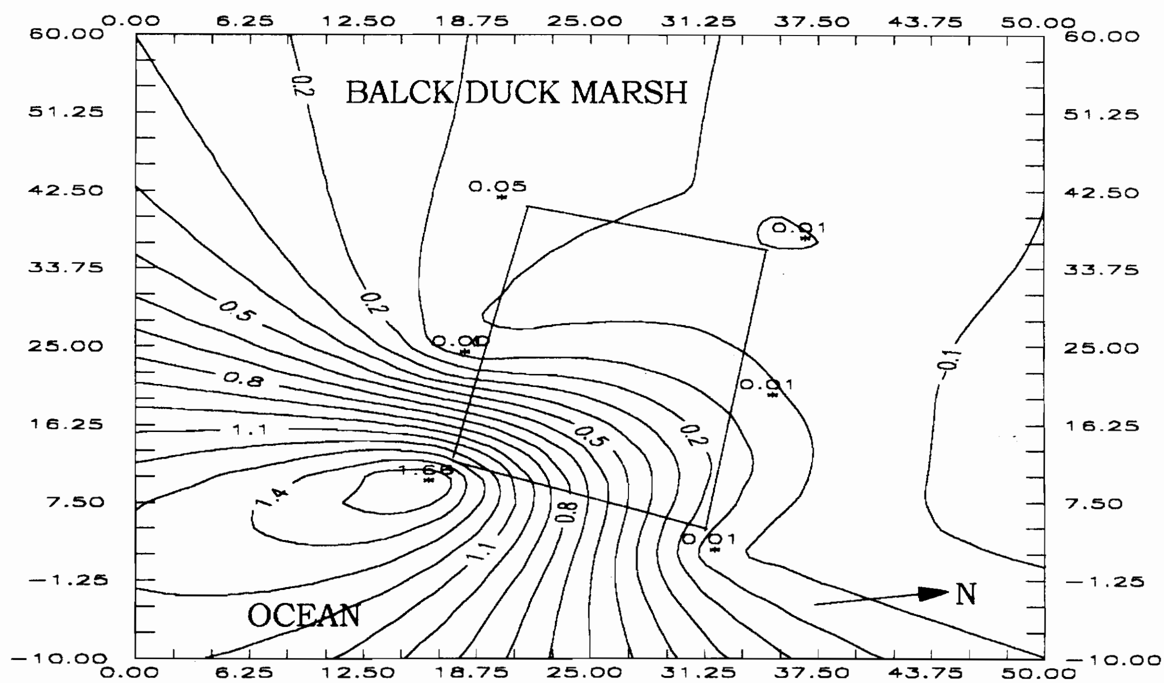
Even though ammonium was the major inorganic nitrogen species (90%) in the first tier wells, nitrate concentrations up to 44.67 mg/L were measured during the study. On average 64% of the nitrate measurements in the first tier wells were below 0.5 mg/L. Thus, nitrification signs were almost negligible. The nitrification process was not inhibited by temperature or pH. The average pH was on the acid side measuring 6.61 (Range= 6.15-7.29). Temperatures varied between 14-24 °C.

General anoxic conditions were measured in the upland wells. Additionally, anoxic conditions were measured on several occasions in some of the first tier wells. Bouma et al. (1974b) found that nitrification was completed within the fill at the mound system they studied. The findings by Bouma et al. (1974b) were in agreement with the results obtained by Magdoff et al. (1974a) during nitrification studies in sand columns representing mound systems. Harikin et al. (1979) found that denitrification removed, on average, 44% of the nitrate formed in a study of thirty-three mound systems. During December, the mound drainfield at the site was opened to observe the condition of the sand fill. No evidence that anoxic conditions existed were found, the sand was clean and clear. Thus, denitrification could be the reason why low nitrate values were measured immediately adjacent to the drainfield in this study. Magadoff et al. (1974b) found that denitirfication is possible at the fill-topsoil interface where nearly saturated conditions could exist. Additionally, the mound site showed characteristics suitable for denitrification to occur. For example, general anoxic conditions prevailed in the upland wells where ammonium concentrations represented approximately 98% of the DIN. Therefore, it is feasible that, in addition to the possible nitrate reduction of 44% manifested by Harkin et al., high denitrification potentials existed in the soil and ground water at the mound site in this study. Thus, if this was the case all the nitrate formed was transformed to nitrogen gas by denitrification.

To observed how the wastewater was nitrified around in the mound drainfield system, ammonium and nitrate contour maps were drawn for May, September, and November (Figures 32-34). The three nitrate contours were mainly driven by three immediately adjacent wells to the drainfield (wells 2, 4 and 5). No nitrification seemed to have occurred with distance from the drainfield. During the three months, ammonium seemed to be the major nitrogen species leaching through the ground water. Therefore, given the high ammonium concentrations found around the mound drainfield, it seems that nitrification at the mound was not in




**(a)**




**(b)**

Figure 32. Ammonium (a) and nitrate (b) contour maps for May sampling period at the mound site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.

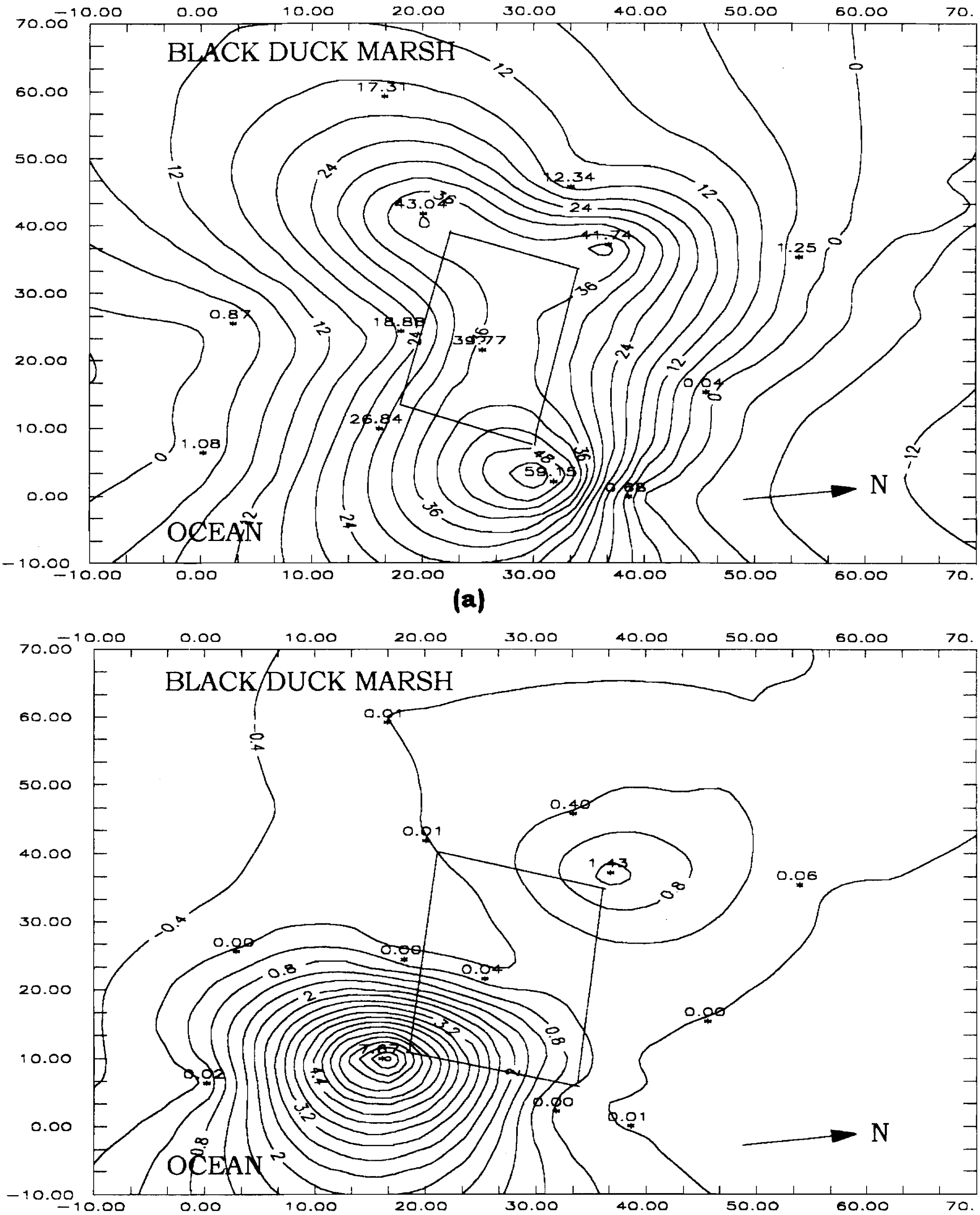



Figure 33. Ammonium (a) and nitrate (b) contour maps for September sampling period at the mound site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.

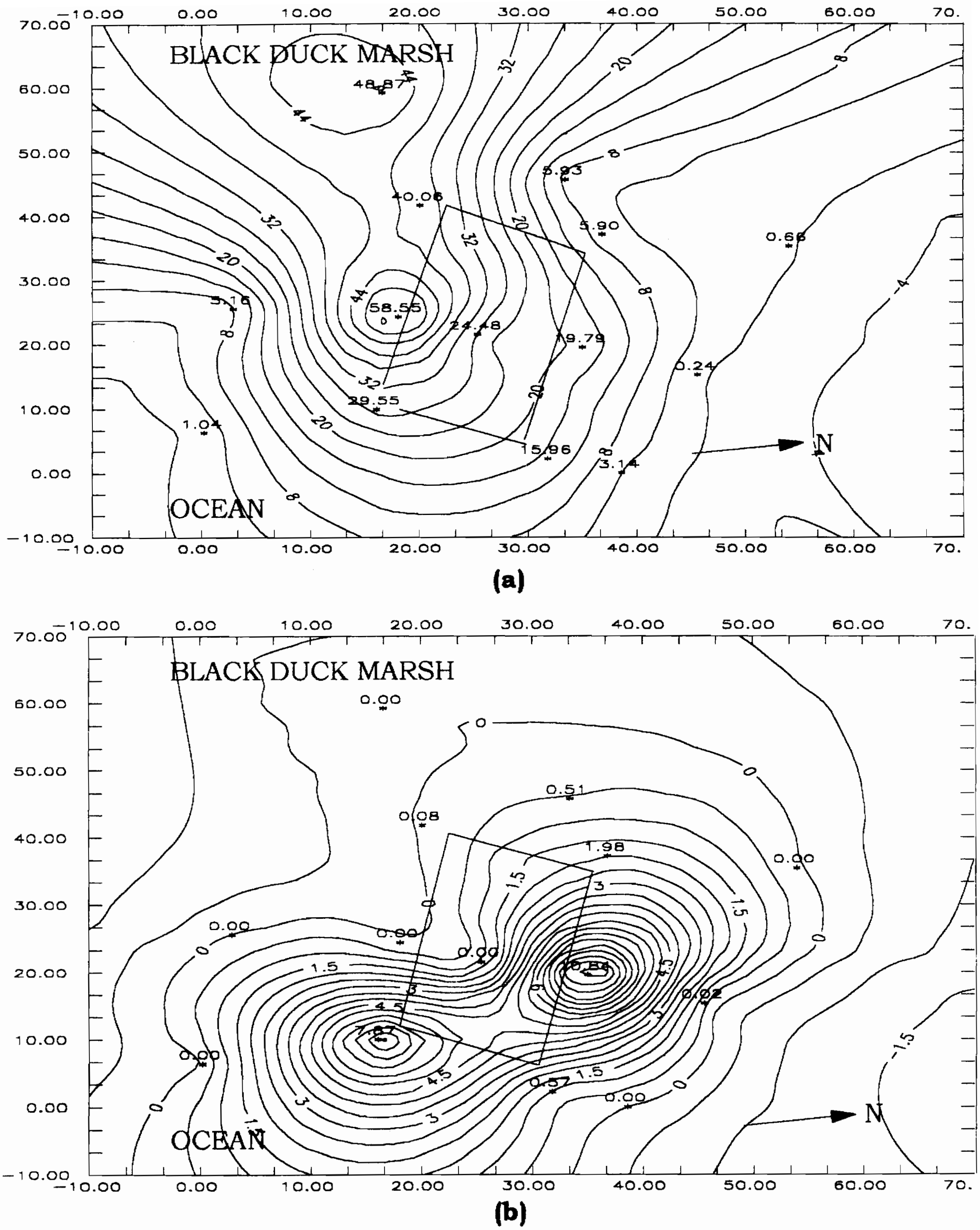



Figure 34. Ammonium (a) and nitrate (b) contour maps for November sampling period at the mound site. Axis are in meters. Nutrient concentrations are expressed in mg per liter.

agreement with results found by Bouma et al. (1974b), and Magadoff et al. (1974a).

Similar to the conventional site, DIN contour maps were drawn to give a general indication of net ground water flow patterns at the mound site (Figures 35-37). The DIN contour maps showed preferential flow inland, toward Black Duck marsh. During August (Figure 35), the Kriging DIN value of the middle point of the mound drainfield was one to two orders of magnitude higher than any other upland or beach well. By September (Figure 36) the DIN at the center point of the drainfield was approximately 33 mg/L. During this month the upland wells in the direction of Black Duck marsh, wells 10 and 11, had comparable DIN values to those found immediately adjacent to the mound (well 10, 12.75 mg/L; well 11, 17.32 mg/L). While further increase in DIN was measured in well 11 (48.89 mg/L), by November (Figure 37), the DIN in well 10 decreased to 6.47 mg/L. During this month the upland well with the highest percent change in DIN from September values was well 12. The DIN in well 12 increased from 0.87 mg/L to 5.17 mg/L, an increase of approximately of 495%. During the same period, the immediately adjacent well 3 which was approximately 15 m from well 12 measured a DIN increased of approximately 210%.

It is most probable that the wastewater also traveled toward the ocean, but the continuous monitoring of the ground water in that direction was restricted by the sand dune. The sand dune was almost immediately adjacent to the drainfield, and it extended approximately to the high tide mark. Thus, the monitoring of the wastewater toward the ocean could only be performed with the beach wells. Quick dilution probably affected the concentration of any solute in the region between the sand dune and the ocean. Given that no beach well measured

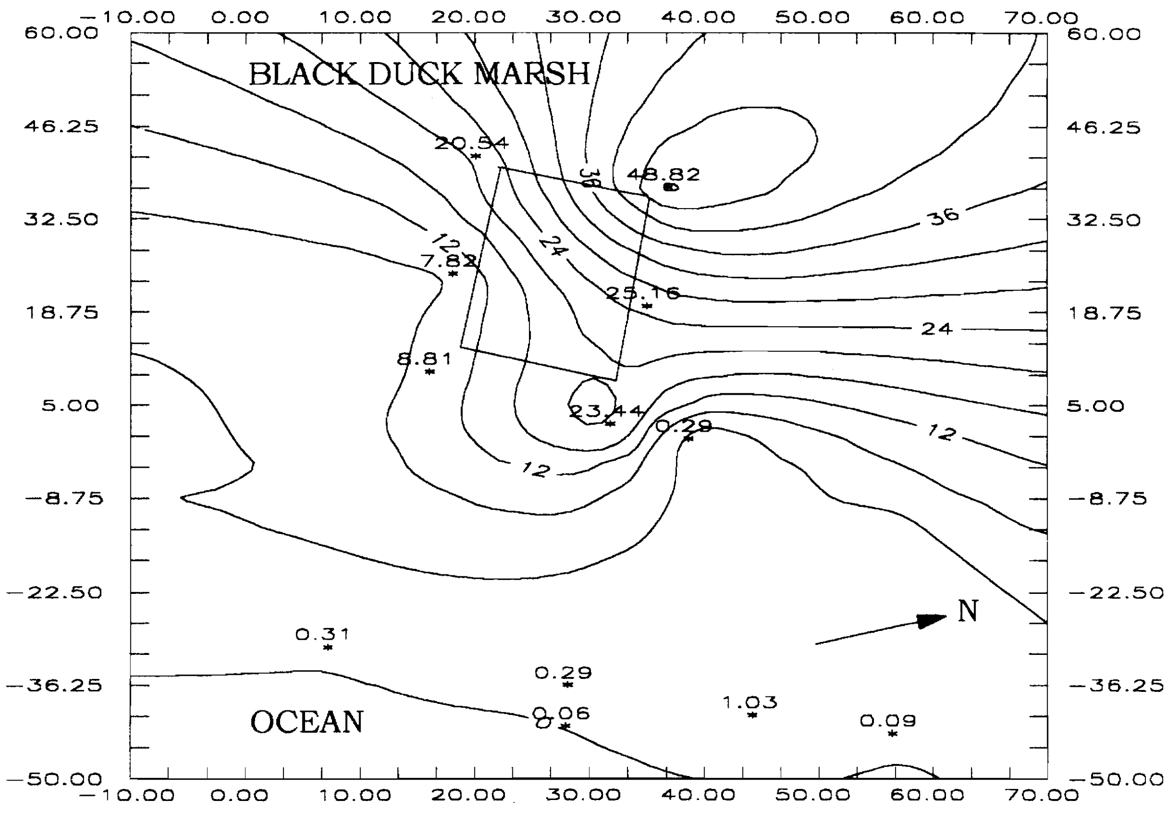



Figure 35. Dissolved inorganic nitrogen (DIN) countour map for August sampling period. Axis are in meters. DIN concentrations are expresssed in mg per liter.

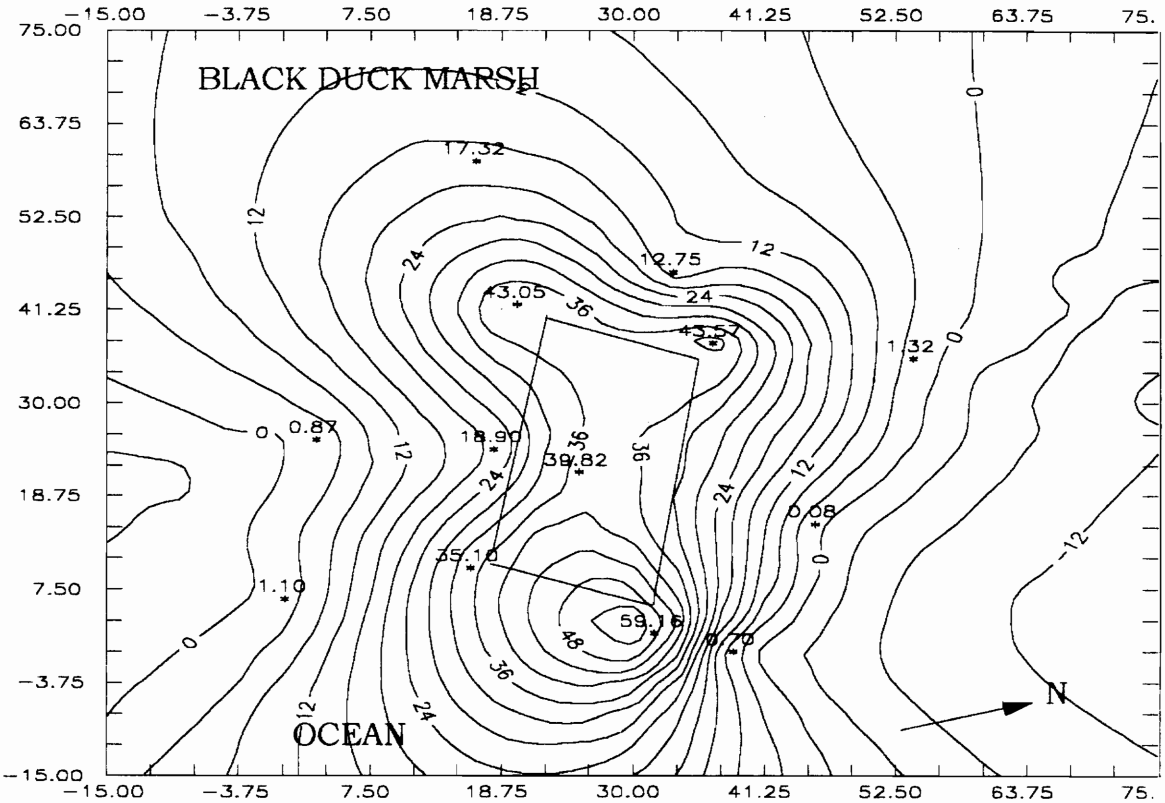



Figure 36. Dissolved inorganic nitrogen (DIN) countour map for September sampling period. Axis are in meters. DIN concentrations are expresssed in mg per liter.

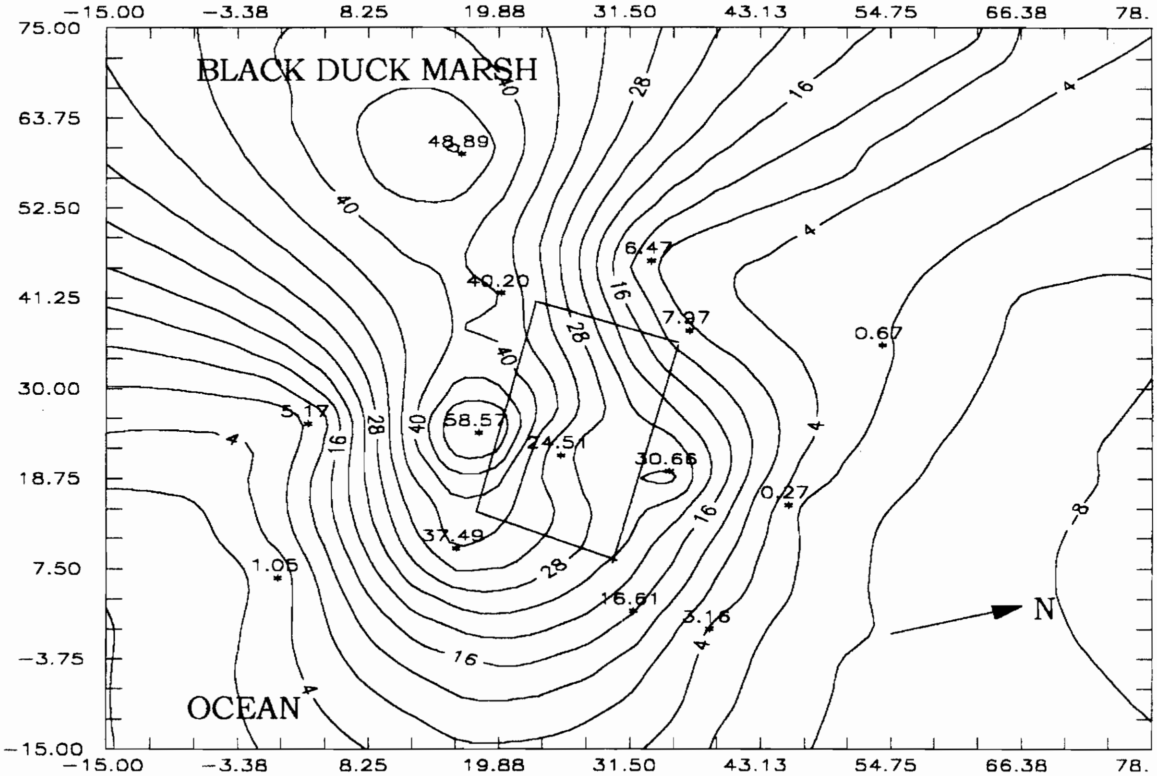



Figure 37. Dissolved inorganic nitrogen (DIN) countour map for November sampling period. Axis are in meters. DIN concentrations are expresssed in mg per liter.

significant nutrient concentrations, the advective-dispersion equations could not be used to calculate possible ground water velocities toward the ocean. In the Chapter Geohydrologic Characteristics of this thesis, ground water flow patterns toward the ocean were calculated with fresh hydraulic heads. Consequently, even though the solute data did not show clear ground water flow patterns, this is not conclusive, and it is possible that wastewater was flowing toward the ocean. In addition, preferential flow paths could have existed, but these were not detected by the beach wells.

The same procedure, as previously used for the conventional site, was employed to determine ground water velocity from the solute data, at the mound site. The velocities calculated using the concentrations in well 10 were 0.295 m/day for September, 0.231 m/day for October, and 0.220 m/day for November. For well 11, the ground water velocities were 0.468-0.476 m/day for September, 0.315-0.321 m/day for October and between 0.241-0.246 m/day for November. For well 12, the ground water velocities were 0.296 m/day for September, 0.198 m/day for October, and 0.162 m/day for November. With these velocities, it would take the wastewater between 3 and 6 months to reach the Black Duck Marsh. It is interesting to note that all velocities, at each well, decreased from September to November which correspond with decreased bath house usage.

The mound site did not show significant DIP variations during the study. The average DIP in the first tier wells was less than 0.2 mg/L. This low DIP could be due to the fact that the mound was constructed with clean materials and in an unused section of land. Thus, the soils in the mound and under it had a considerable phosphorus absorption capacity. To give an idea of the soil phosphorus absorption capacity under the mound drainfield, calculations similar to the ones made for the conventional drainfield were made for the mound drainfield. For the mound drainfield, the volume through which the wastewater had to percolate was equal to the mound bed area times the distance from the top of the mound sand fill to one meter below ground level. With a daily input of total phosphorus between 0.19-0.51 kg, it would take approximately four to 10 months for soil saturation.

### 5.3.3 Conventional and mound drainfields

To further facilitate the comparison of the performance between the conventional and the mound OSWDS, a summary analysis between the two was done. All statistical tests were done using the Mann-Whitney U Test (Systat, 1992). No significance differences ($p>0.63$; $N=14$) were observed in DIN and DIP between the two sites for the time prior to drainfield operation. In the conventional drainfield, DIN showed a good relationship with bath house operation. Figure 38 shows the kriged values of the center point of the distal half section of the conventional drainfield. The DIN increased from 4.4 mg/L in May to a maximum of 86 mg/L during August. On the other hand, the DIN in the ground water immediately adjacent to the mound drainfield (Figure 38) increased with bath house usage, and after fall close-down. The DIN measured in the last two months were 36 and 41 mg/L. While no significant differences in DIN ($p=0.270$; $N=32$) were found during June-August between the two drainfields, significant differences ($p=0.043$; $N=24$) were measured during October-November. Like the DIN, the DIP immediately adjacent to the conventional drainfield showed a good relationship with the bath house operation (Figure 39). In contrast, the average DIP immediately adjacent

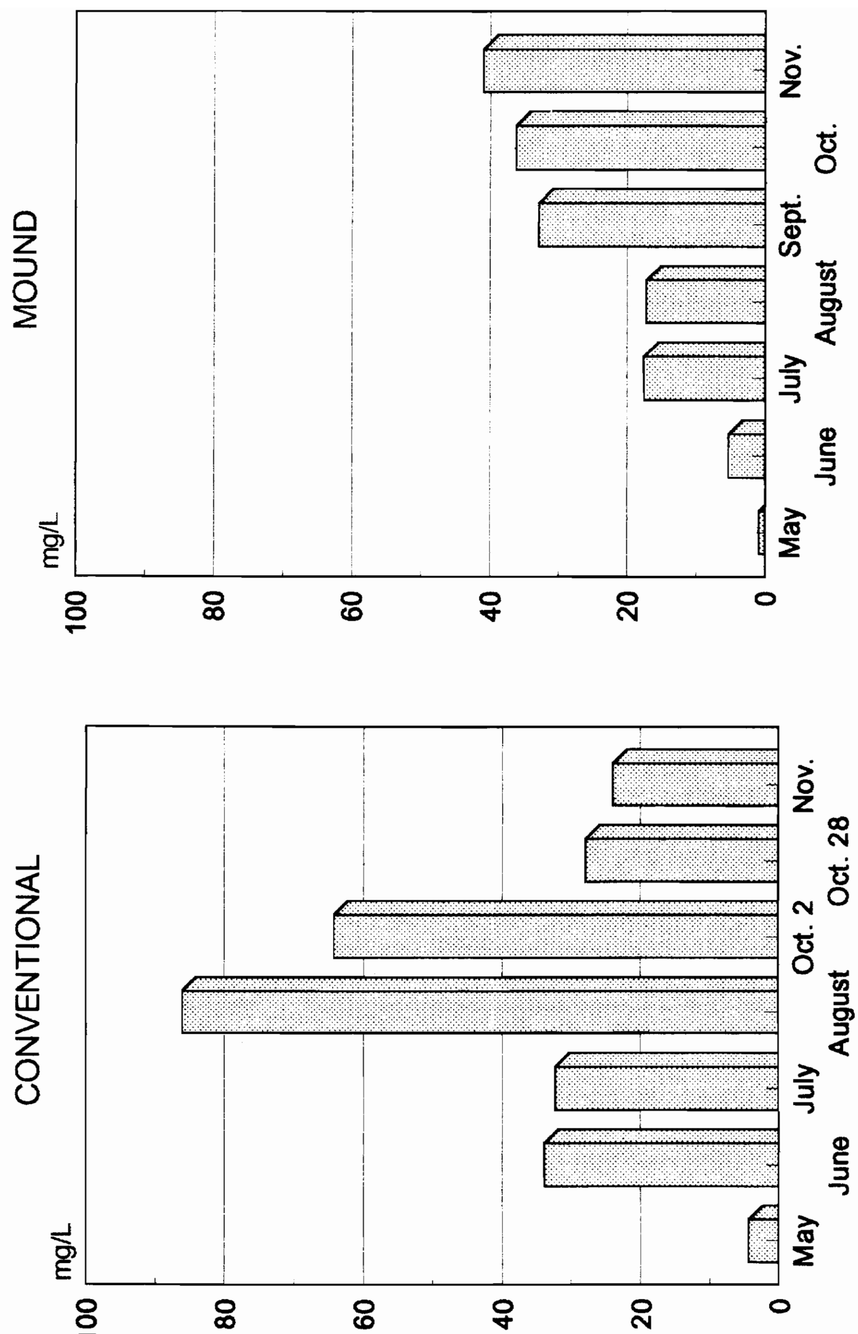



Figure 38. Dissolved inorganic nitrogen concentrations immediately adjacent to the two drainfields. Charts show the Kriging value of the center point of the drainfields. Concentrations are given in mg/L.

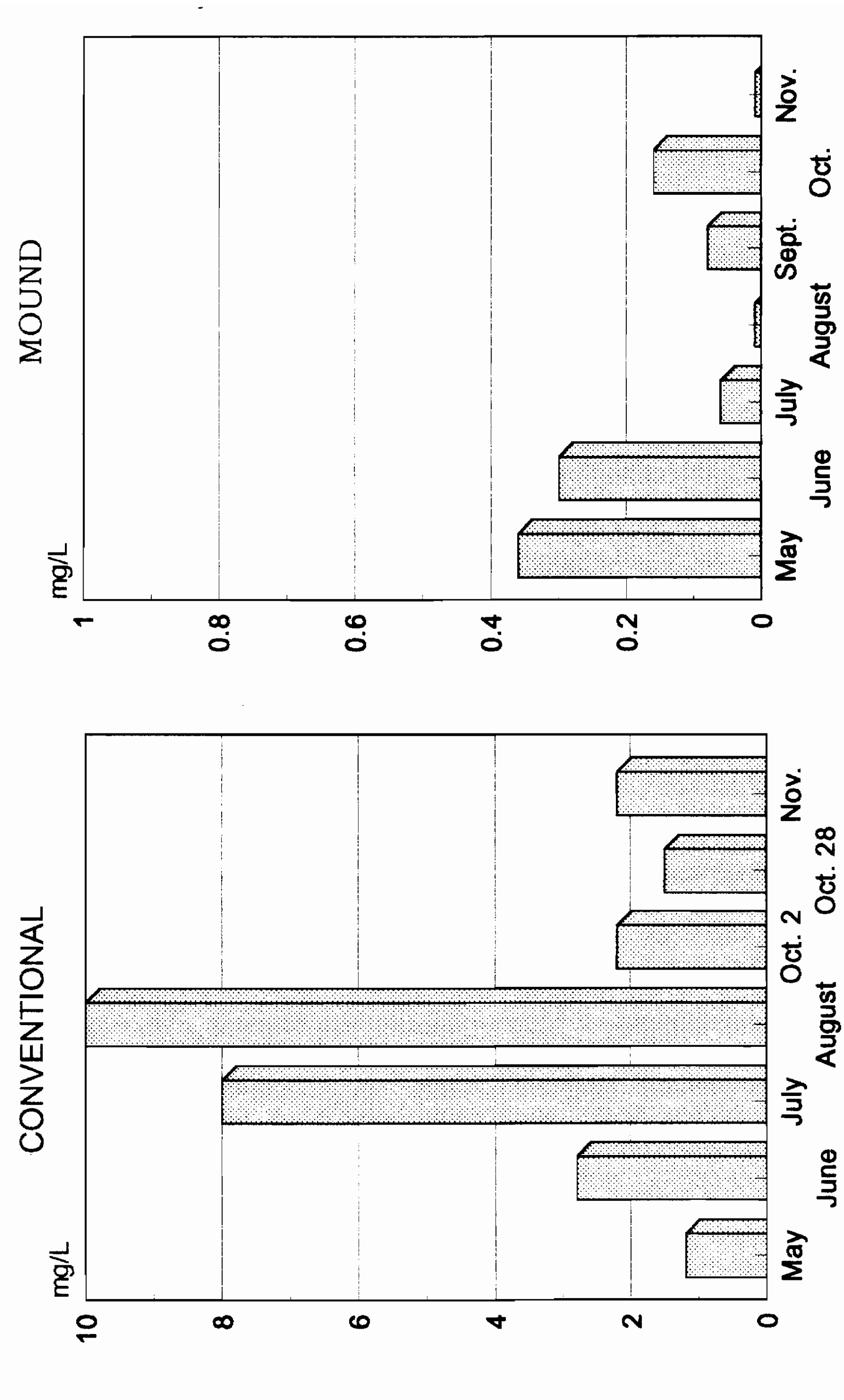



Figure 39. Dissolved inorganic phosphorus concentrations immediately adjacent to the two drainfields. Charts show the Kriging value of the center point of the drainfields. Concentrations are given in mg/L.

to the mound was less than 0.2 mg/L. Significant differences ($p \leq 0.001$; $N=40$) were found in DIP concentrations between the two drainfields. This significant difference was most likely due to the higher phosphorus adsorption capacity of the mound drainfield compared with the conventional drainfield.

In terms of the inorganic nitrogen species, no significant differences ($p=0.963$; $N=32$) were found in ammonium concentrations between the two drainfields while both bath houses were operational. Significant differences ($p=0.007$; $N=24$) were found in ammonium concentrations once the bath houses closed down in the fall. Ammonium was the major inorganic nitrogen species at the mound drainfield, and averaged 90% of the DIN. On the other hand, ammonium represented on average between 47-56% of the DIN in the conventional drainfield. Significant differences ($p=0.001$; $N=56$) were found in nitrate during the study between the two drainfields. While significant differences ($p=0.01$; $N=32$) were measured in nitrite concentrations between the two drainfields during June-August, no significant differences ($p=0.698$; $N=24$) were found during October-November.

Nitrification was not inhibited by pH or temperature at the two sites. Significance differences ($p<0.008$; $N=70$) were found in pH between the two sites, before, during and after bath house operation. The average pH at the conventional site (7.13) was a little higher than the pH at the mound site (6.61). No significance differences ($p \leq 0.001$; $N=70$) was found in ground water temperature between the two sites. The temperature showed temporal variation, ranging between 8-23°C. Significant differences ($p=0.006$; $N=14$) were measured in DO between the two sites in the region immediately adjacent to the drainfield, before bath house operation. During this time, the DO oxygen at the conventional drainfield was one order of magnitude higher than at the mound site. After June, no significant differences ($p>0.213$; $N=54$) were measured in DO between the two drainfield.

Significant differences ($p=0.01$; $N=14$) were measured in salinity before bath house operation. The average salinity at the mound site was 1.921 psu, while at the conventional site it was 0.262 psu. No significant differences ($p=0.219$; $N=40$) were found during bath house operation, and the average salinity for the two sites was 3.544 psu. Even though no significant differences ($p=0.064$; $N=20$) were found in the salinities between the two drainfields, once bath house closure occurred in the fall, the salinities had different behaviors at the two sites. The salinity at the mound drainfield did not show appreciable changes after July. In contrast, the average salinities in the conventional drainfield decreased in half of the October values by November.

In summary, the conventional and the mound sites displayed similar ground water quality characteristics with respect to DIN and DIP concentrations prior to the beginning of the season. Once the drainfields were operational, the mound system showed less phosphorus, but relatively similar nitrogen ground water contamination when compared to the conventional drainfield.

# 6.0 BACTERIOLOGICAL CHARACTERISTICS OF GROUND WATER

## 6.1 Introduction

There is considerable controversy regarding what type of indicator should be used to ensure marine water quality. A number of investigators have pointed out a number of deficiencies the traditional coliform group possesses to accurately identify fecal pollution. Some of these deficiencies are: fecal coliforms are not specific to the source of fecal contamination; they can persist, or even grow in aquatic environments; and they are prey to microbial grazers. In addition, research focused in the traditional culture methods have expressed concern over recoverability of the organisms (Kator, 1992). Rhodes et al. (1983) stated that *E. coli* can be sublethally injured after exposure to the environment, and resuscitation methods would not recover some of these bacteria, which would bias the results if enumeration methods are used. Given that this debate is not yet resolved, the techniques and indicator used during the study were the traditional ones. In addition, even though the MPN (Most Probable Number) method is less precise than the plate count, to be consistent with the enumeration method used by the State of Virginia, the MPN method was employed.

## 6.2 Methodology

Individual samples for microbial analysis were collected with a peristaltic pump using sterilized tubing for each well. Samples were stored in autoclaved bottles, placed on ice, and returned within eight hours to The Virginia Tech Coastal Ground Water Research Laboratory (Eastern Shore Station). Fecal coliform group density was determined by a standard fecal coliform MPN test using A-1 medium (APHA, 1992; Method 9921-E). Fluorogenic crystals MUG (4-methylumbelliferyl-β-D-glucuronide) were added to the A-1 medium for quick detection of *Escherichia coli* (*E. coli*) (Kilian and Bulow, 1976; Hansen and Yourassowsky, 1984). The API 20E System was utilized for confirmation purposes and determination of false positives (bioMérieux Vitek, 1992). For a comprehensive analysis of the fecal contamination indicator methods, the reader is referred to Kator (1992).

### 6.3 Results

Densities of *E. coli* by sampling period are presented in Table A12 for the conventional site, and in Table A21 for the mound site, in Appendix A. Concentrations of *E. coli* measured during January-March were considered as reference values for the conventional site. Temporal and spatial variation were observed in *E. coli* concentrations in the distal half section of the drainfield. From the four wells in the distal half section of the drainfield, only wells (1-2) (Figure 9) showed recurrent high *E. coli* concentrations during bath house operations. Concentrations of *E. coli* in wells (1-2) increased from ≤2 MPN/100 ml during (January-March), to a maximum of ≥16,000 MPN/100 ml during bath house operations. After the season finished, the *E. coli* counts decreased in these two wells. During October, concentrations were 20 MPN/100 ml, and increased during November to 500 and 50 MPN/100 ml for wells 1 and 2, respectively. A 3.45 cm rain fell during the two days prior to the October sampling, and a 1.85 cm rain fell the day of the sampling. In the other two wells of the distal half section of the drainfield, wells (3-4), almost no appreciable increase in *E. coli* concentration was observed during the study. The maximum concentration measured in these two wells was 23 MPN/100 ml. The remaining upland wells did not show any difference from reference values during the study, and had an average value of <2 MPN/100 ml. Due to problems with the incubator, October 2 samples were lost.

Concentrations of *E. coli* were one order of magnitude higher at 0.5 m below WT, than at 1.0 m below WT in the piezometer set (Figure 10), during August-September. At 0.5 m below WT, *E. coli* concentrations were ≥1,600 and ≥ 16,000 MPN/100 ml for August, and September, respectively. It is most probable that *E. coli* concentrations were the same during the two periods, but only dilutions to the 0.1 ml limit were run during August. Alternatively, at 1.0 m below WT, *E. coli* concentrations were between 500-900 MPN/100 ml for August, and between 1100-5000 MPN/100 ml for September. During August, equal *E. coli* concentrations were measured at 0.5 and 1.0 m below WT, only immediately adjacent to the drainfield. During October, average *E. coli* concentrations decreased below 20 MPN/100 ml at 0.5 and 1.0 m below WT, most probably due to the rain. By November, no differences were found in *E. coli* concentrations between 0.5 and 1.0 m below WT (900 MPN/100 ml), immediately adjacent to the drainfield. On the other hand, at 1 and 2 m from the drainfield (PZA and PZB), *E. coli* concentrations were, one to two orders of magnitude higher at 1.0 m than at 0.5 m below. At 1.0 m below WT, *E. coli* concentrations were 1600 and 130 MPN/100 ml, for 1 and 2 m from the drainfield, respectively, and decreased to 170 and 2 MPN/100 ml at 0.5 m below WT.

Elevated *E. coli* concentrations were measured twice at distances greater than 40 meters from the conventional drainfield. The measured values were 80 MPN/100 ml, during July in beach well B2 (Figure 13a), and 130 MPN/100 ml in PZD1 (Figure 9), during August. Positive *E. coli* identifications were made in some of the beach wells, but these were below 8 MPN/100 ml. On average, *E. coli* concentrations were below 2 MPN/100 ml.

The mound system did not show, on average, ground water fecal contamination. After the first month of the mound usage, two of the first tier wells (1-3) (Figure 8) measured 80 and 300 MPN/100 ml. On the second month, only one of the first tier wells, Well 6, measured 30 MPN/100 ml. After the second month, all upland wells measured below 2 MPN/100 ml. The well in the middle of the mound measured 50 MPN/100 ml in September, and less than 2 MPN/100 ml during October and November. Only on one occasion did a beach well show an *E. coli* concentration higher than 8 MPN/100 ml. During September, beach well B2 (Figure 11a) approximately 40 m from the drainfield measured 23 MPN/100 ml.

Other Gram-negative bacteria, able to produce gas in A-1 medium but incapable of hydrolyzing the MUG, identified with the API 20 E system, were: *Enterobacter cloacae, Enterobacter sakazakii, Klebsiella pneumoniae, Klebsiella oxytoca, Klebsiella ssp.*, and *Kluyvera ssp.*.

### 6.4 Discussion

The behavior of *E. coli* found in the first tier wells at the conventional drainfield was in agreement with the nutrient and dye data. Only the two most distal wells at the drainfield edge showed high recurrent *E. coli* concentrations. Even though the nutrient and dye data between wells (1-2) and 3 was very similar, higher fecal contamination was found in the first two wells. If every hydraulic parameter was considered identical for the three wells, the bacterial reduction under the drainfield at well 3 was greater than under wells (1-2). Two possible factors which could have enhanced the greater *E. coli* removal under the drainfield at well 3, were: the higher clay-silt soil content between 0.5-1.0 m below ground level, and/or the biomat had a higher removal efficiency in that region. Similar behavior between *E. coli* and nutrient data was observed in well 4. The *E. coli* concentration increased from below 2 MPN/100 ml during the first week of August, to 13 MPN/100 ml during the second week of August, and to 23 MPN/100 ml during the first week of September. The DIN and DIP increased from 46.42 mg/L and 0.89 mg/L during the first week of August to 67.05 mg/L and 1.62 mg/L by the first week of September, respectively.

Concentrations of *E. coli* at 0.5 and 1.0 m below WT immediately adjacent to the drainfield were in agreement with other studies in similar settings (Reneau and Pettry, 1975). Reneau and Pettry (1975) concluded that *E. coli* moved predominantly in a horizontal direction, and it would not move into the aquifer because of restrictive soil layers. Ziebell et al. (1975) measured a two-fold decrease in fecal coliform bacteria at 0.34 m below a trench compared with samples collected at 0.34 m lateral to the trench. These characteristics were observed during bath house usage in the piezometer set where the *E. coli* concentration at 0.5 m below WT was one order of magnitude higher than at 1.0 m below WT.

Rainfall affects bacterial retention by lowering ionic concentrations and increasing infiltration rates (Gerba and Bitton, 1984). Even though little is known about direct surface interaction between soil components and microbes *in situ* (Stotzky, 1985), Pethica (1961) stated the absorption processes were charge related. This relationship between rain and bacteria absorption was probably seen during the last week of October. During this sampling period, the average *E. coli* concentrations in the distal section of the drainfield, as well as, in the piezometers, were below 20 MPN/100 ml. During November, the *E. coli* concentration increased in six of the eight monitoring well devices. DeWalle et al. (1980) measured that coliform leaching coincided with periods of heavy rainfall. Thus, during the 5.3 cm rain, the sediments were flushed with rain water desorbing the *E. coli*. Two possible reasons why the rain effect was seen only during this sampling and not before, were: 1) this was the only sampling taken immediately after a rainfall; 2) there was no wastewater input into the soil system, and therefore, there was no replenishment of *E. coli*. By November, even though the bath house was closed, *E. coli* concentrations increased. If adequate environmental conditions exist, *E. coli* can survive long periods of time. Simmons (personal communication) measured that *E. coli* survived for more than seven months with levels of >1,600 MPN/100 ml in deer feces located on the soil surface in shade next to a marsh area. Gerba et al. (1975) cited work were *E. coli* survival in soil was up to three and a half months. Reddy et al. (1981), calculated from literature data a first-order die-off rate constant for fecal coliforms of 1.14 day$^{-1}$. Thus, it is possible that the *E. coli* increases during November was due to *E. coli* found under the drainfield. Hagedorn et al. (1978) studied bacterial transport with antibiotic-resistant strains of *E. coli* from a gravel pit. They measured that elevated water tables, created after periods of rainfall, intercepted and carried the organisms by saturated flow. During the 5.3 cm rain, the WT was 0.4 m higher than the WT during November. Thus, it is possible that as the WT decreased, it carried *E. coli* organisms to lower parts of the aquifer, where they were measured during November.

A vast number of studies exist in the literature which referred to transport of bacteria through the solum. Travel distances from 1 to 830 m in different soils with different land application of domestic wastewater are cited (Hagedorn et al., 1981). On two occasions, relatively high *E. coli* concentrations were measured at distances greater than 40 meters from the conventional drainfield. During the first week of July, beach well B2, approximately 60 m form the drainfield, showed 80 MPN/100 ml. Beach well B2 had a salinity value (16.856 ppt) approximately half the salinity of any of the other beach wells during the sampling period. In addition, its nitrate concentration (1.27 mg/L) was one order of magnitude higher than any of the other beach wells. The second occasion, PZD1 approximately 45 m from the drainfield, measured 130 MPN/100 ml, during the second week of August. Almost no difference in salinity was measured between PZD1 (4.293 ppt), the water used in the bath house (4.072 ppt), or the salinity in the distal wells (3.965 ppt) of the drainfield for the first week of August. A 5.82 cm rain felt six days before the sampling day. Hagedorn et al. (1978) measured peaks of bacterial numbers at increasing distance from the pit only after periods of precipitation. On another study, Rahe et al. (1978) concluded that the antibiotic-resistant *E. coli* traveled in regions in the solum which contained a large volume of micropores which produce rapid saturated flow rates. Similar conclusions could be made from these two measurements of *E. coli* concentrations with distance from the drainfield. Possible preferential path flows, enhanced by rain effects, contributed to the *E. coli* existence with distance from the drainfield.

Research by Converse and Tyler (1985) monitored the performance of 40 mound systems in Wisconsin. On average, the mound system achieved excellent purification of the wastewater, and only in very few occasions, showed leakage under extremely wet weather. The mound at the Assateague site also showed excellent performance of purification of the wastewater. In the chapter Physical and Chemical Characteristics of Ground Water it was determined that there was approximately 819 toilet flushes per day. Considering conservative values of 70 g/day/person of average fecal weight, $10^6$ coliform cells per gram of feces (Literature Review, 2.3), and an average of 400 fecal flushes per day, a total input of $28.7\times 10^9$ coliforms cell per day were introduced to the soil. After the first month of mound usage only two wells showed *E. coli* concentrations below 300 MPN/100 ml, and only one well measured 30 MPN/100 ml after the second month. From the first day the mound was operational, wastewater input loading rates were very close to and even over the design value. Given that the greatest bacterial removal takes place at the soil surface where the biological mat occurs, and this mat takes some time to develop, it is most likely that the initial *E. coli* concentrations measured were during the biomat development. In addition, the first time a soil volume is wetted, the soil particles relocate and possible preferential flow develops until the soil takes a well-established structure. Thus, once the biomat was formed and/or the soil configuration was uniform, even though there were high *E. coli* loadings, the mound drainfield performance in wastewater purification was very good.

The different gram-negative bacteria, other than *E. coli,* identified in the ground water are widely distributed in humans and animals, as well as, in nature. For example, the main strains of *Klebsiellae* that occur in soil and water are mainly of the two species *K. terrigena* and *K. planticola*. The problem is that without a specific isolation procedure, these species cannot be distinguished form *K. pneumoniae*, or, *E. cloacae* which is the most frequently isolated *Enterobacter* species found in human and animal feces. It is also found in water, sewage, soil and meat (Krieg and Holt, 1984). Given that these gram-negative bacteria were not found immediately adjacent to the drainfields, and the adequate study of these bacteria was beyond the scope of the project, it was impossible to conclude anything with regard to their contribution in the fecal contamination of the ground and surface waters.

## 7.0 GENERAL SUMMARY AND DISCUSSION

During the year 1993, wastewater effluent from two absorption systems were studied in the Assateague Island National Seashore Park. The study focused on the wastewater effluents from a mound and a conventional OSWDS, located on Parking Lot #1 and #2, respectively. The study duration was from January through November. The research objectives were to determine: 1) the potential for nutrient loadings to coastal waters by way of the ground water, and 2) the potential for ground water to transport fecal coliforms.

The study approach was subdivided into three areas: geohydrology, microbiology, and water quality. Geohydrology included the setting of well transects, well logging, and the study of ground water hydrodynamics. Microbiology included the determination of *E. coli* concentrations. Finally, water quality included the study of chemical and physical characteristics of ground and surface waters.

The study was singular in its kind, not only due to the comparison of two different type of drainfield under similar conditions, but also, due to the peculiar characteristics the study site had. First, the study site was an island. The island width at the conventional site was approximately 200 m and at the mound site it was approximately 130 m. Second, the water table fluctuated due to tides. Third, the coastal beach sediments had high hydraulic conductivities. Fourth, there were high water tables. Finally, there was seasonal effect in the OSWDS operation. Both OSWDS were used at almost full capacity during three to four months of the year.

The average daily wastewater input to each of the drainfields was 10,800 L. This translates, using literature values, to a daily input of 1.8 to 2.8 kg of total nitrogen, 0.2 to 0.5 kg of total phosphorus, and approximately $3\times10^{10}$ coliforms. Average loading values were, for the conventional system, 65 L/day/$m^2$, and for the mound, 47 L/day/$m^2$. On average, the conventional drainfield was over-loaded by approximately 33%, and the mound was under-loaded by approximately 4%.

The two sites showed variations in DIN and DIP during drainfield operation compared with off-season values. The DIN immediately adjacent to the conventional drainfield showed a good relationship with bath house operation. The DIN increased from reference value once the bath house operational and further decreased once the bath house closed down. On the other hand, the DIN in the ground water immediately adjacent to the mound drainfield increased while bath house was operational but it not decreased once the bath house was closed down.

Like the DIN, the DIP immediately adjacent to the conventional drainfield showed a good relationship with bath house operation. The DIP in the drainfield region increased during the season and decreased to once the season ended. In contrast, the DIP immediately adjacent to the mound drainfield did not show significant differences from off-season values.

High ground water DIN concentrations compared with reference values were measured approximately 60 m toward the ocean and 30 m toward Toms Cove from the conventional drainfield. Likewise, at the mound site elevated DIN concentrations compared with reference were measured at 18 m for the drainfield.

The existence of preferential flow patterns from the drainfields to surface waters was acknowledged. Analytical solutions of two advection-dispersion equations were used to determine the average linear ground water velocities. These velocities were greater than those calculated employing the fresh water heads reaffirming the possibility of preferential flow paths from the septic fields to the surface waters.

The three main mechanisms that reduce nitrogen in the soil environment are: denitrification, volatilization of free ammonia, and plant assimilation. Given that a nitrogen mass balance was not performed in either drainfield, it was difficult to accurately conclude which of the two drainfields had a superior performance in the reduction of nitrogen to the ground and surface waters. Based on previous research, under similar working conditions, the mound system would have a superior performance in the nitrogen reduction to the ground and surface waters than the conventional drainfield.

Elevated *E. coli* concentrations were measured in the ground water immediately adjacent to the conventional drainfield. Concentrations of *E. coli* ≥16,000 MPN/100 ml were measured up to 2 m from the drainfield. On one occasion *E. coli* concentration measured 130 MPN/100 ml at a distance greater than 40 m form the drainfield. On the other hand, the mound did not show, on average, ground water fecal contamination. After the second month of drainfield operation, all the immediately adjacent wells to the drainfield measured below 2 MPN/100 ml.

In summary the study conclusions were:

. The septic systems were sources of DIN to the ground water.

. Only the conventional drainfield showed DIP loadings to the ground water.

. Potential loadings to the surface waters occurred only for DIN.

. Only the conventional drainfield showed high level of *E. coli* contamination to the ground water.

. No potential fecal contamination to the coastal waters was detected using *E. coli* as an indicator.

. The mound drainfield, overall, showed a superior performance in the wastewater treatment than the conventional drainfield.

## 7.1 Management Recommendations to the Park Service

In order to assure adequate soil absorption performance, average loading values must be maintained below suggested limits, and an adequate maintenance program must be followed. Dosing and resting is recommended as a good management technique to restore the capacity of the absorption system (Otis, 1984). In addition, resting has been demonstrated to increase the regeneration of phosphorus soil sorption sites (Sawhney and Starr, 1977). Resting is performed by the park service given that the drainfield is used only between four to five months of the year. The average loading values were not very different from the design values. The conventional drainfield was, on average, overloaded by 33%. Thus, if proper loading values were desired, then a bigger drainfield must be constructed. The remaining alternative to support an adequate soil adsorption performance are; septic tank and drainfield maintenance. It is recommended that each septic tank be inspected regularly and pumped when the sludge depth reaches, or exceeds, 1/4 of the total liquid depth (Cotteral et al., 1969). Drainfield maintenance also requires regular inspections to detect failures in its performance. The mound was constructed on 1993, and during its monitoring, no signs were observed which indicated that maintenance was required. In contrast, the conventional system did show signs which suggested that maintenance was required. A number of factors contribute to soil adsorption system failure. Two of them are; distribution pipes clogging, and broken pipes or modification of soil structure by heavy vehicles driven on the surface soil. It was recommended to the park service that the drainfield be inspected for any malfunction and a replacement drainfield be constructed if needed.

Given the possibility of the conventional drainfield having a malfunction due to structural damage, the study results are very limited to completely support the conclusion that a conventional drainfield does not work properly under the tested operational conditions. Nevertheless, if a new conventional drainfield was constructed, the mound drainfield would still probably have a superior performance. This assumption is based on the results of other monitoring studies done on these two types of soil adsorption systems (University of Wisconsin, 1978; Hagedorn et al., 1981; Converse and Tyler, 1984; Reneau et al., 1989).

Given the controversy that the actual indicators used to determine fecal contamination, certain precautions must be taken even though the reasons, but also a setback distance must be regulated between the drainfield and surface waters to minimize euthrophication.

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# APPENDIX A. RAW DATA

Table A1. Surficial upland sediments for the immediately adjacent region to the conventional drainfield. Grain size are expressed as mass ratios of the total mass. Organic matter is expressed as percenage weight loss from combustion of the dried sample. BD denotes below detection limits.

| Sample | Gravel | Sand | Silt | Caly | Organic Matter % |
|---|---|---|---|---|---|
| W1 surface | 3.64 | 95.71 | 0.12 | 0.53 | 0.3 |
| W1 0.50 | 0.00 | 99.63 | 0.28 | 0.09 | 0.2 |
| W1 1.00 | 0.26 | 99.22 | 0.23 | 0.29 | BD |
| W1 1.25 | 5.51 | 94.28 | 0.12 | 0.09 | 0.3 |
| W1 1.50 | 1.62 | 97.54 | 0.33 | 0.51 | 0.2 |
| W2 surface | 0.00 | 37.49 | 0.00 | 62.50 | 0.3 |
| W2 0.50 | 0.00 | 100.00 | 0.00 | 0.00 | 0.2 |
| W2 0.75 | 0.08 | 99.47 | 0.00 | 0.45 | 0.2 |
| W2 1.00 | 0.08 | 99.29 | 0.00 | 0.62 | 0.1 |
| W2 1.25 | 0.00 | 98.61 | 0.51 | 0.87 | 0.1 |
| W2 1.40 | 0.00 | 98.57 | 1.43 | 0.00 | 0.1 |
| W3 surface | 0.34 | 97.40 | 1.61 | 0.64 | 1.1 |
| W3 0.50 | 0.00 | 98.67 | 0.00 | 1.33 | 0.2 |
| W3 0.75 | 0.00 | 97.62 | 0.72 | 1.65 | 0.3 |
| W3 1.00 | 0.40 | 97.90 | 0.46 | 1.24 | 0.2 |
| W3 1.25 | 0.00 | 99.53 | 0.00 | 0.47 | 0.1 |
| W3 1.50 | 0.04 | 98.72 | 0.00 | 1.24 | 0.1 |
| W3 1.70 | 5.78 | 92.79 | 1.42 | 0.00 | 0.2 |
| W4 surface | 0.71 | 97.05 | 1.09 | 1.14 | 0.4 |
| W4 0.50 | 0.00 | 99.34 | 0.31 | 0.35 | 0.1 |
| W4 0.75 | 0.06 | 99.21 | 0.31 | 0.42 | 0.2 |
| W4 1.00 | 1.88 | 97.39 | 0.00 | 0.73 | 0.1 |
| W4 1.25 | 0.00 | 99.68 | 0.05 | 0.27 | 0.1 |
| W4 1.50 | 0.06 | 99.20 | 0.02 | 0.71 | 0.1 |
| W5 surface | 0.29 | 98.64 | 0.31 | 0.76 | 0.4 |
| W5 0.50 | 0.00 | 99.05 | 0.12 | 0.83 | 0.2 |
| W5 0.75 | 0.00 | 98.42 | 0.53 | 1.05 | 0.2 |
| W5 1.00 | 0.00 | 99.75 | 0.00 | 0.25 | 0.1 |
| W5 1.25 | 0.00 | 99.37 | 0.08 | 0.55 | 0.1 |
| W5 1.50 | 0.00 | 99.81 | 0.00 | 0.19 | 0.1 |
| W6 surface | 0.22 | 97.79 | 0.99 | 1.00 | 0.6 |
| W6 0.50 | 0.00 | 98.82 | 0.31 | 0.87 | 0.1 |
| W6 0.75 | 0.00 | 99.26 | 0.00 | 0.74 | 0.1 |
| W6 1.00 | 0.00 | 99.45 | 0.02 | 0.53 | 0.1 |
| W6 1.25 | 0.07 | 99.13 | 0.09 | 0.71 | 0.1 |

Table A2. Surficial upland sediments for the immediately adjacent region to the mound drainfield. Grain size are expressed as mass ratios of the total mass. Organic matter is expressed as percenage weight loss from combustion of the dried sample. MD denotes missing data.

| Sample | Gravel | Sand | Silt | Caly | Organic Matter % |
|---|---|---|---|---|---|
| W1 0.60 | 0.00 | 93.14 | 5.25 | 1.61 | MD |
| W1 0.75 | 0.00 | 99.63 | 0.14 | 0.23 | 0.2 |
| W1 1.00 | 3.75 | 96.14 | 0.00 | 0.11 | 0.2 |
| W1 1.25 | 0.00 | 99.31 | 0.00 | 0.69 | 0.1 |
| W2 0.75 | 0.00 | 96.05 | 2.44 | 1.51 | 2.7 |
| W2 1.00 | 0.00 | 99.48 | 0.14 | 0.38 | 0.2 |
| W2 1.25 | 0.00 | 99.42 | 0.10 | 0.48 | 0.2 |
| W4 0.75 | 0.00 | 92.23 | 4.77 | 3.0 | 0.7 |
| W4 1.00 | 0.00 | 99.43 | 0.00 | 0.57 | 0.3 |
| W4 1.25 | 0.00 | 99.71 | 0.00 | 0.29 | 0.2 |
| W5 0.75 | 0.00 | 100.00 | 0.00 | 0.00 | 0.2 |
| W5 1.00 | 0.00 | 99.35 | 0.00 | 0.65 | 0.2 |
| W6 0.75 | 0.14 | 99.53 | 0.33 | 0.00 | 0.8 |
| W6 1.00 | 0.00 | 99.35 | 0.36 | 0.29 | 0.5 |
| W6 1.25 | 0.00 | 99.60 | 0.00 | 0.40 | 0.3 |

Table A3. Rhodamine WT 20% concentrations by sampling period at the conventional site. Concentrations are expressed as part per billion. BD denotes below detection limits. MD denotes missing data.

| Sample | 9/5 | 9/16 | 9/28 | 10/2 | 10/28 | 11/19 |
|---|---|---|---|---|---|---|
| 1 | 1.4 | 7.8 | 11.2 | 13.9 | 29.0 | 54.0 |
| PZ 1 | 1.8 | 18.0 | 11.4 | 13.4 | 8.5 | 4.7 |
| PZA 0.5 | 1.4 | 20.0 | 4.7 | 6.3 | 12.3 | 15.8 |
| PZA 1.0 | 1.5 | 1.5 | 14.2 | 11.7 | 8.2 | 11.7 |
| PZB 0.5 | 1.5 | 13.8 | 5.7 | 6.6 | 5.2 | 2.6 |
| PZB 1.0 | 1.5 | 1.4 | 11.7 | 8.8 | 15.8 | 3.8 |
| 2 | 1.3 | 9.1 | 19.9 | 25.9 | 34.0 | 18.0 |
| 3 | 0.9 | 18.0 | 31.6 | 20.0 | 9.2 | 12.3 |
| 4 | 0.5 | 0.9 | 1.9 | 6.9 | 1.9 | 6.9 |
| PZ 4 | BD | BD | 0.1 | 0.1 | 0.1 | 0.1 |
| 5 | BD | BD | 0.3 | 0.1 | 0.3 | 0.1 |
| 6 | 0.1 | BD | 0.1 | 0.1 | 0.1 | 0.1 |
| 7 | BD | 0.3 | 0.4 | 0.4 | 0.4 | 0.4 |
| 8 | BD | BD | 0.1 | 0.1 | 0.1 | 0.1 |
| 9 | BD | 0.2 | 0.2 | 0.4 | 0.2 | 0.4 |
| 10 | MD | 0.1 | 0.8 | 0.9 | 0.8 | 0.9 |
| 11 | MD | MD | MD | 0.9 | MD | 0.9 |
| 12 | MD | 0.6 | 0.9 | 0.9 | 0.9 | 0.9 |
| 13 | MD | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 |
| D1 | MD | 0.3 | 0.4 | 0.4 | 0.4 | 0.4 |
| PZ D1 | 0.3 | 0.5 | 0.7 | 0.8 | 0.7 | 0.8 |
| D2 | MD | 0.6 | 0.7 | 0.7 | 0.7 | 0.7 |
| D3 | 1.2 | 16.0 | 0.6 | 0.5 | 0.6 | 0.5 |

Table A4. Ground water and surface water ammonium concentration by samping date for the conventional septic system site. Concentrations are expressed in mg•liter$^{-1}$ as $N\text{-}NH_4$. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 0.07 | 0.01 | 4.80 | 30.54 | 55.70 | 117.56 | 79.65 | 99.88 | 54.90 | 15.62 | 12.00 |
| 2 | 0.04 | 0.03 | 0.01 | 4.74 | 11.72 | 36.19 | MD | MD | 9.52 | 17.86 | 3.59 |
| 3 | 0.26 | 0.04 | 0.73 | 6.62 | 0.17 | 64.11 | MD | MD | 44.14 | 9.80 | 19.48 |
| 4 | 0.05 | 0.02 | 0.01 | 0.07 | 0.02 | 0.03 | 0.02 | 17.46 | MD | 0.02 | 0.27 |
| 5 | 0.03 | 0.01 | 0.01 | 0.04 | 0.01 | 0.03 | MD | MD | MD | 0.03 | 0.03 |
| 6 | 0.04 | 0.02 | 0.01 | 0.01 | 0.02 | 0.03 | MD | MD | MD | 0.03 | 0.02 |
| 7 | MD | 0.02 | BD | 0.04 | 0.01 | 0.02 | MD | MD | MD | 0.03 | 4.34 |
| 8 | MD | 0.05 | BD | 0.01 | MD | 0.03 | MD | MD | MD | 0.02 | 0.03 |
| 9 | MD | 0.04 | 0.02 | 0.01 | MD | 0.03 | MD | MD | MD | 0.03 | 0.03 |
| 10 | MD | MD | 0.02 | 0.06 | 0.03 | 0.03 | MD | MD | 0.05 | 0.04 | 0.05 |
| 11 | MD | MD | MD | MD | 0.03 | 0.05 | MD | MD | 0.06 | 0.02 | 0.03 |
| 12 | MD | MD | MD | MD | 0.01 | 0.01 | MD | MD | MD | 0.01 | 0.03 |
| 13 | MD | MD | MD | MD | 0.02 | 0.04 | MD | MD | MD | 0.03 | 0.04 |
| D1 | MD | MD | MD | MD | MD | 0.04 | 0.11 | 0.11 | 0.03 | 0.18 | 0.69 |
| D2 | MD | MD | MD | MD | MD | 0.12 | 0.20 | MD | MD | 0.22 | 0.21 |
| D3 | MD | MD | MD | MD | MD | 3.10 | MD | MD | MD | 0.16 | 0.04 |
| Ocean | 0.02 | 0.01 | 0.01 | 0.03 | 0.01 | 0.02 | 0.12 | 0.01 | MD | 0.03 | 0.02 |
| Bath House | MD | MD | 5.92 | 5.46 | 5.22 | MD | MD | MD | MD | MD | MD |

Table A5. Ground water and surface water nitrate concentration by samping date for the conventional septic system site. Concentrations are expressed in mg•liter$^{-1}$ as N-$NO_3$. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1.30 | 1.70 | 1.50 | 6.30 | 1.40 | 5.30 | 0.18 | BD | 13.72 | 16.26 | 19.78 |
| 2 | 0.46 | 0.85 | 2.59 | 41.24 | 10.11 | 29.45 | MD | MD | 34.51 | 1.57 | 12.67 |
| 3 | 0.25 | 0.94 | 6.57 | 34.47 | 38.80 | 16.47 | MD | MD | 26.31 | 11.23 | 12.02 |
| 4 | 0.57 | 0.96 | 0.28 | 1.40 | 3.39 | 46.21 | 20.01 | 47.41 | 62.63 | 41.82 | 16.05 |
| 5 | 1.04 | 2.29 | 0.35 | 1.99 | 2.26 | 1.85 | MD | MD | MD | 0.61 | 3.87 |
| 6 | 1.41 | 1.24 | 1.28 | 1.42 | 2.09 | 2.29 | MD | MD | MD | 1.18 | 2.26 |
| 7 | MD | 0.50 | 0.24 | 0.16 | 3.38 | 3.39 | MD | MD | MD | 0.36 | 0.36 |
| 8 | MD | 1.47 | 1.60 | 1.99 | MD | 1.58 | MD | MD | MD | 1.18 | 4.59 |
| 9 | MD | 0.17 | 0.05 | 0.27 | MD | 0.71 | MD | MD | MD | 0.23 | 0.21 |
| 10 | MD | 0.52 | 0.10 | 0.16 | 0.79 | 0.57 | MD | MD | 0.13 | 6.85 | 0.26 |
| 11 | MD | MD | MD | MD | 0.39 | 0.99 | MD | MD | 0.14 | 0.11 | 0.09 |
| 12 | MD | MD | MD | MD | 3.19 | 3.16 | MD | MD | MD | 2.20 | 19.24 |
| 13 | MD | MD | MD | MD | 1.17 | 0.19 | MD | MD | MD | 0.23 | 2.05 |
| D1 | MD | MD | MD | MD | MD | 0.43 | BD | MD | BD | BD | BD |
| D2 | MD | MD | MD | MD | MD | 0.43 | BD | MD | MD | BD | 0.01 |
| D3 | MD | MD | MD | MD | MD | 58.65 | MD | MD | 79.06 | 0.81 | 0.50 |
| Ocean | 0.09 | 0.06 | 0.05 | 0.01 | 0.01 | 0.02 | BD | 0.01 | MD | 0.01 | 0.01 |
| Bath House | MD | MD | 0.06 | 0.07 | 0.01 | MD | MD | MD | MD | MD | MD |

Table A6. Ground water and surface water nitrIte concentration by samping date for the conventional septic system site. Concentrations are expressed in mg•liter$^{-1}$ as N-$NO_2$. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 0.02 | BD | 0.09 | 0.01 | 0.09 | 0.91 | 0.08 | BD | 0.40 | 1.00 | 0.05 |
| 2 | 0.01 | BD | BD | 0.66 | 0.91 | 4.93 | MD | MD | 0.84 | 0.10 | 0.04 |
| 3 | 0.01 | 0.01 | 0.02 | 0.01 | 0.89 | 1.78 | MD | MD | 0.84 | 0.01 | BD |
| 4 | BD | BD | BD | BD | 0.02 | 0.18 | 0.08 | 2.18 | 8.14 | 0.01 | 0.04 |
| 5 | 0.01 | BD | BD | BD | BD | 0.01 | MD | MD | MD | 0.02 | BD |
| 6 | BD | BD | BD | BD | BD | BD | MD | MD | MD | BD | BD |
| 7 | MD | BD | BD | BD | BD | BD | MD | MD | MD | BD | 0.33 |
| 8 | MD | BD | BD | BD | BD | BD | MD | MD | MD | BD | BD |
| 9 | MD | BD | BD | BD | BD | BD | MD | MD | MD | 0.01 | BD |
| 10 | MD | BD | BD | BD | BD | 0.04 | MD | MD | 0.05 | 0.13 | BD |
| 11 | MD | MD | MD | MD | MD | 0.04 | MD | MD | 0.05 | 0.03 | 0.03 |
| 12 | MD | MD | MD | MD | MD | BD | MD | MD | MD | 0.24 | BD |
| 13 | MD | MD | MD | MD | MD | BD | MD | MD | MD | BD | BD |
| D1 | MD | MD | MD | MD | MD | 0.01 | BD | MD | BD | BD | BD |
| D2 | MD | MD | MD | MD | MD | 0.01 | BD | MD | MD | BD | BD |
| D3 | MD | MD | MD | MD | MD | 0.53 | MD | MD | 0.26 | 0.01 | 0.05 |
| Ocean | BD | BD | BD | BD | BD | BD | BD | BD | MD | BD | BD |
| Bath House | MD | MD | 0.01 | 0.01 | BD | MD | MD | MD | MD | MD | MD |

Table A7. Ground water and surface water dissolved inorganic phosphorus concentration by samping date for the conventional septic system site. Concentrations are expressed in mg•liter$^{-1}$ as P-$PO_4$. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1.22 | 1.08 | 2.05 | 5.98 | 10.36 | 23.54 | 18.84 | 15.09 | 2.36 | 1.16 | 0.38 |
| 2 | 0.10 | 0.41 | 0.25 | 1.71 | 9.85 | 7.80 | MD | MD | 2.88 | 1.80 | 4.31 |
| 3 | 4.14 | 2.11 | 2.11 | 2.92 | 10.54 | 7.32 | MD | MD | 1.85 | 2.24 | 2.03 |
| 4 | 0.30 | 0.23 | 0.26 | 0.08 | 0.06 | 0.89 | 0.95 | 1.05 | 1.62 | 0.99 | 1.99 |
| 5 | 0.09 | 0.05 | 0.04 | 0.04 | 0.05 | 0.05 | MD | MD | MD | 0.02 | 0.04 |
| 6 | 0.12 | 0.09 | 0.05 | 0.05 | 0.07 | 0.06 | MD | MD | MD | 0.05 | 0.07 |
| 7 | MD | 0.02 | BD | 0.01 | 0.01 | 0.02 | MD | MD | MD | 0.04 | 0.16 |
| 8 | MD | 0.12 | 0.07 | 0.08 | MD | 0.05 | MD | MD | MD | 0.05 | 0.05 |
| 9 | MD | 0.09 | 0.04 | 0.02 | MD | 0.02 | MD | MD | MD | 0.03 | 0.05 |
| 10 | MD | 0.05 | 0.08 | 0.16 | 0.23 | 0.13 | MD | MD | 0.12 | 0.08 | 0.05 |
| 11 | MD | MD | MD | MD | 0.12 | 0.08 | MD | MD | 0.06 | 0.03 | 0.05 |
| 12 | MD | MD | MD | MD | 0.08 | 0.12 | MD | MD | MD | 0.12 | 0.15 |
| 13 | MD | MD | MD | MD | 0.04 | 0.03 | MD | MD | MD | 0.03 | 0.05 |
| D1 | MD | MD | MD | MD | MD | 0.13 | 0.12 | MD | 0.01 | 0.04 | 0.07 |
| D2 | MD | MD | MD | MD | MD | 0.01 | 0.02 | MD | MD | 0.02 | 0.04 |
| D3 | MD | MD | MD | MD | MD | 4.13 | MD | MD | 2.34 | 0.02 | 1.50 |
| Ocean | 0.02 | BD | BD | BD | 0.01 | 0.01 | 0.06 | 0.01 | MD | 0.02 | 0.01 |
| Bath House | MD | MD | 0.05 | 0.04 | 0.05 | MD | MD | MD | MD | MD | MD |

Table A8. Ground water and surface water dissolved oxygen concentration by samping date for the conventional septic system site. Concentrations are expressed in mg•liter$^{-1}$. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 7.9 | 5.6 | 1.1 | 1.0 | 1.5 | BD | BD | BS | 1.7 | 0.1 | 3.4 |
| 2 | 10.6 | 8.8 | 4.5 | 2.0 | MD | 0.9 | MD | MD | 3.2 | 0.2 | 1.6 |
| 3 | 2.1 | 6.1 | 1.3 | 0.7 | 2.0 | 2.3 | MD | MD | 1.6 | 0.5 | 0.7 |
| 4 | 9.8 | 9.8 | MD | 6.3 | MD | 2.9 | 1.4 | 1.0 | 4.9 | 4.5 | 3.0 |
| 5 | 11.4 | 10.7 | 7.7 | 7.4 | 6.8 | 1.8 | MD | MD | MD | 1.4 | 6.5 |
| 6 | 10.6 | 9.9 | 8.1 | 7.2 | MD | 13.0 | MD | MD | MD | 3.7 | 5.3 |
| 7 | MD | 11.2 | 8.3 | 6.2 | 7.7 | 5.6 | MD | MD | MD | 4.4 | 1.6 |
| 8 | MD | 11.6 | 7.7 | 7.4 | MD | 10.3 | MD | MD | MD | 4.8 | 8.1 |
| 9 | MD | 10.1 | 8.1 | 7.2 | MD | 2.7 | MD | MD | MD | 1.3 | 2.7 |
| 10 | MD | 3.44 | 3.2 | 1.7 | 2.8 | 2.8 | MD | MD | 1.4 | 0.6 | MD |
| 11 | MD | MD | MD | MD | 4.1 | 2.0 | MD | MD | 8.3 | 1.0 | 1.3 |
| 12 | MD | MD | MD | MD | 3.0 | 4.2 | MD | MD | MD | 1.1 | 6.9 |
| 13 | MD | MD | MD | MD | 7.4 | 1.5 | MD | MD | MD | 1.4 | 3.1 |
| D1 | MD | MD | MD | MD | MD | 2.7 | 1.3 | MD | 1.7 | 0.3 | MD |
| D2 | MD | MD | MD | MD | MD | 0.4 | 0.8 | MD | MD | 0.5 | 0.8 |
| D3 | MD | MD | MD | MD | MD | 1.3 | MD | MD | 1.0 | 1.3 | MD |
| Ocean | 10.2 | 10.3 | 8.5 | 8.7 | 8.0 | 16.8 | 6.4 | MD | MD | MD | MD |
| Bath House | MD | MD | 1.8 | 1.0 | 2.1 | MD | MD | MD | MD | MD | MD |

Table A9. Ground water and surface water salinity by samping date for the conventional septic system site. Concentrations are expressed in practical salinity units. MD denotes missing data. BD denotes below detection limits.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 0.213 | 0.292 | 3.674 | 3.611 | 4.209 | 4.356 | 4.239 | MD | MD | 4.160 | 3.825 |
| 2 | 0.636 | 0.138 | 0.623 | 3.538 | 4.083 | 4.186 | MD | MD | MD | 4.019 | 1.160 |
| 3 | 0.337 | 0.148 | 1.562 | 3.808 | 4.226 | 4.360 | MD | MD | MD | 3.987 | 3.632 |
| 4 | 0.110 | 0.121 | 0.095 | 0.122 | 0.276 | 2.958 | 2.486 | 2.621 | MD | 3.987 | 1.209 |
| 5 | 0.131 | 0.208 | 0.133 | 0.141 | 0.134 | 0.134 | MD | MD | MD | 0.203 | 0.205 |
| 6 | 0.148 | 0.190 | 0.188 | 0.119 | 0.103 | 0.060 | MD | MD | MD | 0.095 | 0.156 |
| 7 | MD | 0.078 | 0.159 | 0.086 | 0.818 | 4.117 | MD | MD | MD | 0.742 | 3.984 |
| 8 | MD | 0.123 | 0.131 | 0.137 | MD | 0.107 | MD | MD | MD | 0.098 | 0.151 |
| 9 | MD | 0.106 | 0.100 | 0.114 | MD | 0.082 | MD | MD | MD | 0.120 | 0.128 |
| 10 | MD | 0.697 | 0.120 | 0.114 | 0.093 | 0.072 | MD | MD | 0.141 | 0.177 | 0.372 |
| 11 | MD | MD | MD | MD | 0.117 | 0.112 | MD | MD | 0.161 | 0.239 | 0.192 |
| 12 | MD | MD | MD | MD | 0.897 | 4.327 | MD | MD | MD | 3.545 | 1.214 |
| 13 | MD | MD | MD | MD | 0.615 | 1.003 | MD | MD | MD | 0.843 | 0.715 |
| D1 | MD | MD | MD | MD | MD | 4.448 | 4.514 | MD | 4.334 | 4.538 | 4.120 |
| D2 | MD | MD | MD | MD | MD | 4.484 | 4.410 | MD | MD | 4.322 | 4.202 |
| D3 | MD | MD | MD | MD | MD | 4.358 | MD | MD | MD | 0.143 | 0.095 |
| Ocean | 30.868 | 29.500 | 29.498 | 30.766 | 31.035 | 31.953 | 30.465 | 31.275 | MD | 31.235 | 30.690 |
| Bath House | MD | MD | 3.990 | 4.090 | 4.136 | MD | MD | MD | MD | MD | MD |

Table A10. Ground water and surface pH by sampling date for the conventional septic system site. MD denotes missing data.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 8.27 | 7.74 | 7.55 | 7.53 | 7.35 | 7.05 | 7.44 | 7.52 | 7.42 | 7.11 | 7.33 |
| 2 | 7.19 | 7.08 | 7.32 | 7.21 | 7.18 | 6.97 | MD | MD | 7.13 | 7.05 | 7.35 |
| 3 | 8.31 | 7.45 | 7.20 | 7.31 | 7.09 | 7.15 | MD | MD | 7.02 | 6.96 | 7.05 |
| 4 | 7.74 | 7.25 | 7.40 | 7.10 | 6.84 | 6.84 | 7.15 | 7.10 | 7.91 | 7.14 | 7.35 |
| 5 | 7.57 | 7.33 | 7.50 | 7.46 | 7.30 | 7.14 | MD | MD | MD | 7.12 | 7.16 |
| 6 | 7.97 | 7.31 | 7.44 | 7.36 | 7.20 | 7.19 | MD | MD | MD | 7.34 | 7.40 |
| 7 | MD | 7.08 | 7.26 | 7.07 | 6.87 | 7.39 | MD | MD | MD | 7.59 | 7.74 |
| 8 | MD | 7.90 | 7.68 | 7.52 | MD | 7.36 | MD | MD | MD | 7.52 | 7.60 |
| 9 | MD | 6.94 | 7.17 | 6.91 | MD | 6.86 | MD | MD | MD | 7.31 | 7.20 |
| 10 | MD | 6.23 | 7.00 | 6.62 | 6.81 | 6.72 | MD | MD | 6.94 | 6.59 | 6.36 |
| 11 | MD | MD | MD | MD | 7.02 | 6.77 | MD | MD | 6.90 | 6.75 | 6.68 |
| 12 | MD | MD | MD | MD | 7.03 | 7.08 | MD | MD | MD | 7.25 | 7.47 |
| 13 | MD | MD | MD | MD | 7.24 | 6.96 | MD | MD | MD | 7.22 | 7.26 |
| D1 | MD | MD | MD | MD | MD | 7.69 | 7.94 | MD | 8.00 | 7.72 | 7.67 |
| D2 | MD | MD | MD | MD | MD | 7.60 | 8.12 | MD | MD | 7.75 | 8.08 |
| D3 | MD | MD | MD | MD | MD | 6.98 | MD | MD | 7.20 | 7.68 | 7.76 |
| Ocean | 8.13 | 7.57 | 7.98 | 7.82 | 7.87 | 8.01 | 8.15 | 8.09 | MD | 7.98 | 8.01 |
| Bath House | MD | MD | 7.98 | 7.96 | 7.94 | MD | MD | MD | MD | MD | MD |

Table A11. Ground water and surface temperature by sampling date for the conventional septic system site. Temperature is given in degrees Celsius. MD denotes missing data.

| Well | January | March | May | June | July | August 5 | September | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 8.5 | 8.3 | 13.4 | 18.0 | 24.0 | 25.1 | 24.5 | 18.0 | 15.0 |
| 2 | 8.5 | 7.9 | 13.4 | 18.0 | 24.3 | 26.2 | MD | 17.5 | 15.0 |
| 3 | 10.0 | 8.9 | 13.2 | 18.0 | 22.0 | 23.0 | MD | 18.0 | 15.5 |
| 4 | 9.5 | 8.0 | 13.9 | 18.0 | 23.0 | 24.5 | 25.0 | 17.5 | 14.5 |
| 5 | 9.0 | 7.9 | 13.2 | 18.0 | 21.5 | 22.8 | MD | 18.0 | 15.0 |
| 6 | 8.5 | 7.6 | 13.6 | 18.0 | 22.7 | 23.0 | MD | 15.5 | 14.5 |
| 7 | MD | 8.3 | 12.6 | 17.0 | 19.9 | 21.9 | MD | 17.0 | 14.0 |
| 8 | MD | 8.5 | 14.1 | 18.0 | MD | 23.1 | MD | 17.5 | 15.0 |
| 9 | MD | 8.2 | 13.4 | 18.0 | MD | 23.0 | MD | 18.0 | 14.5 |
| 10 | MD | 8.0 | 13.3 | 17.0 | 21.5 | 23.3 | MD | 17.5 | 15.0 |
| 11 | MD | MD | MD | MD | MD | 24.3 | MD | 19.0 | 16.0 |
| 12 | MD | MD | MD | MD | 20.1 | 21.5 | MD | 18.0 | 14.8 |
| 13 | MD | MD | MD | MD | 20.1 | 20.1 | MD | 17.5 | 15.0 |
| D1 | MD | MD | MD | MD | MD | 19.4 | MD | MD | 16.0 |
| D2 | MD | MD | MD | MD | MD | 20.0 | MD | 18.0 | 16.0 |
| D3 | MD | MD | MD | MD | MD | 25.3 | MD | 18.0 | 14.5 |
| Ocean | 6.0 | 5.6 | 11.0 | 19.0 | MD | MD | MD | MD | 12.5 |
| Bath House | MD | MD | MD | MD | MD | MD | MD | MD | MD |

Table A12. Ground water and surface *Escherichia coli* density by sampling date for the conventional septic system site. Densities are expressed as Most Probable Number per 100 milliliters. MD denotes missing data.

| Well | January | March | May | June | July | August 5 | August 12 | September | October 28 | November |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 2 | <2 | 50 | ≥1600 | ≥1600 | ≥1600 | ≥1600 | ≥16000 | 20 | 900 |
| 2 | 2 | <2 | <2 | 17 | ≥1600 | ≥1600 | MD | MD | 20 | 50 |
| 3 | <2 | <2 | <2 | <2 | 11 | 11 | MD | MD | 13 | 7 |
| 4 | <2 | <2 | <2 | <2 | <2 | <2 | 13 | 23 | 11 | MD |
| 5 | 4 | <2 | <2 | <2 | <2 | 2 | MD | MD | MD | MD |
| 6 | <2 | <2 | <2 | <2 | <2 | <2 | MD | MD | MD | MD |
| 7 | MD | <2 | <2 | <2 | <2 | <2 | MD | MD | MD | MD |
| 8 | MD | <2 | <2 | <2 | MD | MD | MD | MD | MD | MD |
| 9 | MD | <2 | <2 | <2 | MD | MD | MD | MD | MD | MD |
| 10 | MD | 2 | 8 | 2 | <2 | <2 | MD | MD | MD | MD |
| 11 | MD | MD | MD | MD | <2 | MD | MD | MD | MD | MD |
| 12 | MD | MD | MD | MD | <2 | <2 | MD | MD | MD | MD |
| 13 | MD | MD | MD | MD | <2 | MD | MD | MD | MD | MD |
| D1 | MD | MD | MD | MD | MD | <2 | <2 | MD | MD | MD |
| D2 | MD | MD | MD | MD | MD | <2 | <2 | MD | MD | MD |
| D3 | MD | MD | MD | MD | MD | <2 | MD | MD | <2 | <2 |
| Ocean | <2 | <2 | <2 | <2 | 2 | 2 | <2 | 4 | <2 | MD |
| Bath House | MD | MD | <2 | <2 | <2 | MD | MD | MD | MD | MD |

Table A13. Nutrients at the piezometer set on the conventional site. Ammonium as $N\text{-}NH_4$, nitrate as $N\text{-}NO_3$, nitirte as $N\text{-}NO_2$ and dissolved inorganic phosphorus (DIP) as $P\text{-}PO_4$. Nutrient concentrations. are given in mg•liter$^{-1}$. MD denotes missing data. BD denotes below detection limits.

| | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|
| Ammonium | | | | | |
| 1 | 79.65 | 99.88 | 54.90 | 15.62 | 12.00 |
| PZ1 | 86.34 | 95.91 | 96.61 | 26.32 | 22.60 |
| PZA 0.5 | 91.05 | 116.15 | 21.82 | 0.11 | 5.43 |
| PZA 1.0 | 97.12 | 85.11 | 121.32 | 12.43 | 11.15 |
| PZB 0.5 | 27.50 | 103.05 | 0.73 | 0.04 | 0.03 |
| PZB 1.0 | 91.05 | 98.77 | 109.23 | 0.07 | 0.02 |
| PZ 4 | 0.04 | 0.04 | 0.05 | 1.46 | 1.14 |
| PZ D1 | 3.87 | 0.39 | 0.12 | 0.02 | 0.03 |
| Nitrate | | | | | |
| 1 | 0.18 | BD | 13.72 | 16.26 | 19.78 |
| PZ1 | 0.23 | BD | BD | 0.03 | 0.50 |
| PZA 0.5 | 0.20 | BD | 8.22 | 21.36 | 24.87 |
| PZA 1.0 | 0.23 | BD | 0.01 | 5.14 | 24.13 |
| PZB 0.5 | 16.20 | BD | 56.61 | 12.91 | 0.98 |
| PZB 1.0 | 2.04 | 0.01 | 0.01 | 12.13 | 12.78 |
| PZ 4 | 2.31 | 1.32 | 1.08 | 0.32 | 3.55 |
| PZ D1 | 0.10 | 1.76 | 0.08 | 0.05 | 0.76 |
| Nitrite | | | | | |
| 1 | 0.08 | BD | 0.40 | 1.00 | 0.05 |
| PZ1 | 0.02 | BD | BD | 0.01 | BD |
| PZA 0.5 | 0.01 | BD | 0.04 | 0.10 | BD |
| PZA 1.0 | 0.04 | BD | 0.02 | 3.45 | 0.02 |
| PZB 0.5 | 0.09 | BD | 0.30 | 0.20 | 0.58 |
| PZB 1.0 | 0.01 | BD | 0.02 | 4.03 | 0.81 |
| PZ 4 | BD | BD | BD | BD | BD |
| PZ D1 | 0.04 | 0.05 | BD | 0.02 | 0.04 |
| DIP | | | | | |
| 1 | 18.84 | 15.09 | 2.36 | 1.16 | 0.38 |
| PZ1 | 29.99 | 24.98 | 18.07 | 0.58 | 0.95 |
| PZA 0.5 | 26.43 | 16.92 | 30.69 | 0.12 | 0.35 |
| PZA 1.0 | 29.99 | 25.53 | 12.41 | 0.41 | 0.28 |
| PZB 0.5 | 12.37 | 17.11 | 27.09 | 1.17 | 1.28 |
| PZB 1.0 | 26.49 | 26.99 | 12.92 | 0.43 | 0.52 |
| PZ 4 | 0.06 | 0.03 | BD | 0.03 | 0.03 |
| PZ D1 | 0.43 | 0.07 | 0.04 | 0.11 | 0.13 |

Table A14. Dissolved oxygen, ground water pH, salinity and *Escherichia coli* data for the piezometer set on the conventional site. Dissloved oxygen is given in mg per liter. *E. coli* in MPN.100 mls. Salinity in part per thousand. MD denotes missing data.

| | August 12 | September | October 2 | October 28 | November |
|---|---|---|---|---|---|
| D. Oxygen | | | | | |
| 1 | BD | BD | 1.7 | 0.1 | 3.4 |
| PZ1 | 1.0 | 0.3 | MD | BD | 0.5 |
| PZA 0.5 | MD | MD | MD | 0.4 | MD |
| PZA 1.0 | BD | BD | MD | MD | MD |
| PZB 0.5 | 0.6 | 0.1 | MD | 1.1 | 0.6 |
| PZB 1.0 | 0.3 | BD | MD | 0.8 | 0.8 |
| PZ 4 | 3.4 | 1.5 | 3.6 | 4.5 | MD |
| PZ D1 | MD | 0.9 | 1.5 | 2.5 | 1.3 |
| pH | | | | | |
| 1 | 7.4 | 7.5 | 7.4 | 7.1 | 7.3 |
| PZ1 | 7.6 | 7.8 | 8.8 | 7.2 | 7.6 |
| PZA 0.5 | 7.4 | 7.7 | 7.8 | 7.5 | 7.6 |
| PZA 1.0 | 7.5 | 7.6 | 7.9 | 7.6 | 7.5 |
| PZB 0.5 | 7.3 | 7.6 | 7.9 | 7.7 | 8.3 |
| PZB 1.0 | 7.5 | 7.6 | 7.5 | 7.7 | 7.9 |
| PZ 4 | 7.2 | 7.1 | 7.3 | 7.2 | 7.4 |
| PZ D1 | 6.8 | 6.7 | 7.9 | 7.9 | 6.8 |
| Salinity | | | | | |
| 1 | 4.239 | MD | MD | 4.160 | 3.825 |
| PZ1 | 3.960 | MD | MD | 4.160 | 3.852 |
| PZA 0.5 | 4.439 | MD | MD | 3.959 | 2.889 |
| PZA 1.0 | 4.355 | MD | MD | 4.113 | 3.434 |
| PZB 0.5 | 3.849 | MD | MD | 1.441 | 0.198 |
| PZB 1.0 | 4.386 | MD | MD | 4.022 | 1.507 |
| PZ 4 | 0.393 | 1.724 | MD | 0.200 | 0.169 |
| PZ D1 | 4.293 | 17.068 | 8.931 | 3.975 | 4.489 |
| E. coli | | | | | |
| 1 | ≥1600 | ≥16000 | MD | 20 | 900 |
| PZ1 | ≥1600 | 1100 | MD | 130 | 900 |
| PZA 0.5 | ≥1600 | ≥16000 | MD | <20 | 170 |
| PZA 1.0 | 500 | 5000 | MD | 20 | 1600 |
| PZB 0.5 | ≥1600 | ≥16000 | MD | <20 | 2 |
| PZB 1.0 | 900 | 1300 | MD | <20 | 130 |
| PZ 4 | <2 | <2 | MD | MD | MD |
| PZ D1 | 130 | 30 | MD | 9 | <2 |

Table A15. Ground water quality for beach wells on the conventional septic system site. Nutrient concentrations, and dissolved oxygen (DO) are given in mg per liter. Salinity is expressed as practical salinity units. *Escherichia coli* densities are expressed as MPN per 100 milliliters. MD denotes missing data. BD denotes below detection limit.

| Well | $NH_4$ ($N\text{-}NH_4$) | $NO_3$ ($N\text{-}NO_3$) | $NO_2$ ($N\text{-}NO_2$) | DIP ($P\text{-}PO_4$) | DO | pH | Salinity | E.coli |
|---|---|---|---|---|---|---|---|---|
| July | | | | | | | | |
| B1 | 0.05 | 0.28 | BD | 0.04 | 3.3 | 7.06 | 27.611 | 4 |
| B2 | 0.01 | 1.27 | BD | 0.05 | 3.5 | 7.21 | 16.856 | 80 |
| B3 | 0.01 | 0.35 | BD | 0.05 | 5.9 | 7.34 | 29.590 | <2 |
| Ocean | 0.01 | 0.01 | BD | 0.01 | 8.0 | 7.87 | 31.035 | 2 |
| August 5 | | | | | | | | |
| B1 | 0.02 | 0.36 | BD | 0.06 | 10.8 | 7.64 | 32.089 | 4 |
| B2 | 0.01 | 0.59 | BD | 0.06 | 7.6 | 7.69 | 32.643 | <2 |
| B3 | 0.01 | 0.36 | BD | 0.08 | 11.3 | 7.70 | 32.387 | <2 |
| B4 | 0.48 | 0.11 | BD | 0.15 | 2.2 | 7.20 | 11.236 | 2 |
| B5 | 0.34 | 0.13 | BD | 0.07 | 1.6 | 6.95 | 15.806 | <2 |
| B6 | 0.06 | 2.20 | 0.03 | 0.04 | 3.6 | 7.32 | 9.728 | 2 |
| Ocean | 0.02 | 0.02 | BD | 0.01 | 16.9 | 8.01 | 31.953 | 2 |
| August 12 | | | | | | | | |
| B1 | 0.01 | 0.26 | BD | 0.05 | 5.89 | 7.84 | 31.190 | 2 |
| B2 | 2.46 | 1.13 | 0.01 | 0.12 | 0.86 | 6.97 | 14.018 | 2 |
| B3 | 0.30 | 0.44 | BD | 0.10 | BD | 6.97 | 18.736 | <2 |
| B4 | 0.11 | 2.21 | BD | 0.04 | 2.48 | 7.04 | 26.068 | 2 |
| B5 | 0.36 | 0.35 | BD | 0.06 | 0.94 | 7.00 | 16.527 | 8 |
| Ocean | 0.12 | BD | BD | 0.01 | 6.45 | 8.15 | 30.465 | <2 |
| September 16 | | | | | | | | |
| B1 | 1.73 | 0.06 | BD | 0.17 | MD | 7.18 | 12.346 | <2 |
| B2 | 1.31 | 0.09 | BD | 0.18 | MD | 6.96 | 9.431 | <2 |
| B3 | 1.10 | 0.02 | BD | 0.08 | MD | 6.98 | 11.579 | <2 |
| B4 | 0.69 | 0.03 | BD | 0.09 | MD | 6.97 | 13.309 | <2 |
| B5 | 0.64 | 0.21 | BD | 0.08 | MD | 7.00 | 14.106 | <2 |
| B7 | 0.01 | 0.29 | BD | 0.08 | MD | 7.45 | 30.071 | <2 |
| B8 | 0.03 | 0.94 | BD | 0.05 | MD | 7.57 | 31.071 | <2 |
| B9 | 0.02 | 0.28 | BD | 0.03 | MD | 7.61 | 31.001 | <2 |
| B10 | 0.02 | 2.63 | BD | 0.04 | MD | 7.62 | 31.156 | <2 |
| Ocean | 0.03 | BD | BD | 0.02 | MD | 7.89 | 30.932 | <2 |

Table A16. Ground water quality for beach wells on the conventional septic system site. Nutrient concentrations, and dissolved oxygen (DO) are given in mg per liter. Salinity is expressed as practical salinity units. *Escherichia coli* densities are expressed as MPN per 100 milliliters. MD denotes missing data. BD denotes below detection limit.

| Well | $NH_4$ ($N\text{-}NH_4$) | $NO_3$ ($N\text{-}NO_3$) | $NO_2$ ($N\text{-}NO_2$) | DIP ($P\text{-}PO_4$) | DO | pH | Salinity | *E.coli* |
|---|---|---|---|---|---|---|---|---|
| October 2 | | | | | | | | |
| BT1 | 0.03 | 5.48 | 0.08 | 0.04 | 3.2 | 7.15 | 0.625 | MD |
| BT2 | 1.24 | 1.30 | 0.01 | 0.04 | MD | 6.73 | 4.722 | MD |
| BT3 | 0.03 | 6.13 | BD | 0.02 | 2.6 | 7.52 | 0.300 | MD |
| B1 | 0.91 | 0.01 | BD | 0.17 | 6.7 | 6.97 | 13.487 | MD |
| B2 | 0.10 | 0.28 | BD | 0.04 | 3.5 | 6.74 | 31.078 | MD |
| October 28 | | | | | | | | |
| B1 | 0.03 | 15.17 | BD | 0.04 | MD | 7.70 | 0.488 | 4 |
| B2 | 0.05 | 0.16 | BD | 0.06 | MD | 7.53 | 30.111 | 4 |
| B3 | 0.03 | 0.06 | BD | 0.06 | MD | 7.89 | 30.989 | 2 |
| B4 | 0.02 | 0.02 | BD | 0.06 | MD | 7.93 | 31.019 | 4 |
| B5 | 0.03 | 0.54 | BD | 0.08 | MD | 7.69 | 23.450 | <2 |
| B6 | 0.04 | 0.11 | BD | 0.10 | MD | 7.43 | 29.548 | <2 |
| Ocean | 0.03 | 0.01 | BD | 0.02 | MD | 7.98 | 31.235 | <2 |
| November 19 | | | | | | | | |
| B1 | 0.75 | 0.01 | BD | 0.24 | BD | 6.96 | 4.558 | <2 |
| B2 | 0.02 | 6.07 | BD | 0.04 | 6.14 | 7.55 | 0.516 | 2 |
| B3 | 0.31 | 0.01 | BD | 0.30 | BD | 7.13 | 0.391 | <2 |
| B4 | 0.03 | 3.66 | BD | 0.01 | 3.18 | 6.96 | 0.418 | <2 |
| B5 | 0.83 | 0.04 | BD | 0.26 | 1.34 | 7.12 | 0.503 | <2 |
| Ocean | 0.02 | 0.01 | BD | 0.01 | MD | 8.01 | 30.690 | MD |

Table A17. Ground water and surface ammonium and nitrate concentrations by sampling date for the mound septic system site. Nutrient concentrations are given in mg per liter. MD denotes missing data. BD denotes below detection limit.

| **Well** | **May** | **June** | **July** | **August** | **September 3** | **September 16** | **October** | **November** |
|---|---|---|---|---|---|---|---|---|
| Ammonium ($N\text{-}NH_4$) | | | | | | | | |
| 1 | 0.15 | 13.39 | 16.83 | 20.54 | MD | 43.04 | 51.76 | 40.06 |
| 2 | 0.27 | MD | 2.96 | 48.82 | MD | 41.74 | 34.48 | 5.90 |
| 3 | 0.32 | 0.78 | 9.03 | 7.81 | MD | 18.88 | 51.38 | 58.55 |
| 4 | 1.72 | 9.19 | 19.63 | 11.23 | MD | MD | 20.96 | 19.79 |
| 5 | 0.08 | MD | 0.54 | 2.11 | 15.20 | 26.84 | 31.79 | 29.55 |
| 6 | 1.75 | 0.90 | 22.45 | 23.44 | 27.00 | 59.15 | 1.78 | 15.96 |
| 7 | MD | MD | MD | 0.29 | 1.27 | 0.68 | 0.94 | 3.14 |
| 8 | MD | MD | MD | MD | MD | 0.04 | 0.27 | 0.24 |
| 9 | MD | MD | MD | MD | MD | 1.25 | 0.78 | 0.66 |
| 10 | MD | MD | MD | MD | MD | 12.34 | 14.02 | 5.93 |
| 11 | MD | MD | MD | MD | MD | 17.31 | 36.33 | 48.87 |
| 12 | MD | MD | MD | MD | MD | 0.87 | 4.42 | 5.16 |
| 13 | MD | MD | MD | MD | MD | 1.08 | 0.23 | 1.04 |
| Top | MD | MD | MD | MD | MD | 39.77 | 41.05 | 24.48 |
| Ocean | MD | MD | 0.01 | 0.02 | 0.01 | 0.03 | 0.03 | 0.02 |
| Nitrate ($N\text{-}NO_3$) | | | | | | | | |
| 1 | 0.05 | 0.01 | BD | BD | MD | 0.01 | 0.12 | 0.08 |
| 2 | 0.01 | 0.02 | BD | BD | MD | 1.43 | 0.93 | 1.98 |
| 3 | BD | 0.03 | BD | BD | MD | BD | BD | BD |
| 4 | 0.01 | 0.45 | 10.37 | 13.12 | MD | MD | BD | 10.84 |
| 5 | 1.66 | 1.47 | 11.09 | 6.68 | 44.67 | 7.67 | 2.13 | 7.87 |
| 6 | 0.01 | 0.11 | BD | BD | BD | BD | BD | 0.57 |
| 7 | MD | MD | MD | BD | BD | 0.01 | BD | BD |
| 8 | MD | MD | MD | MD | MD | MD | BD | 0.02 |
| 9 | MD | MD | MD | MD | MD | 0.06 | BD | BD |
| 10 | MD | MD | MD | MD | MD | 0.40 | BD | 0.51 |
| 11 | MD | MD | MD | MD | MD | 0.01 | BD | BD |
| 12 | MD | MD | MD | MD | MD | BD | BD | BD |
| 13 | MD | MD | MD | MD | MD | 0.02 | BD | BD |
| Top | MD | MD | MD | MD | MD | 0.04 | BD | BD |
| Ocean | MD | MD | BD | 0.02 | 0.01 | BD | 0.02 | 0.01 |

Table A18. Ground water and surface water nitrite and dissolved inorganic phosphorus (DIP) concentrations by sampling date for the mound septic system site. Nutrient concentrations are given in mg per liter. MD denotes missing data. BD denotes below detection limit.

| **Well** | **May** | **June** | **July** | **August** | **September 3** | **September16** | **October** | **November** |
|---|---|---|---|---|---|---|---|---|
| Nitrite ($N\text{-}NO_2$) | | | | | | | | |
| 1 | BD | BD | 0.01 | BD | MD | BD | 0.03 | 0.05 |
| 2 | BD | 0.01 | 0.01 | BD | MD | 0.40 | 0.01 | 0.09 |
| 3 | 0.01 | 0.02 | BD | 0.01 | MD | 0.02 | 0.02 | 0.02 |
| 4 | 0.01 | 0.03 | 1.51 | 0.81 | MD | MD | 0.03 | 0.02 |
| 5 | 0.02 | 0.03 | 0.07 | 0.02 | 0.89 | 0.59 | BD | 0.07 |
| 6 | BD | 0.03 | 0.02 | BD | 0.01 | BD | 0.01 | 0.08 |
| 7 | MD | MD | MD | BD | 0.01 | BD | 0.02 | 0.01 |
| 8 | MD | MD | MD | MD | MD | 0.03 | BD | BD |
| 9 | MD | MD | MD | MD | MD | 0.01 | BD | 0.01 |
| 10 | MD | MD | MD | MD | MD | 0.01 | 0.03 | 0.02 |
| 11 | MD | MD | MD | MD | MD | BD | 0.03 | 0.02 |
| 12 | MD | MD | MD | MD | MD | BD | 0.01 | 0.01 |
| 13 | MD | MD | MD | MD | MD | BD | 0.01 | 0.01 |
| Top | MD | MD | MD | MD | MD | BD | 0.01 | 0.02 |
| Ocean | MD | MD | BD | BD | BD | BD | BD | BD |
| DIP ($P\text{-}PO_4$) | | | | | | | | |
| 1 | 0.14 | 0.11 | 0.03 | BD | MD | BD | 0.18 | BD |
| 2 | 0.50 | 0.69 | 0.31 | BD | MD | 0.02 | 0.10 | 0.11 |
| 3 | 0.13 | 0.47 | 0.05 | BD | MD | 0.08 | 0.15 | 0.01 |
| 4 | 0.66 | 0.12 | 0.01 | 0.01 | MD | MD | 0.19 | 0.01 |
| 5 | 0.14 | 0.11 | 0.04 | 0.01 | 0.04 | 0.06 | 0.03 | 0.05 |
| 6 | 0.47 | 1.49 | 0.74 | 1.75 | 1.73 | 0.20 | 0.75 | 0.90 |
| 7 | MD | MD | MD | 0.22 | 0.23 | 0.03 | 0.16 | 0.23 |
| 8 | MD | MD | MD | MD | MD | 0.20 | 0.05 | 0.12 |
| 9 | MD | MD | MD | MD | MD | 0.27 | 0.02 | 0.40 |
| 10 | MD | MD | MD | MD | MD | 0.04 | 0.22 | 0.25 |
| 11 | MD | MD | MD | MD | MD | 0.01 | 0.22 | BD |
| 12 | MD | MD | MD | MD | MD | 0.02 | 0.08 | BD |
| 13 | MD | MD | MD | MD | MD | 0.01 | 0.06 | BD |
| Top | MD | MD | MD | MD | MD | 0.06 | 0.09 | 0.10 |
| Ocean | MD | MD | 0.01 | 0.01 | 0.01 | 0.02 | 0.01 | 0.01 |

Table A19. Ground water and surface water dissolved oxygen (DO) and temperature by sampling date for the mound septic system site. Disssolved oxygen concentrations are given in mg per liter. Temperature is given in degrees Celsius. MD denotes missing data. BD denotes below detection limit.

| Well | May | June | July | August | September 3 | October | November |
|---|---|---|---|---|---|---|---|
| DO | | | | | | BD | MD |
| 1 | 0.4 | 0.4 | 5.4 | 0.5 | MD | BD | MD |
| 2 | 0.6 | BD | 1.8 | 2.0 | MD | 1.0 | MD |
| 3 | 0.6 | BD | MD | 1.5 | MD | MD | MD |
| 4 | 0.3 | 0.7 | MD | 5.4 | MD | 0.5 | MD |
| 5 | MD | 1.0 | MD | 1.0 | 1.2 | BD | MD |
| 6 | 0.3 | BD | BD | MD | BD | BD | MD |
| 7 | MD | MD | MD | MD | 0.2 | BD | MD |
| 8 | MD | MD | MD | MD | MD | BD | MD |
| 9 | MD | MD | MD | MD | MD | BD | MD |
| 10 | MD | MD | MD | MD | MD | BD | MD |
| 11 | MD | MD | MD | MD | MD | BD | MD |
| 12 | MD | MD | MD | MD | MD | 0.1 | MD |
| 13 | MD | MD | MD | MD | MD | 0.1 | MD |
| Top | MD | MD | MD | MD | MD | BD | MD |
| Ocean | MD | MD | 9.37 | 13.72 | MD | MD | MD |
| Temperature | | | | | | | |
| 1 | 14.5 | 18.0 | 23.4 | MD | MD | MD | 15.0 |
| 2 | MD | 17.0 | 23.0 | MD | MD | 17.5 | 14.0 |
| 3 | MD | 17.0 | 22.1 | MD | MD | 19.0 | 16.0 |
| 4 | 14.2 | 20.0 | MD | MD | MD | 17.0 | 15.0 |
| 5 | MD | 18.0 | 23.7 | MD | 23.0 | 18.0 | 15.5 |
| 6 | 14.0 | 18.0 | 23.0 | MD | 23.5 | 17.0 | 14.4 |
| 7 | MD | MD | MD | MD | 22.0 | 17.0 | 14.5 |
| 8 | MD | MD | MD | MD | MD | 16.0 | 14.0 |
| 9 | MD | MD | MD | MD | MD | 17.5 | 15.0 |
| 10 | MD | MD | MD | MD | MD | 18.5 | 15.5 |
| 11 | MD | MD | MD | MD | MD | 17.5 | 14.5 |
| 12 | MD | MD | MD | MD | MD | 20.5 | 18.8 |
| 13 | MD | MD | MD | MD | MD | 17.0 | 14.5 |
| Top | MD | MD | MD | MD | MD | 18.0 | 16.0 |
| Ocean | 11.0 | 19.0 | MD | MD | MD | MD | 12.5 |

Table A20. Ground water and surface water pH and salinity by sampling date for the mound septic system site. Salinity is given in practical salinity units. MD denotes missing data.

| Well | May | June | July | August | September 3 | September16 | October | November |
|---|---|---|---|---|---|---|---|---|
| pH | | | | | | | | |
| 1 | 7.57 | 7.08 | 6.79 | 6.64 | MD | 6.53 | 6.68 | 6.84 |
| 2 | 6.99 | 7.22 | 6.44 | 6.93 | MD | 6.88 | 6.65 | 6.86 |
| 3 | 6.27 | 7.29 | 6.93 | 6.84 | MD | 6.48 | 6.88 | 6.86 |
| 4 | 6.54 | 6.15 | md | 6.61 | MD | MD | 6.57 | 6.83 |
| 5 | 7.12 | 6.93 | 6.26 | 6.10 | 6.53 | 6.74 | 7.03 | 6.94 |
| 6 | 6.54 | 6.69 | 6.82 | 6.87 | 6.81 | 6.99 | 6.80 | 7.06 |
| 7 | MD | MD | MD | 6.91 | 6.87 | 6.78 | 7.23 | 6.96 |
| 8 | MD | MD | MD | MD | MD | 5.25 | 6.53 | 6.76 |
| 9 | MD | MD | MD | MD | MD | 7.00 | 7.51 | 7.36 |
| 10 | MD | MD | MD | MD | MD | 6.72 | 6.83 | 6.76 |
| 11 | MD | MD | MD | MD | MD | 6.58 | 6.83 | 6.95 |
| 12 | MD | MD | MD | MD | MD | 6.63 | 7.08 | 7.19 |
| 13 | MD | MD | MD | MD | MD | 6.61 | 7.16 | 7.15 |
| Top | MD | MD | MD | MD | MD | 7.04 | 7.02 | 6.85 |
| Ocean | MD | MD | 7.90 | 8.06 | 8.09 | 7.89 | 8.00 | 8.01 |
| Salinity | | | | | | | | |
| 1 | 0.410 | 2.525 | 2.450 | 3.677 | MD | 4.251 | 4.237 | 4.254 |
| 2 | 1.228 | 0.720 | 3.245 | 4.398 | MD | 4.201 | 4.140 | 1.325 |
| 3 | 0.869 | 0.408 | 3.040 | 3.612 | MD | 3.440 | 4.325 | 4.370 |
| 4 | 2.823 | 5.285 | MD | 4.203 | MD | MD | 3.761 | 4.211 |
| 5 | 0.429 | 0.256 | 2.257 | 4.028 | 4.302 | 4.218 | 4.414 | 4.224 |
| 6 | 5.766 | 0.992 | 3.674 | 3.174 | 3.423 | 4.218 | 3.411 | 3.428 |
| 7 | MD | MD | MD | 2.950 | 4.754 | 3.338 | 5.013 | 3.913 |
| 8 | MD | MD | MD | MD | MD | 1.982 | 2.088 | 3.114 |
| 9 | MD | MD | MD | MD | MD | 3.696 | 2.971 | 2.541 |
| 10 | MD | MD | MD | MD | MD | 3.187 | 3.145 | 2.999 |
| 11 | MD | MD | MD | MD | MD | 3.439 | 4.303 | 4.331 |
| 12 | MD | MD | MD | MD | MD | 4.117 | 5.787 | 4.905 |
| 13 | MD | MD | MD | MD | MD | 6.880 | 9.056 | 2.927 |
| Top | MD | MD | MD | MD | MD | 4.183 | 4.228 | 3.882 |
| Ocean | 29.50 | 30.76 | 31.01 | 31.973 | 31.275 | 30.932 | 31.151 | 30.690 |

Table A21. Ground water and surface water *Escherichia coli* densities by sampling date for the mound septic system site. Densities are expressed in MPN per 100 milliliters. MD denotes missing data..

| Well | May | June | July | August | September 3 | September16 | October |
|---|---|---|---|---|---|---|---|
| 1 | <2 | 300 | <2 | <2 | MD | MD | <2 |
| 2 | <2 | <2 | <2 | <2 | MD | MD | <2 |
| 3 | <2 | 80 | <2 | <2 | MD | MD | <2 |
| 4 | 2 | 2 | <2 | <2 | MD | MD | <2 |
| 5 | <2 | <2 | <2 | <2 | <2 | MD | <2 |
| 6 | <2 | 2 | 30 | <2 | <2 | MD | <2 |
| 7 | MD | MD | MD | MD | <2 | MD | <2 |
| 8 | MD | MD | MD | MD | MD | <2 | <2 |
| 9 | MD | MD | MD | MD | MD | <2 | <2 |
| 10 | MD | MD | MD | MD | MD | <2 | <2 |
| 11 | MD | MD | MD | MD | MD | <2 | <2 |
| 12 | MD | MD | MD | MD | MD | <2 | <2 |
| 13 | MD | MD | MD | MD | MD | <2 | <2 |
| Top | MD | MD | MD | MD | MD | 50 | <2 |
| Ocean | MD | MD | <2 | 4 | 4 | MD | MD |

Table A22. Ground water quality data for beach wells for the mound septic system site by sampling period. Nutrient and dissolved oxygen concentrations are given in mg per liter. Salinity is given in practical salinity units. Escherichia coli densities are expressed as MPN per 100 milliliters. BD denotes below detection limits. MD denotes missing data.

| Well | $NH_4$ ($N\text{-}NH_4$) | $NO_3$ ($N\text{-}NO_3$) | $NO_2$ ($N\text{-}NO_2$) | DIP ($P\text{-}PO_4$) | DO | pH | Salinity | *E.coli* |
|---|---|---|---|---|---|---|---|---|
| July | | | | | | | | |
| B1 | BD | 0.12 | 0.03 | 0.03 | 1.2 | 7.21 | 29.772 | <2 |
| B2 | 0.01 | 0.32 | BD | 0.08 | 1.2 | 7.37 | 29.897 | <2 |
| B3 | 0.02 | 0.29 | BD | 0.09 | 1.5 | 7.67 | 29.659 | <2 |
| Ocean | 0.01 | BD | BD | 0.01 | 9.4 | 7.90 | 31.018 | <2 |
| August | | | | | | | | |
| B2 | 0.03 | 0.03 | BD | 0.05 | 8.9 | 7.67 | 31.024 | 2 |
| B3 | 0.03 | 0.06 | BD | 0.04 | 10.7 | 7.83 | 32.165 | 8 |
| B4 | 0.32 | 0.71 | BD | 0.08 | 5.6 | 7.40 | 32.243 | <2 |
| B5 | 0.01 | 0.28 | BD | 0.03 | 6.9 | 7.49 | 32.148 | <2 |
| B6 | 0.03 | MD | BD | 0.06 | 7.9 | 7.66 | 31.965 | <2 |
| Ocean | 0.02 | 0.02 | BD | 0.01 | 13.7 | 8.06 | 31.973 | 4 |
| September | | | | | | | | |
| B1 | 0.10 | 0.02 | BD | 0.04 | MD | 7.01 | 31.407 | <2 |
| B2 | 0.18 | 0.02 | BD | BD | MD | 5.96 | 31.306 | 23 |
| B3 | 0.08 | 0.06 | BD | 0.01 | 2.2 | 6.04 | 31.148 | 2 |
| B4 | 0.03 | 0.32 | BD | 0.02 | 4.9 | 6.30 | 31.619 | 2 |
| B5 | 0.16 | 0.03 | 0.01 | 0.01 | 2.1 | 5.83 | 28.833 | <8 |
| Ocean | 0.01 | 0.01 | BD | 0.01 | MD | 8.09 | 31.275 | 4 |
| October | | | | | | | | |
| B1 | 0.02 | 0.09 | BD | 0.06 | MD | 7.92 | 31.297 | 2 |
| B2 | 0.02 | 0.07 | BD | 0.04 | MD | 7.83 | 31.019 | 2 |
| Ocean | 0.03 | 0.02 | BD | 0.01 | MD | 8.00 | 31.151 | MD |

## APPENDIX B

Samples for dye measurement were taken after a minimum of three volumes of water were removed from wells in order to allow recharge with fresh ground-water. Samples were collected in Nalgene bottles and stored in the dark until analyzed. Samples were filtered with 0.45 mm membrane filters and read in a Turner Fluorometer (Model 10). Some features of the fluorometer are:

. The limit of detectability is under 10 parts per trillion.

. The Fluorometer has a blank suppression, that once set, it holds on all ranges. This allows the instrument to be calibrated to read directly in concentration.

. The sensitivity may be set automatically or manually. The automatic/manual range change allows a total range selection of in excess of 3000:1.

In order to assure an adequate reading the manual operation instructions were followed each time a set of sample was read. Samples were run to determine a practical resolution of 0.1 ppb.