Research Technical Completion Report

INVESTIGATION OF NITROGEN CONTAMINATION OF GROUNDWATER FROM LANDFILL DISPOSAL OF CULL ONIONS

by

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ABSTRACT

Disposal of cull onions in covered landfills has been a suspected source of nitrate contamination of groundwater in the Treasure Valley region of south-western Idaho and eastern Oregon. A field investigation was undertaken which characterized leachate generation with regard to nitrogen composition and transport. Two full-scale operational landfills were instrumented prior to onion disposal to: (1) determine the physical controls on leachate generation, (2) investigate nitrate transport using a conservative anionic tracer, and (3) determine the fate of nitrogen.

Water levels were measured in the landfills using PVC standpipes, and hydraulic conductivity was measured at the base of the landfills using constant head permeameters. Matric suctions were monitored using thermal dissipation sensors at depths of 0.5, 1, 1.5, 2, and 3 m below the landfills to observe changes in water content and hydraulic gradients. A bromide tracer was used to estimate leachate flux. These measurements, and results of an analysis of unsaturated hydraulic properties of the underlying matrix using the RETC method (van Genuchten *et al.*, 1991) were used to estimate vertical flow velocities in the matrix. Leachate samples were obtained using pressure/vacuum samplers placed at 0.5, 1, 2, 3, and 4 m. These samples and water samples taken from within the landfills were analyzed for NO_3-N, NH_4-N and TKN.

Results from 40 months of monitoring indicate that anaerobic conditions were established during the first two to three months of landfill operation and that nitrification was inhibited. It is hypothesized that hydraulic and overburden pressures forced decomposing organic material into sediment pores and fractures, forming a low hydraulic conductivity lining. Water derived from precipitation runoff and from decomposing onions was retained in the landfills because of the low conductivity lining. This "self-lining" retained water in the landfills and restricted leaching in the underlying profiles to unsaturated flow velocities.

Nitrate concentrations measured below the landfills reached 41 mg/l during the initial two months of operation, because the landfills were open and the discarded onions were mostly intact. During this period, nitrification occurred and a small amount of nitrate was leached into the underlying matrix. With the establishment of anaerobic conditions, however, nitrification was inhibited and the concentration of nitrate in the landfills decreased dramatically. Nitrate concentrations below the landfills decreased to less than 0.5 mg/l by the fourth month of landfill operation. Ammonium concentrations increased to a maximum of 390 mg/l over the period of the study, because mineralization of organic nitrogen occurred under both aerobic and anaerobic conditions.

The instrumented cull onion landfills pose a minimal threat to regional groundwater resources, because of: (1) partitioning of nitrogen into organic and ammonium pools, (2) inhibition of nitrification, (3) formation of a low conductivity layer, (4) low rates of leaching, and (5) the relatively large distance to the water table. Continued research is proposed that aims to establish the long-term fate of nitrogen in the landfill systems once they desaturate. In addition, results of this study indicate that long-term nitrification could be mitigated through excavation and land application of the degraded onion waste.

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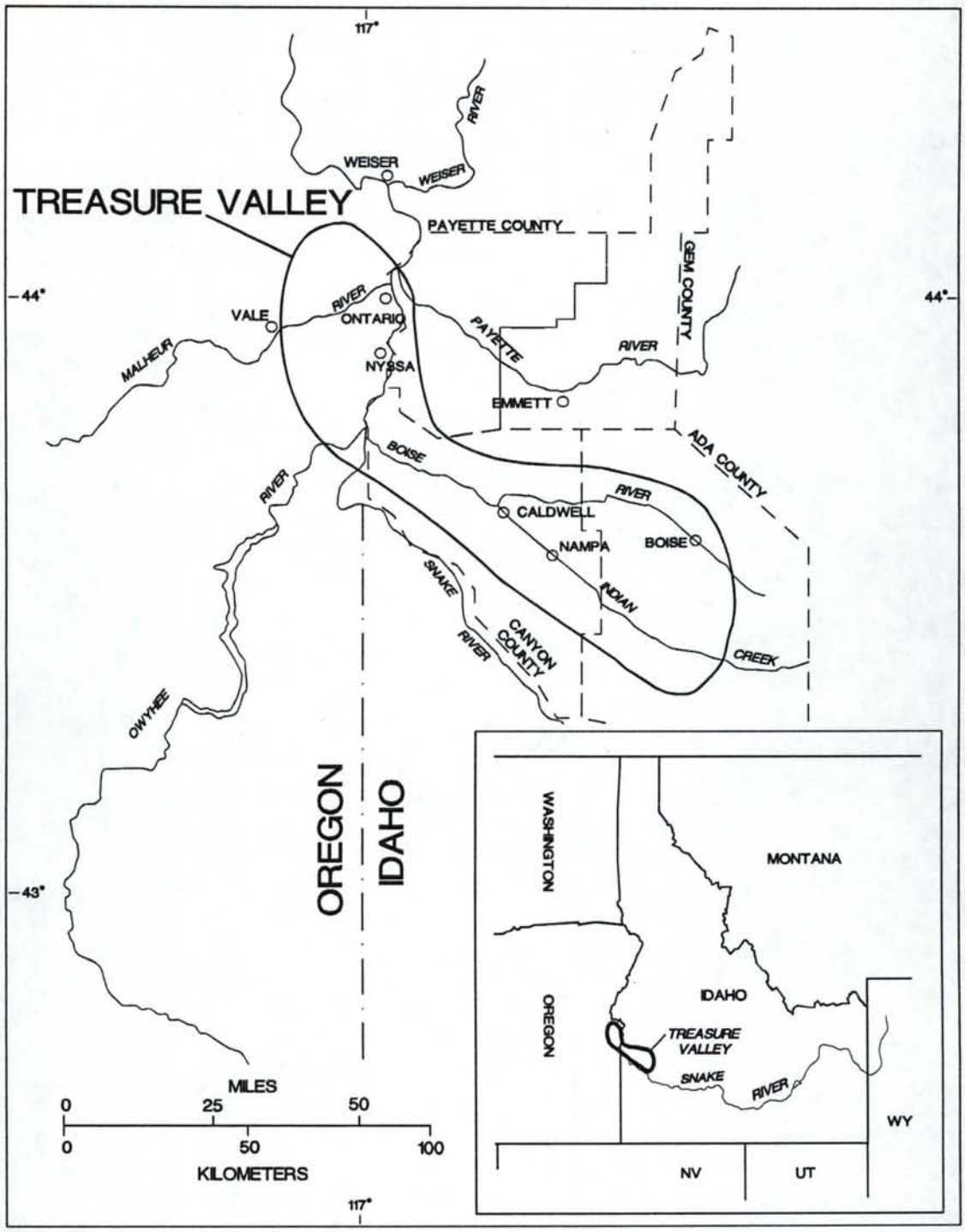
CHAPTER 1

INTRODUCTION

1.1 Background

Locating and evaluating potential sources of nitrate contamination to groundwater has become a focus of attention in the Treasure Valley region of eastern Oregon and western Idaho (Figure 1). In 1989, the Idaho Wellhead Sampling Program showed that about five percent of the wells in Gem and Payette counties, the two western-most Idaho counties in the Treasure Valley, exceeded the federal drinking water standard for nitrate-N (Mahler and Gardner, 1991). The same survey showed that 47 percent of the sampled wells exceeded two mg/L nitrate-N. The disposal of cull onions in unlined landfills was targeted by the Idaho Division of Environmental Quality as a potential source of nitrate contamination of groundwater in the western Treasure Valley. In 1990, the Idaho Division of Environmental Quality, the Idaho-Eastern Oregon Onion Commission, the Oregon Department of Agriculture and the University of Idaho collaborated on a study of cull onion landfills to determine whether concentrated onion waste leachate contained in the landfills posed a threat to the groundwater resources of the western Treasure Valley. This thesis presents the results of a three year study of two "typical", fully operational cull onion landfills.

Onion production in the Treasure Valley constitutes approximately 33 percent of the total annual U.S. storage onion crop and supplies greater than 50 percent of the total U.S. fresh onion market during harvest and storage seasons (Figueroa, 1989, Levi *et al.*, 1990). Production of onions in the Treasure Valley has increased from about 100,000 metric tons in 1975 to more than 230,000 metric tons in 1992 resulting in a farm value of approximately 100 million dollars per annum. The large quantity of onions produced in the Treasure Valley combined with stringent industry inspection standards create a significant volume of waste


TREASURE VALLEY
117°
44°
43°
WEISER
WEISER
RIVER
PAYETTE COUNTY
GEM COUNTY
VALE
ONTARIO
RIVER
MALHEUR
NYSSA
PAYETTE
RIVER
EMMETT
ADA COUNTY
BOISE
RIVER
CALDWELL
NAMPA
BOISE
SNAKE
RIVER
INDIAN
CREEK
CANYON COUNTY
OWYHEE
OREGON
IDAHO
MILES
0
25
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100
KILOMETERS
WASHINGTON
MONTANA
OREGON
IDAHO
TREASURE VALLEY
SNAKE
RIVER
WY
NV
UT

onions. Onions that do not meet size and shape standards, and those onions that exhibit storage-related decay are culled. Although the proportion of the annual Treasure Valley onion crop that is culled has decreased by 10 percent since 1975, safe disposal of 35,000 to 90,000 metric tons is required each year (Levi *et al.*, 1990).

Exposed cull onions can provide overwintering and breeding sites for onion maggots (*Delia antiqua*). Infestations of onion maggots may result in the loss of up to 90 percent of onion seedlings and a drastic reduction in the quality of late season onions for storage. State regulations in Idaho and Oregon require that cullage be tilled into the soil or buried to control the spread of onion maggots and diseases. In Idaho, tillage and burial must be accomplished by March 15 of each year (Title 22, Chapter 1, Idaho Code).

Large quantities of cull onions have resulted in three main disposal alternatives for producers and storage facility operators. Cull onions have been surface applied to agricultural and non-agricultural land, fed to livestock, and buried in landfills (Jensen, 1992). Shipping and application costs, land availability, and the burial deadline limit land application in many cases. Consequently, less than one percent of cullage is land-applied. Currently, 30 to 40 percent of cull onions are fed to livestock (Jensen, 1992). However, declining sheep numbers and poor nutritional value limit the use of cull onions as livestock feed (Levi *et al.*, 1990). Another group of alternative uses for cull onions is processing by frozen or dried onion industries or as an essential oil source. These alternatives have not significantly reduced the volume of onion cullage in the Treasure Valley (Levi *et al.*, 1990).

The most economical method of cull onion disposal is burial in unlined landfills. Disposal of cull onions in covered landfills is a historic management practice used to control insects and diseases, minimize offensive odors, and provide an inexpensive alternative for

discarding storage onion cullage. Sixty to 70 percent of annual onion cullage is discarded in landfills.

1.2 Statement of Purpose and Objectives

The purpose of this field investigation was to determine whether cull onion landfills in the western Treasure Valley are potential sources of nitrate contamination of groundwater. The general objective was to characterize leachate generation from cull onion landfills. The specific objectives were: (1) instrument two full-scale cull onion landfills to facilitate periodic sampling of leachate below the landfill base, (2) determine the physical controls on leachate movement through the landfill base, (3) investigate nitrate transport using a conservative tracer, and (4) determine the fate of nitrogen in cull onion landfill systems.

1.3 Method of Study

An existing landfill site owned and operated by Sun Top, Incorporated, a consortium of regional processing facilities, was evaluated in August, 1991. As a result of the initial site evaluation, two full-scale operational landfills were planned for instrumentation as part of this study. Investigating landfills constructed in a manner consistent with traditional cull onion disposal practices was considered an important aspect of the study, because the results were to be extrapolated to other cull onion landfill locations in the Treasure Valley region. The landfills were instrumented following the onion harvests of two consecutive years, 1991 and 1992. A two year/two landfill schedule was adopted because very little information about the state of landfilled cull onions was available prior to the research. This approach allowed information gained from the first landfill to be considered in the experimental design of the second landfill, thereby potentially improving the experimental results.

The timelines for instrumentation and loading for both cull onion landfill experiments were similar. Instrumentation of the landfills, hereafter referred to as L1 and L2, was

undertaken in September, 1991 and 1992. Loading of the landfills with cull onions occurred during the four months following instrumentation. Cull onions were mounded above the surface elevations of L1 and L2 and soil caps were placed on top of the mounded onions in January, 1992 and February, 1993.

Sampling and analysis of solution from within both landfills and of leachate from the matrix below both landfills were performed over a forty month period from November, 1991 through March, 1994. Solution and leachate samples were analyzed for nitrate, ammonium and organic nitrogen to determine the relative changes in concentrations of these components over time. The samples were also analyzed for tracer concentrations to establish leachate travel times. Sampling, particle-size analyses and in situ saturated hydraulic conductivity measurements of the sedimentary matrix at the base of L2 were conducted in September, 1992. These physical parameters were important in establishing the potential rate at which nitrate in the landfill solution might be transported from the landfills. These data were not obtained for L1 because the information was not considered necessary until after the preliminary results from L1 were reviewed. A geostatistical analysis of the in situ hydraulic conductivity measurements from L2 was conducted in December, 1992 to evaluate the spatial variability of leaching through the landfill base. Laboratory measurements of saturated hydraulic conductivity and water retention for the matrix samples collected in September, 1992 were conducted in March, 1993. A RETC computer analysis of the unsaturated conductivity distribution beneath the cull onion landfills was conducted in February, 1994, after it was hypothesized that leaching through the sedimentary profile was an unsaturated flow problem.

This document is divided into six chapters. Chapter 1 presents a background discussion of the cull onion landfill problem, enumerates the objectives of the study and outlines previous and related research. Chapter 2 presents the materials and methods used in

the study, including a description of the landfill site and the design, instrumentation, sampling and analysis of both L1 and L2. Chapters 3 and 4 embody the results and discussion of the study. Chapter 3 is aimed at analyzing the hydraulic controls on leachate generation from the experimental landfills. This chapter also discusses the tracer experiment, focusing on potential nitrate transport. Chapter 4 reports the results of the nitrogen analyses and discusses the fate of nitrogen in the cull onion landfill systems. Finally, Chapter 5 enumerates the important conclusions of the study and presents recommendations for additional study and improved landfill management.

1.4 Previous and Related Work

Three areas of research are important to the study of cull onion landfills: (1) the general fate of nitrogen in aqueous and solid waste systems, (2) the degradation of onions and onion waste, and (3) the study of systems that are analogous to landfilled onion waste. Although significant research has been dedicated to understanding and managing municipal landfills, a review of the literature shows that most of the reported processes and technical considerations are not applicable to onion waste. Reigert (1992) notes that nitrate is not a contaminant normally associated with municipal landfills and that most municipal waste has low initial quantities of nitrogen.

Characterization of the nitrogen composition of landfill leachate was a primary objective of the field investigation. Therefore, a brief review of mineralization and nitrification of nitrogen is included in this literature review. Many common microbes catalyze the transformation of organic nitrogen to ammonium ion (Jansson and Persson, 1982; Paul and Clark, 1988). This process is termed mineralization, and occurs under both aerobic (well oxygenated) and anaerobic (low oxygen) conditions. In aerobic systems, ammonium may be oxidized to nitrate. This transformation, termed nitrification, is common in the soil and other environments where the necessary obligate aerobic bacteria persist (Schmidt, 1982; Paul and

Clark, 1988). Reduced conditions associated with anaerobic environments generally preclude the persistence of nitrate (Patrick, 1982).

At the time of this investigation, no formal scientific research had been published that addressed the environmental impact of landfill disposal of cull onions or allied crops. Warncke (1991) studied the rate of aerobic onion decomposition and the rate of nitrogen release from buried and unburied onions. Warncke found that the decomposition of whole onions was slow, especially for those left above the soil surface. Whole onions showed little decomposition during the first four weeks, even when buried, but by the end of seven weeks some decomposition had occurred. Fifty percent of the fresh weight of whole onions was lost to dehydration during the 74 day study. Unburied halved and quartered onions dehydrated much more rapidly, and began to decompose after four to six weeks. Buried halved and quartered onions did not dehydrate as rapidly as the unburied onions, but showed similar decomposition rates. Nitrogen loss was measured during the course of degradation. During the study, buried onions released significantly more nitrogen than those left on the surface. Buried halved and quartered onions lost significantly more nitrogen than buried whole onions.

Disposal of animal waste is the nearest analog to landfill disposal of cull onions. Leachate from feedlot-confined livestock and lagoon-treated dairy waste has been studied as a potential source of nitrate contamination of groundwater in the western United States (Stewart *et al.*, 1967; Patrick *et al.*, 1987; Korom and Jeppson, 1994). A number of related studies have shown that compaction of organic waste into feedlot topsoil can create a zone of low hydraulic conductivity beneath the organic layer and cause anaerobic degradation of the nitrogen compounds above the organic layer (Mielke *et al.*, 1974; Schuman and McCalla, 1975; Barrington and Jutras, 1983). These studies are applicable to the cull onion landfill system because of the large quantity of water and organic material contained in the cull onions and the potential inhibition of nitrification. Oxygen diffusion and subsequent nitrate formation

beneath feedlots were reported by Mielke and Ellis (1976) to occur after the facilities were closed. Oxygen diffusion increased aerobic microbial activity under inactive feedlots, resulting in the nitrification of concentrated ammonium retained above the low hydraulic conductivity zone and subsequent degradation of the zone itself. This result may be important in the long-term management of cull onion landfills if nitrification is shown to be inhibited at some point during the degradation of the cull onions.

The physical and chemical processes that control the formation of organic flow boundaries in porous media have been studied by a number of researchers. Barrington and Broughton (1991) reported that physical clogging of soil pores by organic material dominated the formation of low conductivity zones in ponded dairy lagoons. Aggregation of soil and organic material by chemical and biological processes was reported to enhance the sealing of soil pores. The authors hypothesized that porous linings of lagoons and holding ponds filtered out organic material, resulting in a dense organic mat at the lining surface. The formation of self-sealing organic liners in an anaerobic dairy lagoon has been reported to require approximately two months (Sewell, 1978). Brune (1990) noted that other researchers have reported that self-sealing liners may take up to one year to become sufficiently established. In most cases, self-sealing reduces matrix permeability by one to three orders of magnitude (Sweeten, 1990). However, cases have been identified in which adequate organic liners did not develop (Korom and Jepsom, 1994; Brune, 1990). The occurrence of such events is sporadic and not well understood. Even so, installation of constructed self-forming liners beneath ponds and lagoons, a process known as artificial gleization, has been adopted by the Soil Conservation Service (SCS, 1979; SCS, 1990).

CHAPTER 2

MATERIALS AND METHODS

2.1 Introduction

The goals of the following presentation of the materials and methods used in the cull onion landfill study are: (1) describe the experimental site, (2) provide a detailed description of the measurements utilized in the analysis, and (3) provide a chronology of instrumentation and sampling of L1 and L2.

2.2 Site Description

The experimental landfills are located at a landfill site in the western Treasure Valley of south-western Idaho in Payette County (Figure 1). The landfills were constructed in undeveloped rangeland of the terraced foothills that rise from the Snake River flood plain along the border of Idaho and Oregon (Figure 2). The experimental site is adjacent to the D-line canal, approximately 5 km east of Nyssa, Oregon and 4 km east of the Snake River. The elevation of the site is approximately 740 m, roughly 80 m above the elevation of the Snake River.

The experimental landfills were excavated in laminated lacustrine sediments that mantle coarse terraced gravels. The terraced geomorphology of the sediments originated from periodic catastrophic flooding that occurred throughout the western Snake River Plain during the Pleistocene epoch (Malde, 1991; Othberg, 1994). Although the depth of the sediments underlying the experimental site is unknown, sediments in other regions of the western Snake River Plain have been reported to be as much as 1.7 km thick (Malde, 1991). The surface of the experimental site is an Owyhee silt loam soil (coarse-silty, mixed, mesic, Xerollic Camborthid) (SCS, 1976). Vegetation in the immediate vicinity of the experimental

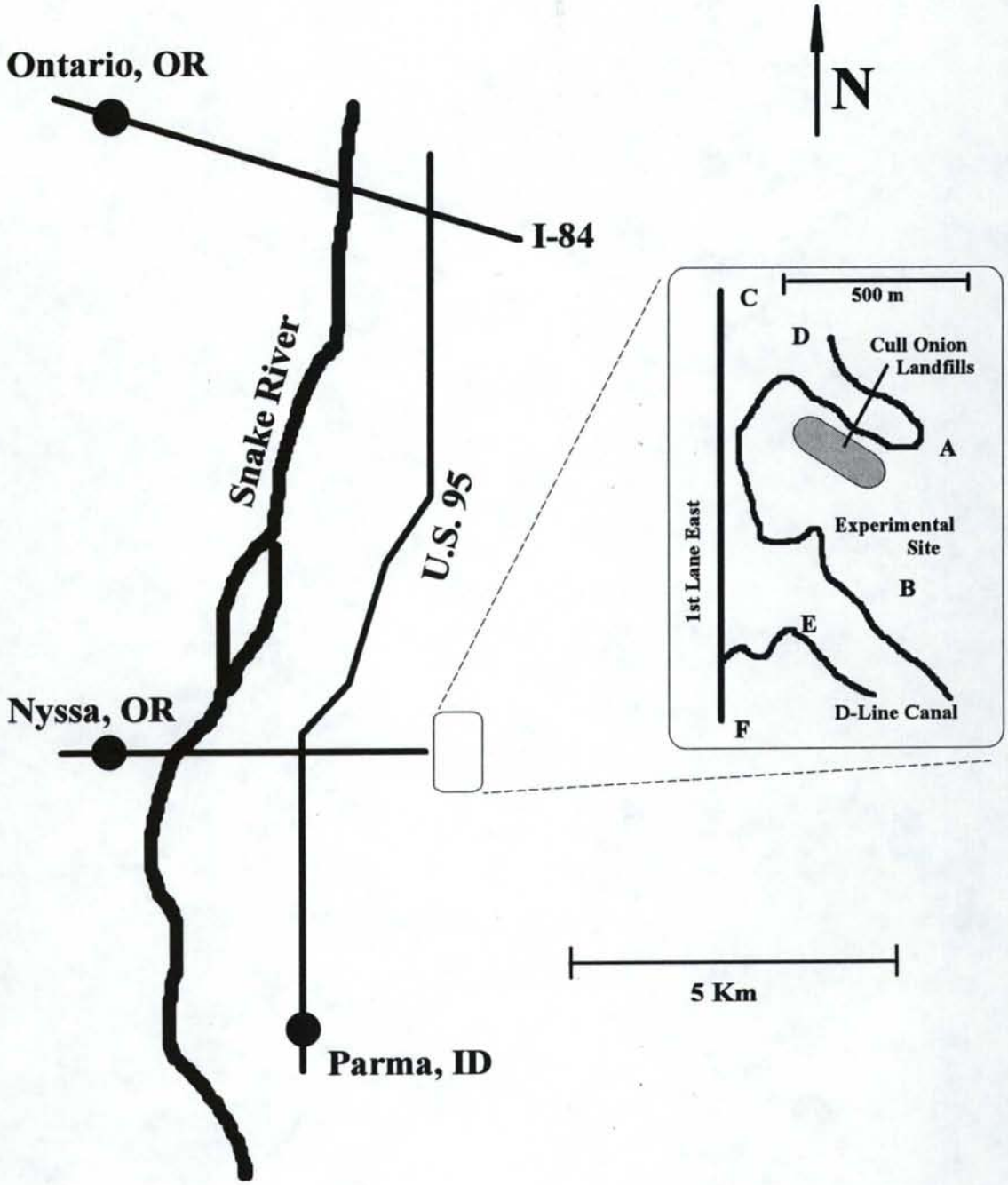

Figure 2. Location of the experimental site and cull onion landfills. The site was situated on secondary land above the Snake River. Six wells (A through F) are shown that were used to obtain ground-water level information.

site is cheatgrass (*Bromus tectorum*) and sagebrush (*Artemisia tridentata*).

The characteristic soil type and vegetation cover at the landfill site suggest a relatively arid environment. Average annual precipitation at the University of Idaho Agricultural Experiment Station in Parma, Idaho, 5 km south of the experimental site, ranges from 23 to 38 cm with the majority of precipitation occurring between the months of October and May. The ten year average precipitation for the period beginning in 1982 is 26.2 cm.

Aquifer depth beneath the experimental landfills was estimated by extrapolating information obtained from well logs of four nearby wells (C, D, E and F) (Figure 2, Table 1). Ground surface elevations for the wells were estimated from topographic maps. Well logs were not available for two unused wells located on the experimental site, so their completion depths were unknown (A and B, Figure 2). However, static water levels were measured to be 27 and 34 m, respectively, in wells A and B.

Table 1. Water levels and aquifer depths in wells near the cull onion landfill site.

Well #	Ground Surface Elevation[1] (m)	Static Water Level (m)	Completion Depth[3] (m)	Water Level Elevation (m)	Aquifer Elevation (m)
A	750	27[2]	----	723	
B	753	34[2]	----	719	
C	732	30[3]	57	702	675
D	720	22[3]	37	698	683
E	723	34[3]	60	689	663
F	722	28[3]	53	694	669

[1]ground surface elevation estimated from topographic map, [2]measured water level, [3]water level obtained from well logs

Based on the completion depth of the off-site wells, the shallowest aquifer used as a drinking water supply for local residents was estimated to be about 60 m below land surface at the experimental site.

2.3 Landfill Construction

The experimental landfills used in this study were constructed in the same manner as historic landfills in the western Treasure Valley. Layers of sediment were excavated in rectangles of approximately 100 m in length and 10 m across using heavy construction equipment (Figure 3A). Sediments were removed until the landfill was 5 to 6 m deep (Figure 3B). The bases of the landfills were relatively flat except for angled entry and exit ramps at each end. Landfill L1 was constructed approximately 5 m above the surface elevation of D-Line canal. The D-Line canal is an unlined irrigation canal located about 15 m north-east of L1. Leakage characteristics of the canal are unknown. Any effect of water leaking from the canal on the sediments underlying L1 could be observed in the water-potential data and could be accounted for. Landfill L2 was located farther away from the canal than L1; therefore, it was not affected by flow in the canal.

2.4 Experimental Design

Different experimental designs were implemented for L1 and L2 in an attempt to quantify the spatial variability of vertical leachate movement through the underlying sediments. Landfill L1 was divided into three 25 m sections in order to allow basic replication of measurements in the landfill; L2 was divided into a 12 block random grid that was amenable to geostatistical statistical analysis (Figures 4 and 5).

Three banks of pressure/vacuum soil-water samplers (suction lysimeters; model 1920, Soil Moisture Equipment Corp., Santa Barbara, CA) were located in each section of L1

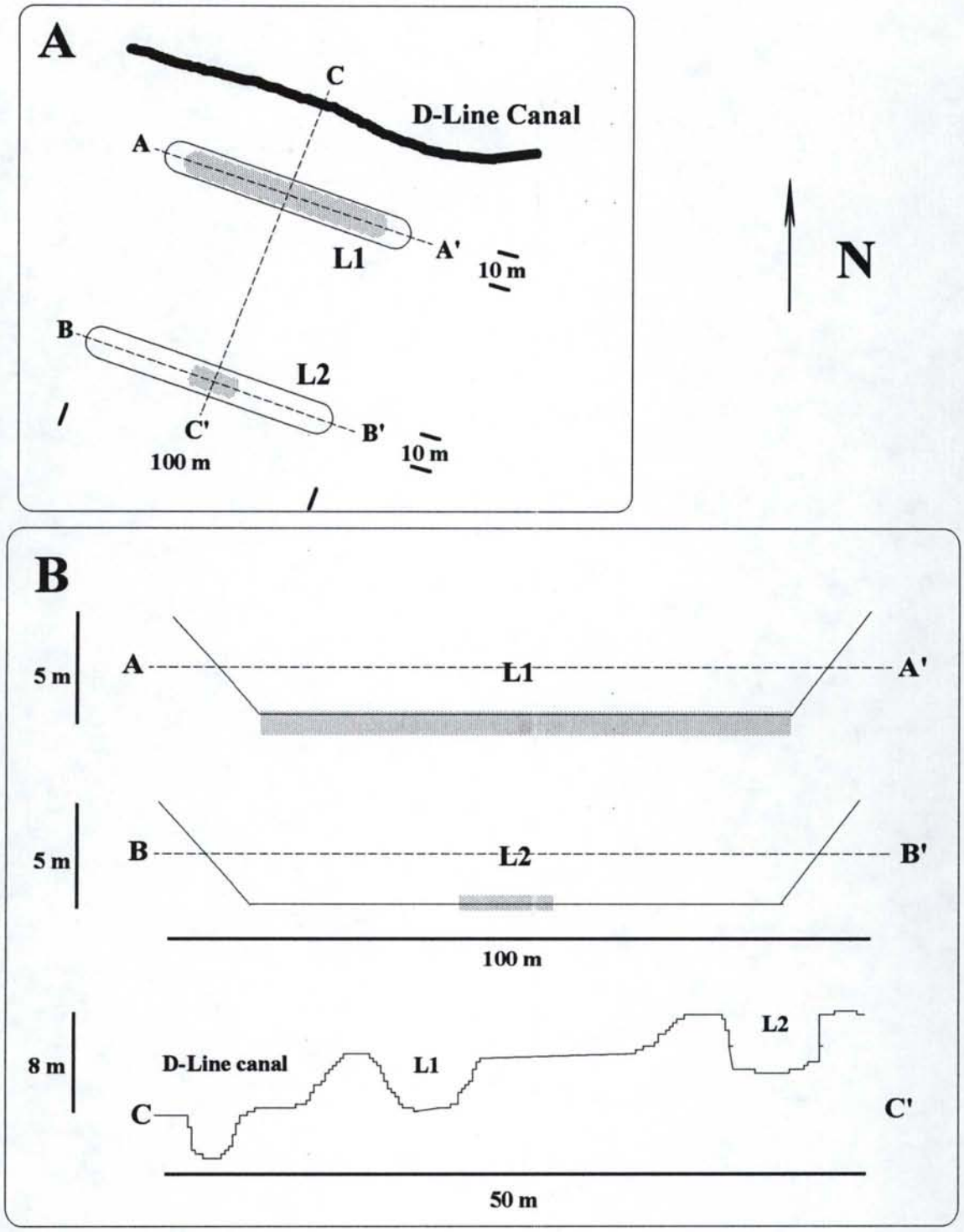

Figure 3. Plan view (A) and cross-sections (B) of landfills L1 and L2. The relative elevations of the landfills are shown in relation to D-Line canal. Shading denotes instrumented regions of the landfills.

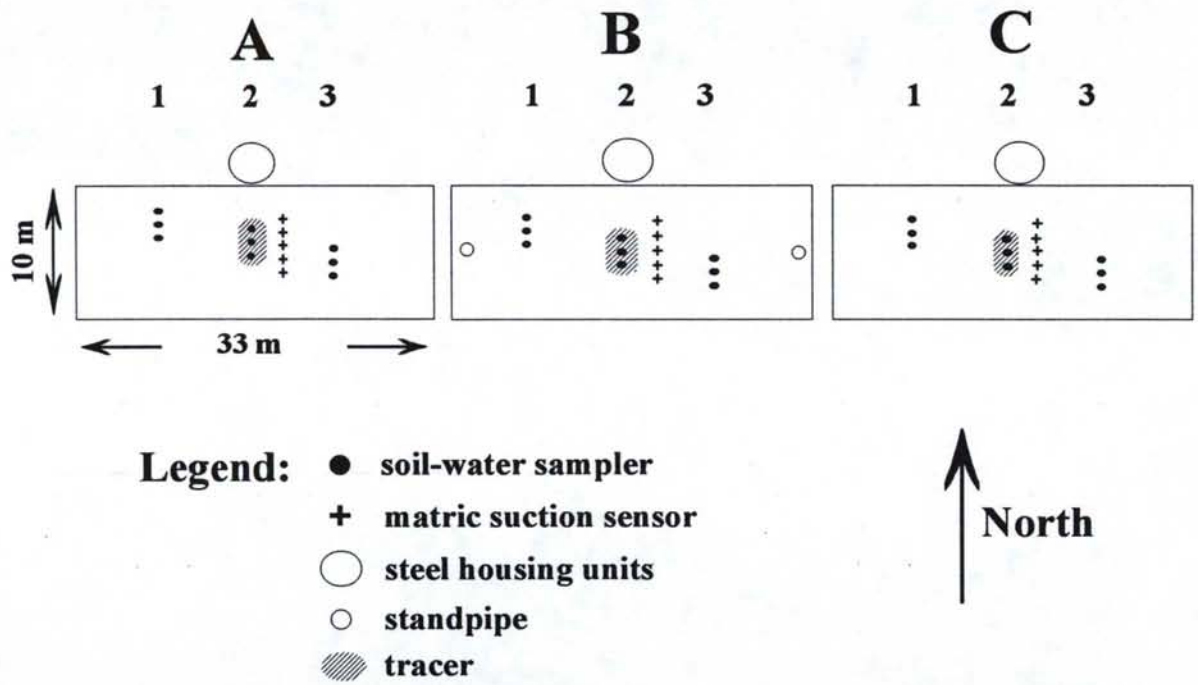

Figure 4. A schematic diagram of L1. Approximate location of soil-water samplers, matric suction sensors and sampling culverts are shown. Three banks of soil-water samplers were installed in each of three sections (A, B and C) with units at 0.5, 1 and 2 m below the landfill base. A bank of matric suction sensors was installed in each section with units at 0.5, 1, 1.5, 2 and 3 m beneath L1. Each group of samplers and sensors were accessible at the adjacent housing unit. Two standpipes were located in section B. Tracer was applied along the center bank of each section.

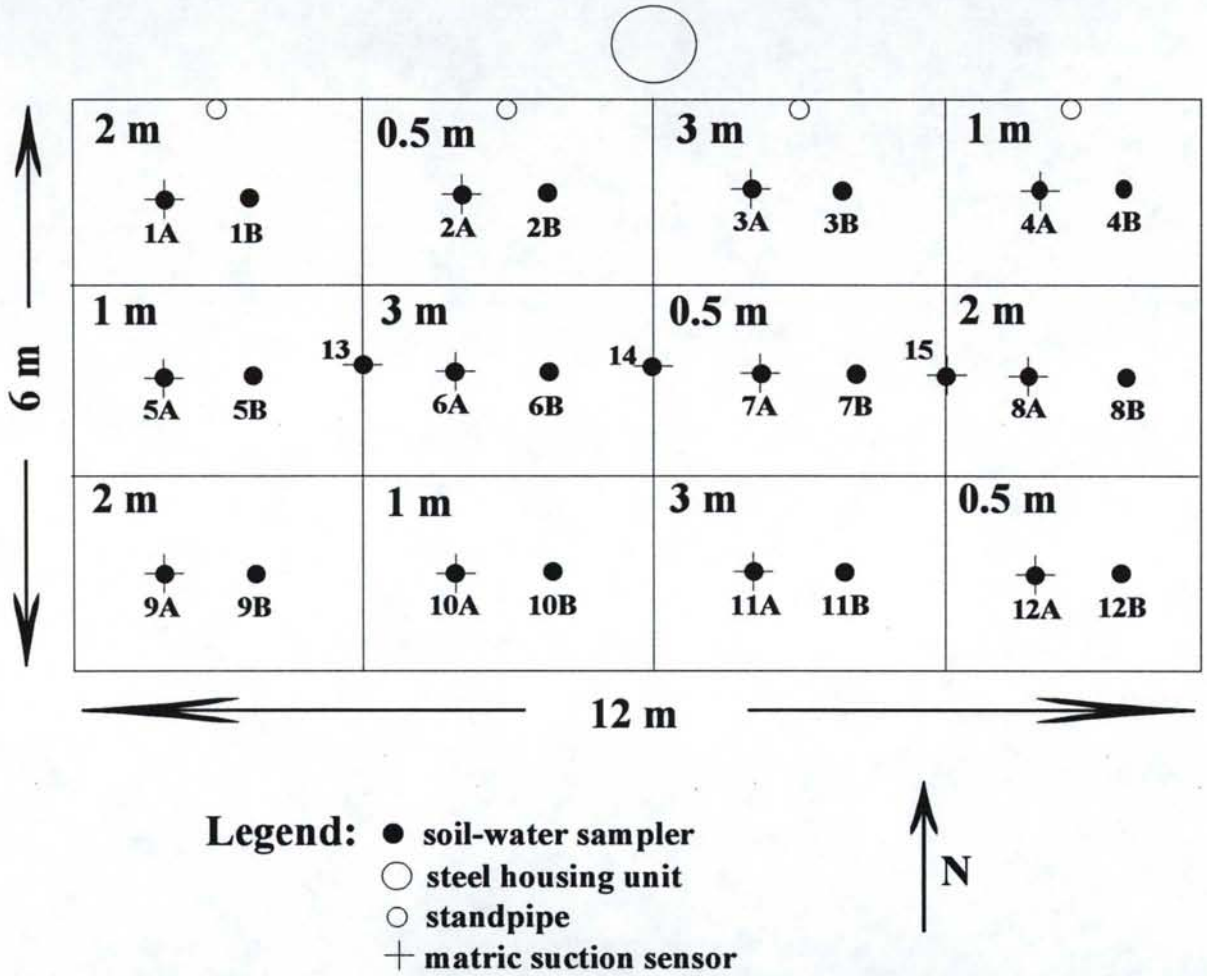

Figure 5. A schematic diagram of L2. Two soil-water samplers were located in each of twelve sections. The distribution of sampler depths was based on a modified random-block design. Samplers were placed at 0.5, 1, 2 and 3 m. Three additional units were placed at 4 m beneath the landfill base. Matric suction sensors were installed with the western-most soil-water sampler in each section. Suction sensors were also installed at 4 m. Tracer was applied around each soil-water sampler in the grid. Four standpipes were located along the north wall of the landfill.

(A, B and C). Each bank consisted of three soil-water samplers located at depths of 0.5, 1, and 2 m beneath the landfill base. Each soil-water sampler was attached to two 8 to 12 m lengths of 3.2 mm poly-ethylene access tubing and vacuum tested for 48 hours. Ten centimeter diameter bore holes were augered into the landfill base to the desired installation depth. A 10 cm thick layer of silica flour was poured into the the bottom of each hole and the sampler cup was positioned firmly into this layer. Silica flour was poured in the annulus about the cup and tamped into place. Alternate pouring and gentle tamping was continued until silica flour extended at least 5 cm above the sampler cup. The remaining volume was back-filled to the surface with granulated bentonite clay to ensure no preferential flow paths existed from the surface of the landfill base to the cup.

In addition to the soil-water samplers, five matric suction sensors (Soiltronics, Inc., Belleview, Wa.) were installed in each section of L1 at depths of 0.5, 1, 1.5, 2, and 3 m. The sensors measured soil suction by thermal dissipation. Each sensor was laboratory tested, and its calibration was established. The suction sensors were installed in individual holes by placing the units at the desired depth and packing the annulus with material removed from the bore hole. Like the soil-water sampler installation, the residual volume was back-filled with bentonite.

Soil-water sampler access tubes and suction sensor wires for each of the three sections of L1 were run to three central facilities (Figure 4). Each central facility housed access tubes from nine soil-water samplers and wires from five suction sensors in a vertical 1.2 m high by 1 m diameter section of steel culvert. The suction sensors were connected to data loggers (Model CR10, Campbell Scientific, Inc., Logan, UT) to record the soil suctions each day. Two poly-vinyl chloride (PVC) standpipes were positioned within Section B to obtain water samples from within L1.

A bromide (Br-) tracer was applied to the base of L1 in order to observe water movement beneath the landfill and to mimic the movement of nitrate through the sediments. Potassium bromide (KBr) was applied in a 0.1 m deep trench of 0.3 m by 5 m dimensions along the center soil-water sampler bank in each section of L1. The application rate was 3.3 kg Br^-/m^2. The granular KBr was mixed with the material removed from the trench and the trench was backfilled with the mixture.

Landfill L2 was instrumented with 24 soil-water samplers and 12 gypsum resistance block soil suction sensors. Soil-water samplers and suction sensors were placed in a 6 m by 12 m grid with six replicates at depths of 0.5, 1, 2, and 3 m beneath the landfill (Figure 5). A suction sensor was installed in the same bore hole as one of the soil-water samplers in each block. Three soil-water sampler/suction sensor combinations were placed at 4 m beneath the landfill. The installation procedure for these soil-water samplers and matric suction sensors was identical to the procedure described for L1. As in L1, access tubes and electrical leads were mounted in a central facility for monitoring. Four standpipes were positioned along the landfill wall to obtain water samples from within the landfill.

Granular potassium bromide was applied around each pair of samplers in the grid. A measured amount of KBr was applied in a 0.15 m diameter circle around each sampler. The rate of application was about 0.4 kg Br^-/m^2.

2.5 Landfill Sampling and Analysis

Soil samples from L1 were collected in 0.3 m increments to a depth of 2 m during the installation of the matric suction sensors. The samples were stored in a cooler for transport to the University of Idaho and air-dried prior to chemical analysis. The samples were analyzed by the University of Idaho Analytical Laboratory (UIAL) to identify background levels of soil

nitrate-N (NO_3-N) and ammonium-N (NH_4-N). Tracer analyses were conducted by myself at the University of Idaho soil physics laboratory.

Instrumentation of L1 was completed and onion disposal commenced in September, 1991. Measurement of soil suction data began in November, 1991. Onion disposal continued through December until the landfill was filled to capacity and capped with a 0.5 m thick soil cap in January, 1992. Soil caps were constructed from material excavated from the landfill. On January 25, 1992 the first set of leachate samples were collected from the soil-water samplers. Sampling continued every two weeks for the first two months and monthly thereafter through June, 1992. Sampling of L1 during the second year of the investigation progressed on a bi-monthly basis through February, 1993, at which time, a quarterly sampling schedule was established. Sampling of the standpipes began in April, 1992, and continued on the same schedule as leachate sampling.

Samplers were maintained under vacuum for 18 to 24 hours prior to sampling to extract leachate from the landfill matrix. Leachate was removed from the soil-water samplers using a pressure/vacuum hand pump, and water samples were drawn from the standpipes using the same pressure/vacuum pump and vacuum flasks. Two 500 ml volumes of leachate were removed from each standpipe before a sample was taken. Samples from the soil-water samplers and standpipes were collected in clean plastic centrifuge tubes. The centrifuge tubes were triple rinsed with millipore filtered deionized water prior to use. Samples were placed on ice for transport and were delivered within 24 hours to the UIAL for analysis. Samples collected during each sampling event included one trip blank per cooler, one transfer blank per 10 samples and one field duplicate per 10 samples. Leachate samples were not acidified because of the high buffering capacity of the sample matrix. After analysis by the UIAL, samples were frozen for later tracer analysis.

Leachate samples from below L1 and water samples from within the landfill were analyzed for nitrogen and bromide (Br^-). Analyses for NO_3-N, NH_4-N, and total Kjeldahl nitrogen (TKN) were performed by the UIAL on an ALPCHEM 300 Series rapid flow analyzer (Alpkem Corporation, Clackamas, OR) using current EPA procedures (EPA 353.2, 350.1 and 353.4). Total Kjeldahl nitrogen is a standard measure of the sum of organic nitrogen and NH_4-N. The analysis of leachate samples for Br- was performed with an Orion model 94-35 bromide ion-specific electrode in conjunction with an Orion model 90-02 double junction reference electrode (Orion Research, Inc., Boston, MA) and a Beckman Phi 71 pH meter (Beckman Instruments, Inc.).

Soil samples from L2 were collected from depths of 0.5, 1, 2, 3 m by compositing material removed from the bottoms of both soil-water sampler bore holes in each grid block. Soil samples were stored in a cooler after sampling for transport to the laboratory. Once in the laboratory, the samples were air-dried and analyzed for chemical composition and physical properties.

Chemical analyses of the L2 soil samples for background NO_3-N and NH_4-N were performed by the UIAL. Particle-size analyses were performed on all soil samples removed from L2 to identify the component size fractions of the sediments. The soil samples were ground and dry-sieved using a standard particle-size analysis procedure (Gee and Bauder, 1986). Bulk-densities were determined by repacking ten oven-dried landfill sediment samples into 8.5 cm diameter by 6.0 cm rings and weighing the repacked samples. In situ bulk-density measurements were not obtained because of the cemented nature of the sediments. Porosity was estimated in order to estimate maximum water holding capacity of the porous matrix. Porosity was estimated from the ratio of bulk-density to particle density. Particle density of most sediments, particularly those with high sand fractions, was reported to be 2.6-2.7 g/cm^3,

approximately that of quartz (Hillel, 1982). Porosities of the landfill sediments were also estimated by measuring the saturated water content of five of the repacked cores.

Saturated conductivity of the unconsolidated sediments was determined using a constant head permeameter (Klute and Dirksen, 1986). In addition, saturated hydraulic conductivity was measured in situ at 27 randomly selected positions across the base of L2 using two Guelph constant head permeameters (model 2800 Guelph Permeameter, Soil Moisture Equipment Corp., Santa Barbara, CA). The in situ saturated hydraulic conductivity measurements were used to estimate the maximum leachate flow velocity through the landfill base. The geostatistical analysis program GEOEAS (USEPA, 1991) was used to estimate the degree of spatial dependency of the in situ conductivity values. Unsaturated flow characteristics of the landfill sediments were evaluated using the RETention Curve (RETC) software (van Genuchten, *et al.* 1981)

Leachate and landfill solution sampling of L2 began on November 23, 1992, and followed approximately the same schedule as L1. Landfill L2 was filled with cull onions to capacity in February, 1993 and capped in March, 1993. Sampling of L2 began on a monthly schedule and was shifted to a quarterly sampling period in May, 1993. Leachate sampling of L2 adhered as closely as possible to U.S. Environmental Protection Agency groundwater sampling protocol (EPA, 1983). Samples collected during each sampling event included one trip blank per cooler, one transfer blank per 10 samples and one field duplicate per 10 samples. Samples were put directly on ice and stored at approximately 4 ^{o}C until delivery to the UIAL. Leachate samples were not acidified because of the high buffering capacity of the sample matrix. After analysis by the UIAL, samples were frozen for later tracer analysis. Leachate samples from below the cull onion landfill and solution samples from within the landfill were analyzed for NO_3-N, NH_4-N, and TKN by the UIAL in the same manner as the samples from L1.

A mass-volume relationship was determined for a sample of the population of cull onions in L1 to facilitate nitrogen loading calculations. Onions were selected at random over the entire landfill, in order to not bias the result to any particular onion type. Onion samples were placed in three 0.1 m^3 containers and weighed. A mean bulk density and standard deviation were determined.

CHAPTER 3

ANALYSIS OF HYDRAULIC DATA

3.1 Introduction

The goal of this analysis is to determine the factors that control leachate movement from the cull onion landfills. In particular, flow characteristics of a representative volume of sediments extending 3 m below the landfill bases are investigated. The properties of interest are the particle size distribution, bulk density and porosity of the sediments beneath the landfills, the hydraulic loading within the landfills, the energy status (matric suction) of the pore water in the underlying sediments, and the hydraulic conductivity of the landfill base.

Evaluation of factors that control leachate movement below the cull onion landfills requires consideration of sources of experimental error associated with measurements and heterogeneity associated with leachate movement. The use of established analytical methods for measuring hydraulic conductivity values and matric suction data and cross checking results against calculated values based on independent data are typical methods for minimizing experimental error. This process is adopted to ensure that experimental error does not lead to an erroneous evaluation of flow velocities. Spatial heterogeneity is evaluated using direct observation as well as stochastic methods. The geostatistical computer program GEOEAS (USEPA, 1991) is used to ascertain whether variability is spatially dependent, and to quantify the effects of spatial variability on the overall conceptual model.

3.2 Landfill Matrix Physical Properties

Particle size analyses indicate that the landfill sediments, removed from their natural aggregated state and ground into their component materials, were similar among all sampling locations (Figure 6). About 85 percent of a given sample was composed of sand and the

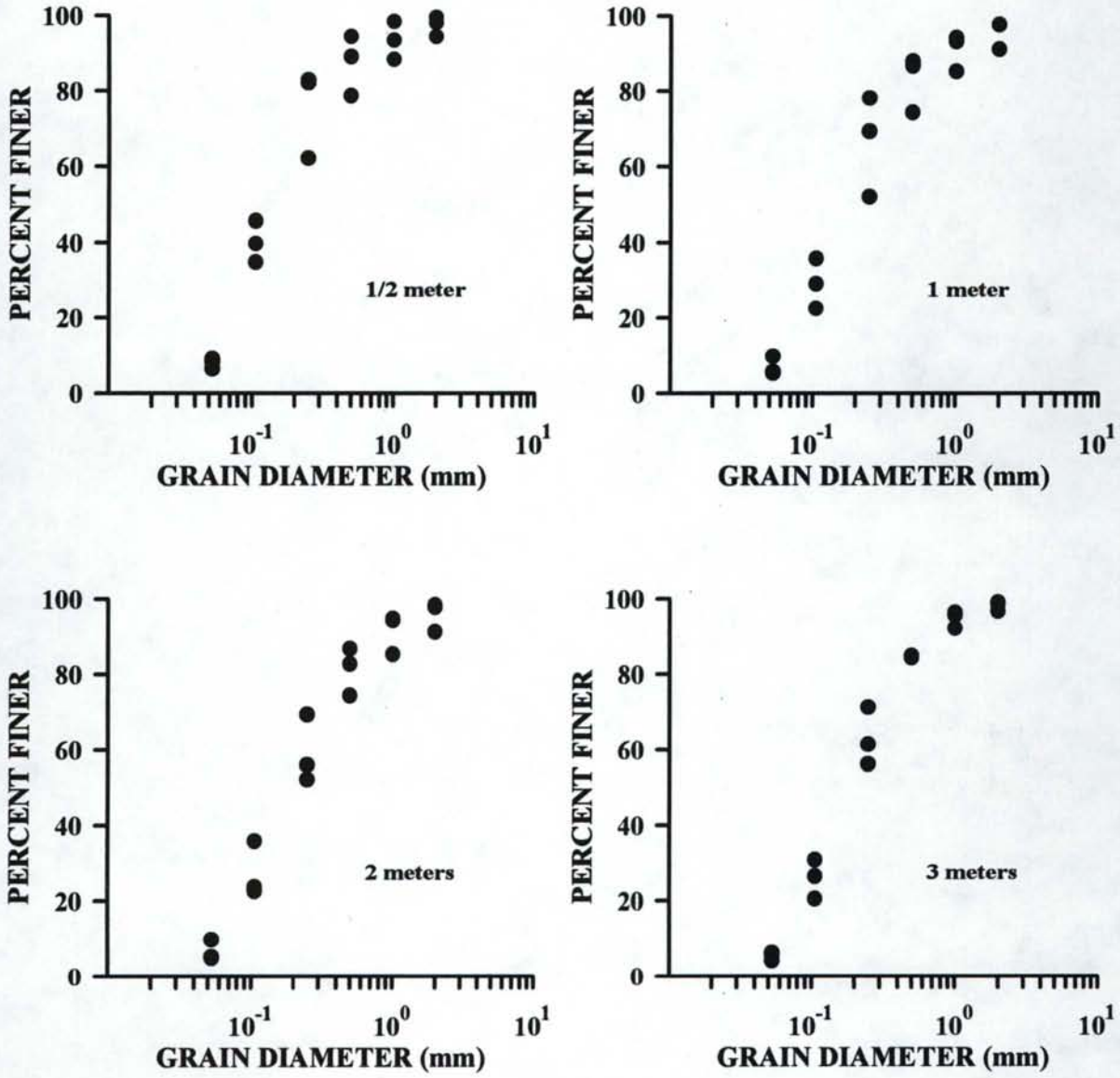

Figure 6. Particle size analyses of cull onion landfill sediments. Soil samples were taken from 0.5, 1, 2 and 3 meters beneath the base of L2.

remaining 15 percent was silt and clay-sized particles. The uniformity coefficient (Iu) is a measure of the distribution of particle diameters. Values of Iu for the landfill soil samples range from 2.5 to 5.0, indicating a very poorly-graded material. Material containing a single size particle has an Iu value of 1, while a well-graded soil may have an Iu value greater than 1000 (Hillel, 1982). This analysis indicated a nearly homogeneous material distributed throughout the representative volume; however, it did not account for structural heterogeneity that may have been introduced during the deposition and genesis of the sediments. Field observations show that, in general, the sediments are quite structured, and suggest that analyses based solely on particle size analysis is likely to be misleading.

Bulk density of the repacked landfill sediments was determined to be 1.44 g/cm^3 with a standard deviation of 0.07 g/cm^3 (Table 2). Determination of bulk density from repacked cores did not account for structural disturbance in the material when it was removed from its natural environment. Porosity of the repacked sediments was estimated by measuring the saturated water content of five of the repacked cores. Using this method, the mean porosity was estimated to be 0.38 with a standard deviation of 0.01 (Table 2).

Table 2: Bulk Density and Porosity of Repacked Landfill Sediments

Sample	B.D. (g/cm^3)	Porosity	Sample	B.D. (g/cm^3)	Porosity
1	1.41	-----	6	1.55	.35
2	1.34	-----	7	1.37	.40
3	1.49	-----	8	1.46	.39
4	1.54	-----	9	1.46	.37
5	1.33	-----	10	1.48	.38

mean B.D.=1.44, std=0.07; mean porosity=0.38, std=.02

The morphology observed in the landfill sediments is probably a result of periodic depositional episodes and subsequent weathering. Horizontal discontinuities were observed in the walls of L2 in the form of laterally extensive lacustrine deposits. Depositional layers varied in thickness from 0.3 to 1.5 m and extended the entire face of each wall. The layered sediments exhibited a range of different structures, some clayey, fractured and penetrated by calcified roots; others consisted of cemented sands relatively free from macropores. Evidence of heterogeneity was also observed in a stream channel deposit that bisects the walls of L2. The residual alluvium forms a stony region that penetrates a number of sedimentary layers. These observations suggest significant structural heterogeneity may exist in the matrix underlying the experimental cull onion landfills.

3.3 Landfill Hydraulic Loading

Measurements of standing water in the landfills showed that both L1 and L2 retained about 2 m of water over the period of this investigation. Water levels in L1 remained about 2 m above the landfill base after eighteen months of operation. The water level in L2 was observed near land surface, approximately 4 m above the landfill base, in February 1993. This maximum water level coincided with significant snow melt running into the landfill. The water level in L2 was approximately 2.3 m above the landfill base in May 1993. Three possible sources of water in the cull onion landfills were: (1) water released from decomposing onions, (2) recharge from direct precipitation, and (3) runoff from the local watershed.

The amount of water released by the decomposing onions was calculated from the bulk density of landfilled onions and an estimate of the water content of onions. The bulk-density of landfilled onions was estimated to be about 630 kg/m^3. Based on this value, an estimated landfill volume of 6000 m^3, and the 90 percent by weight water content reported by Levi *et al.* (1990), the total volume of water in a landfill was calculated to be approximately 3500 m^3, or about 50 percent of the landfill volume. In light of the large contribution of

water from the onions themselves, the 26 cm of annual precipitation was inconsequential. Runoff entering the landfills from the local watershed was extremely variable, depending on the type and intensity of precipitation. Runoff was probably a factor in maintaining standing water in both L1 and L2 during the first winter of their operation, although runoff was only observed to enter L2. Water was also observed to pond on the landfill caps as they subsided into concave configurations. Water levels in the landfills during these periods were about 1 m below the level of the ponded water, indicating a downward gradient and probable recharge to the landfills.

Long term existence of standing water in L1 and L2 suggested that a restrictive layer was formed in cull onion landfills during the first few months of operation. The highly organic nature of the landfill contents and the large volume of water released from decomposing onions suggested that hydraulic and overburden pressures forced decomposing onion material into the matrix. Similar results have been reported in research of leaching beneath cattle feed lots and dairy-waste holding ponds in which organic material was hypothesized to form a low conductivity layer which restricted nitrate leaching (Mielke *et al.*, 1974; Schuman and McCalla, 1975; Sewell, 1978; Barrington and Jutras, 1983; SCS, 1990).

3.4 Matric Suctions Beneath L1 and L2

Matric suction values beneath L1 decreased with depth below the landfill base prior to loading with onions (Figure 7). Higher matric suction values corresponding to drier initial conditions were associated with shallower sensor locations. Initial matric suctions in L1 decreased with depth from 400 to 800 cm of water at 0.5 and 1 m to about 180 cm of water at 3 m. These initial conditions were due to evaporation from the exposed surface of the landfill base. Landfill L1 stood open to the hot, dry atmospheric conditions for three or

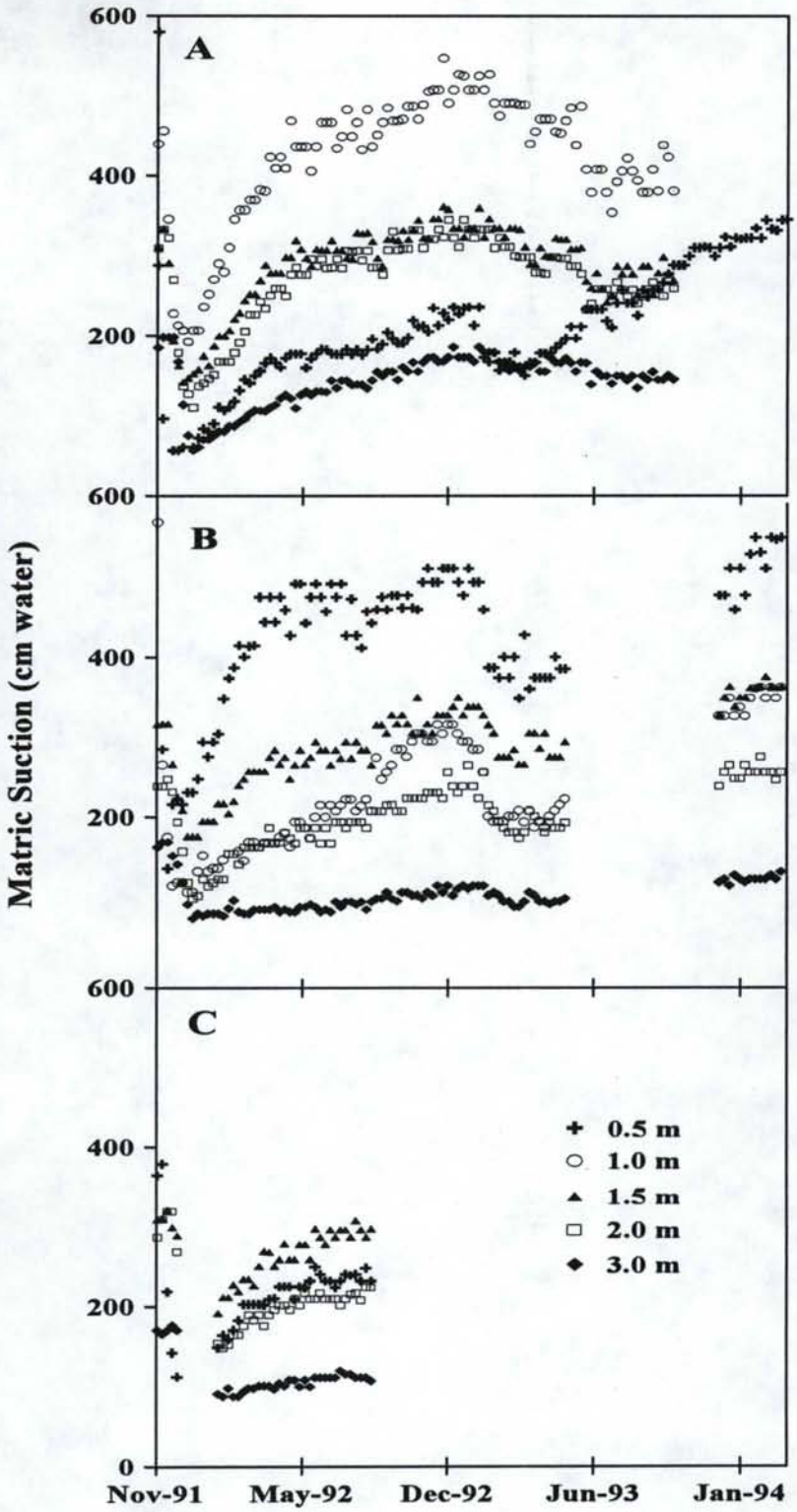

Figure 7. Matric suction below sections A, B, and C of L1. Exposure of the landfill base surface to the summer sun caused high initial matric suctions. Matric suctions dropped rapidly after loading, due to wetting of the matrix. In January, 1991, the matric suctions began to rise, marking the start of a long term drying phenomenon. Missing data are due to equipment malfunction.

more months before loading with cull onions and before measurements were initiated. Considerable variability in the magnitude of measured suctions existed between the banks of sensors in Sections A, B and C of L1; however, the general trend in each section was the same because the suction sensors were arranged close together in banks.

Coincident with the initial decomposition of cull onions and with the onset of winter precipitation, matric suctions decreased rapidly below L1. The profile below L1 showed a significant decrease in matric suction during the first two months of landfill operations; however, the profile did not become completely saturated. The profile reached its minimum matric suction of approximately 75 cm in January, 1992, three months after dumping started. Subsequently, the matrix below L1 began to desaturate. Matric suctions increased by about fifty percent at most depths by April, 1992 (Figure 7). Desaturation continued below L1 at a reduced but steady rate through February, 1993, at which time a second decrease in matric suction (wetting event) was observed. The second decrease in matric suction was much smaller in magnitude than the initial event. The greatest change in suction during the second event was a decrease of approximately 150 cm, measured at 0.5 m in Section B of L1. Matric suction sensors located at 3 m beneath the base of L1 maintained the lowest matric suction after the initial wetting and were affected least by the second event. The second decrease in matric suction was attributed to a significant increase in water level inside the landfill due to the accumulation of snow melt. The increased hydraulic loading increased infiltration through the landfill base. All depths beneath L1 showed renewed drying trends by March, 1993; these drying trends continued through the end of data collection in April, 1994.

Analysis of total heads beneath L1 sheds light on changes in the distribution of water during the evolution of the landfill. When considering unsaturated systems, total head is the sum of the matric potential head and the gravitational head (Hillel, 1982).

$$H_T = H_m + H_g \qquad [1]$$

By convention, matric potential heads are the negative of matric suction values measured in cm of water. Gravitational heads are the negative of the sensor depths when the landfill base is the reference elevation. Mean matric suction values for a ten day window surrounding the date of interest were used in the calculations. Total heads were calculated for Sections A and B of L1 for: (1) the condition prior to dumping onions (November, 1991), (2) the minimum matric suction associated with the first wetting event (January, 1992), and (3) the maximum matric suction following the first wetting event (November, 1992) (Figure 8). Matric suction data in Section C were not complete; therefore, total heads were not calculated.

The results of the analysis of H_T values beneath L1 are: (1) the landfill sediments remained unsaturated for the entire period of study, (2) a downward hydraulic gradient existed during the first two to three months of landfill operation, and (3) leaching during this period was minimal because solute fluxes were limited to unsaturated conditions. The persistent unsaturated condition of the profile is shown in Figure 8. All of the calculated values of H_T for the profile beneath L1 appear to the left of the dashed unit gradient line. The unit gradient ($dH_T/dz=-1$) describes the situation in which the matrix is saturated so that $H_m=0$. Under a unit gradient equation [1] becomes

$$H_T=H_g \qquad [2]$$

and flow is driven by gravity only. Values of H_T that plot to the left of the unit gradient line are influenced by $H_m<0$, and are therefore unsaturated. The persistence of unsaturated conditions through the initial wetting cycle suggests that the rate of infiltration of ponded landfill water remained below the infiltration capacity of the landfill sediments. Under ponded conditions, one would expect the rate of infiltration to be equal to the infiltration capacity of the sediments and the matrix to be under a positive pressure head rather than the measured matric suction head. A reasonable hypothesis for this contradiction is the formation of a low hydraulic conductivity organic layer at the interface of the landfill contents and the underlying sediments. This hypothesis is supported by research in which restrictive layers are

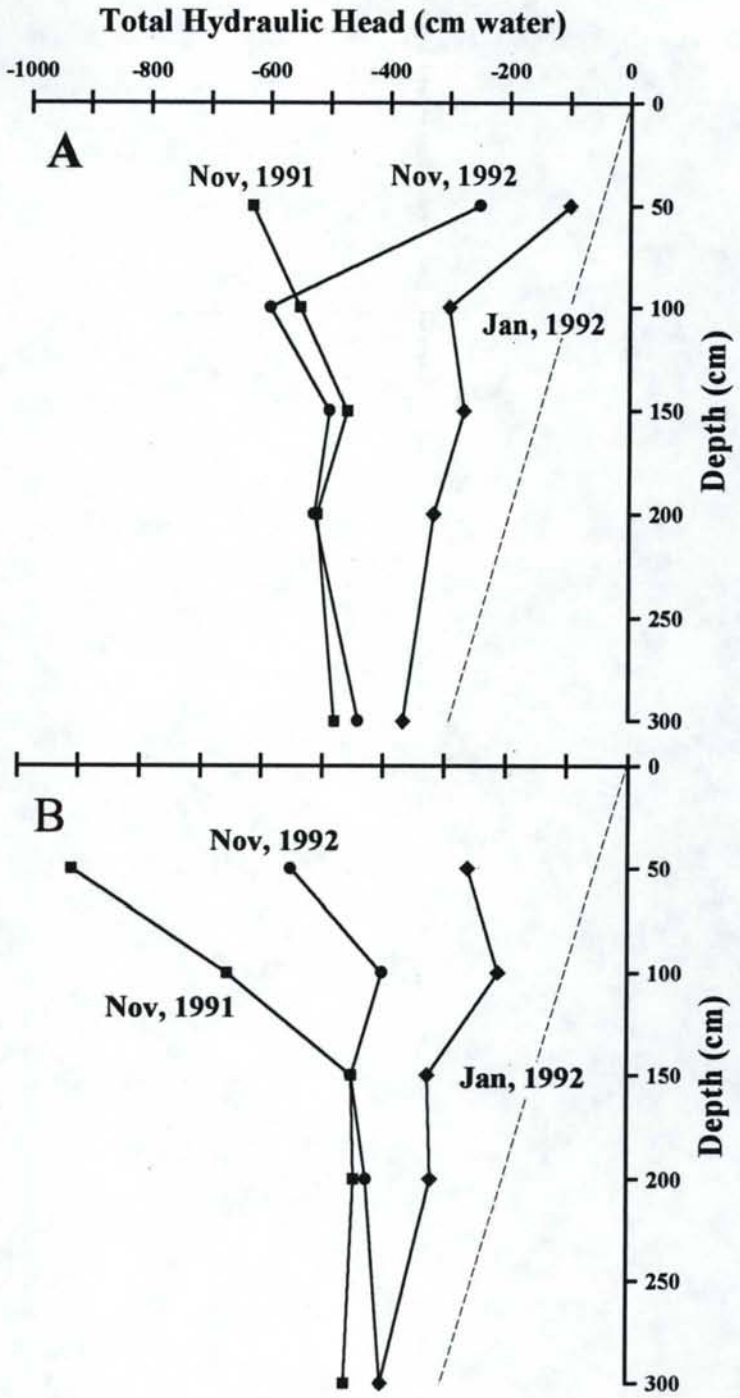

Figure 8. Total hydraulic heads beneath L1 sections A and B. The initial condition of the landfill was quite dry due to exposure of the landfill base to the summer sun. Concurrent with filling the landfill, hydraulic heads in January decreased significantly and indicated a downward gradient (the dotted line indicates a negative unit gradient). By November of the following year, after one year of landfill operation, the sediments below 1 m dried to near their initial condition. Datum is the elevation at the landfill base.

used to establish matric suctions in soils for the purpose of estimating unsaturated conductivity (Hillel, 1982). Clogging of soil pores by decomposing onion material could have established a low hydraulic conductivity layer which restricted flow through the landfill interface and maintained unsaturated conditions in the underlying matrix.

The distribution of H_T values beneath Sections A and B of L1 in November, 1991 indicate that H_T increased with depth (an upward hydraulic gradient) above 1.5 m. The upward gradient was probably associated with evaporation from the landfill base. Below 1.5 m the hydraulic gradient was near zero, suggesting minimal movement of water. Conditions were markedly different in January, 1992. Values of H_T were significantly less negative indicating an overall wetting of the profile. However, the profile remained unsaturated. In addition, the hydraulic gradient reversed direction indicating decreasing H_T with depth and a downward flux of water. The gradient was less than negative one above 1 m in Section A and between zero and negative one below 1 m. This type of curve is typical of redistribution of water in soil profiles and suggests that redistribution of a leachate pulse occurred beneath the landfill (Hillel, 1982). The hydraulic gradient above 1 m in Section B in January, 1992 was greater than zero, and the hydraulic gradient below 1 m was between zero and negative one. This curve is not easily explained; however, it may suggest that the wetting and drying of the profile was not uniform.

In November, 1992 the profile was significantly drier than in January, 1992, indicating that desaturation had occurred. The distribution of hydraulic heads below 1 m was very close to the initial (November, 1991) H_T distribution.

Similar to L1, relatively dry conditions were observed beneath L2 prior to loading of the landfill; however, no consistent trend in the matric suction data was apparent because of large variability in the values measured at each depth (Figure 9). The grid design of L2

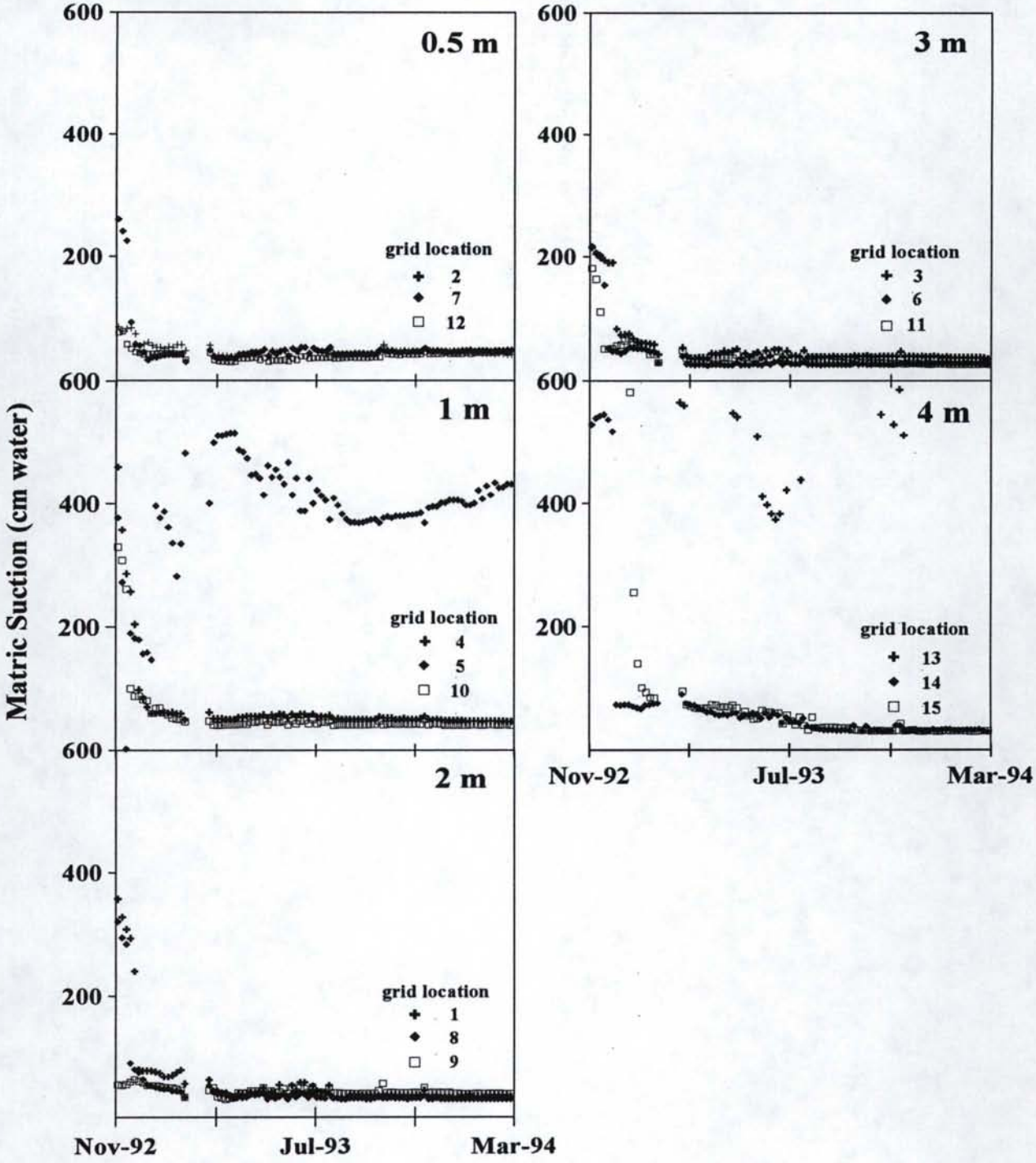

Figure 9. Matric suction below L2. The landfill sediments were initially dry. Matric suctions decreased rapidly after the landfills were filled with onions. The drying phenomenon observed beneath L1 was not apparent beneath L2. It is likely that the use of gypsum block sensors precluded the measurement of long-term desaturation.

probably resulted in more variability than L1 because sensors were not placed in banks, but were distributed across the experimental grid. Evaluation of L2 suction data does suggests that the initial period of downward gradient observed beneath L1 is a common event in cull onion landfills. The lack of evidence of desaturation in the sediments beneath L2 is believed to be due to failure of the gypsum block sensors. The gypsum block sensors may have degraded after prolonged exposure to the acidic leachate and failed to respond to desaturation of the matrix. The L2 matric suction data are therefore unreliable after the initial wetting event. The continued existence of standing water in L2 suggests, however, that the same inhibition of drainage that was observed beneath L1 also occurred beneath L2

In summary, the analysis of the matric suction data for L1 indicates that prior to loading, the system was close to a steady state, undergoing very little redistribution of matric water below 1 m. Subsequently, water released by cull onions and introduced from the local environment infiltrated into the sedimentary profile and was redistributed. Downward percolation of the landfill water was probably slow because the system did not reach saturation. Below 1 m the system returned near to its initial condition within one year. It is concluded that a flow barrier may have been responsible for the unsaturated condition beneath L1 because standing water is known to have existed in the landfill during the entire wetting and drying cycles. A restrictive layer may have been formed by the intrusion of decomposed cull onion material into the porous sediments underlying the cull onion landfills.

3.5 Saturated Hydraulic Conductivity of Landfill Sediments

The Hazen (1911) method for calculating saturated hydraulic conductivity using particle-size distribution data provides a first approximation for the upper limit of hydraulic conductivity in the landfill sediments. The Hazen method was used because it was derived empirically for coarse sedimentary material. The estimate was considered an upper limit for the landfill sediments because it did not account for reduction in flow area or tortuosity

associated with cementation in the matrix. The estimated value of saturated hydraulic conductivity using the mean particle size distribution of the landfill sediments is $5x10^{-3}$ cm/s. This value falls within the range of published values for silty sands and fine sands of $1x10^{-4}$ to $1x10^{-2}$ cm/s, suggesting that it is a reasonable estimation (Fetter, 1988).

Constant head permeameter measurements of the repacked cores provided an additional estimation of the saturated hydraulic conductivity. Saturated conductivity values were calculated using the Darcy relationship $q=-K_s dh/dz$, where q is the specific discharge (volume of water flowing through a unit cross sectional area per unit time), K_s is the saturated hydraulic conductivity, and dh/dz is the hydraulic gradient (Table 3). The range of K_s values was determined to be $1x10^{-4}$ cm/s to $4x10^{-4}$ cm/s. This range was also within the range of published values for unconsolidated sand. The laboratory permeameter analysis showed little variability in K_s values, probably due to the uniform pore-size distribution introduced during preparation of the repacked cores.

Table 3: Saturated Hydraulic Conductivity of Repacked Landfill Sediments

Sample	K_s (cm/s) (10^{-4})	Sample	K_s (cm/s) (10^{-4})	Sample	K_s (cm/s) (10^{-4})
1	3.66	6	2.40	11	2.15
2	2.79	7	1.58	12	1.34
3	3.77	8	1.97	13	2.94
4	2.81	9	2.03	14	1.46
5	3.09	10	1.02	15	1.30

In situ hydraulic conductivity measurements were obtained in L2 using a Guelph permeameter (Elrick *et al.*, 1984). Twenty-seven measurements of K_s ranged from $2x10^{-5}$ to $2x10^{-3}$ cm/s. Visual observations of areal heterogeneity at the landfill base suggested that the

value of K_S was a function of location on the landfill base. A preliminary geostatistical analysis was conducted using EPA software GEOEAS (v. 1.2.1) (USEPA, 1991) in order to identify whether the in situ hydraulic conductivity measurements were spatially dependent.

3.6 Spatial Dependency of Saturated Hydraulic Conductivity in L2

Geostatistical analyses typically consist of three components: (1) exploratory data analysis (EDA), (2) evaluation of spatial dependency (variogram) models, and (3) spatial interpolation (kriging). Exploratory data analysis is a review of the data in the form of post plots, contour maps, and batch statistics, in an attempt to identify obvious spatial trends in the data and to find useful data transformations. The goal of the EDA of the cull onion landfill data was to understand enough about the trends in K_S across the base of L2 to provide insight into the construction of an appropriate variogram model. The utility of kriged estimates were ultimately dependent on how well the variogram model fit the data; therefore, the analysis of the K_S data was directed at developing a realistic variogram model. This geostatistical analysis was preliminary because it utilized a relatively small number of samples to assess the variogram models.

A post plot of cull onion landfill hydraulic conductivity data showed little continuity of K_S over the base of L2 (Figure 10). Data values ranged from 0.11 to 1.3×10^{-3} in an apparently random manner. No obvious trends and many discontinuities were observed. A histogram of the raw data suggested that the K_S values were log-normally distributed (Figure 11). Hydraulic conductivity values are typically log-normally distributed (Domenico and Schwartz, 1990); therefore, a natural log transform was used in the initial variogram analysis (Figure 12). A variogram plot of the log transformed K_S data showed that the variogram does not vary significantly over the distance which a pair of K_S values were compared (lag spacing) (Figure 13, Appendix A). This type of variogram is commonly called a pure nugget effect. A pure nugget effect indicates: (1) no spatial dependency of Ks, or (2) spatial dependency is

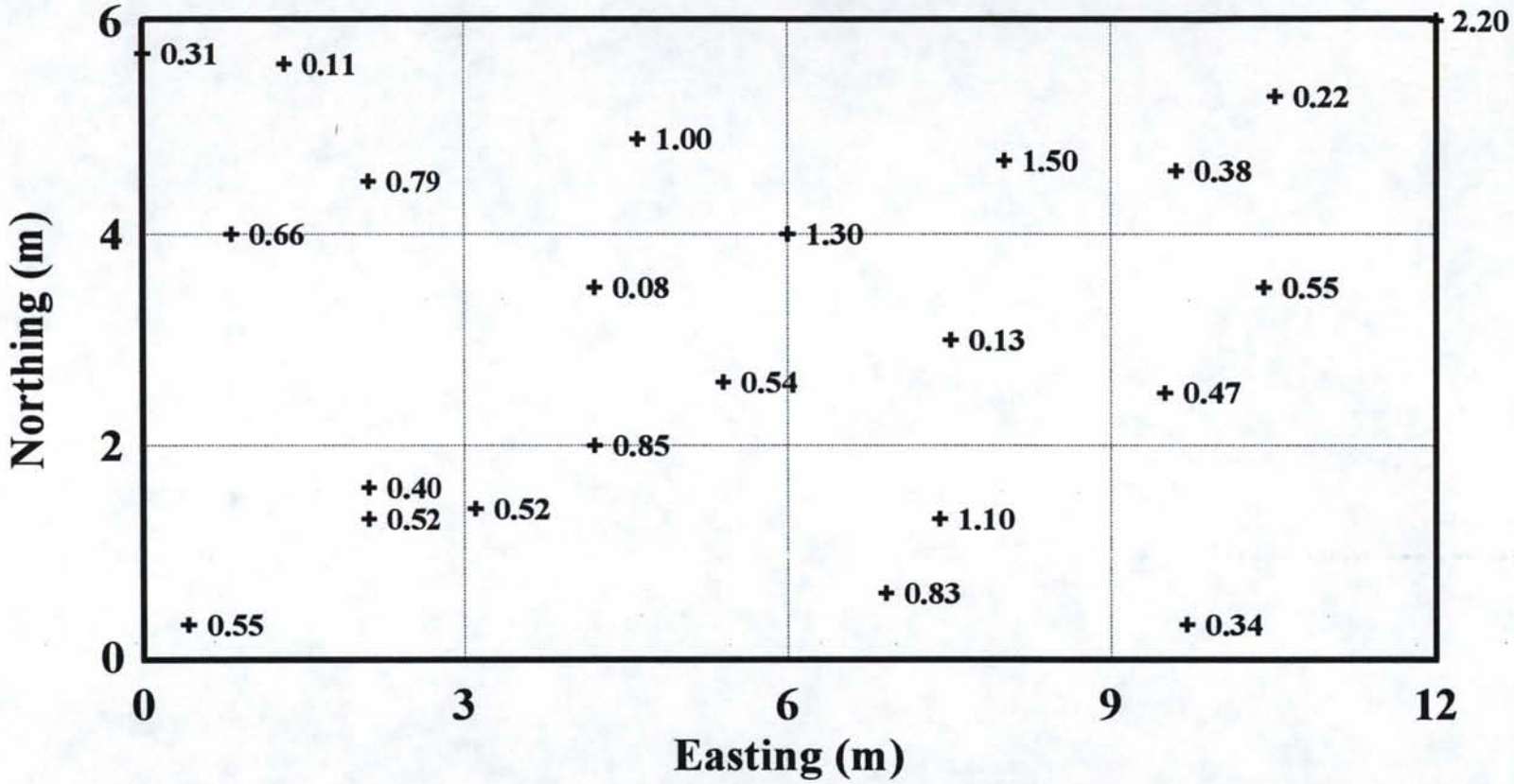

Figure 10. Post plot of saturated conductivity values measured at the base of L2. Conductivity values are given in cm/s (x0.001).

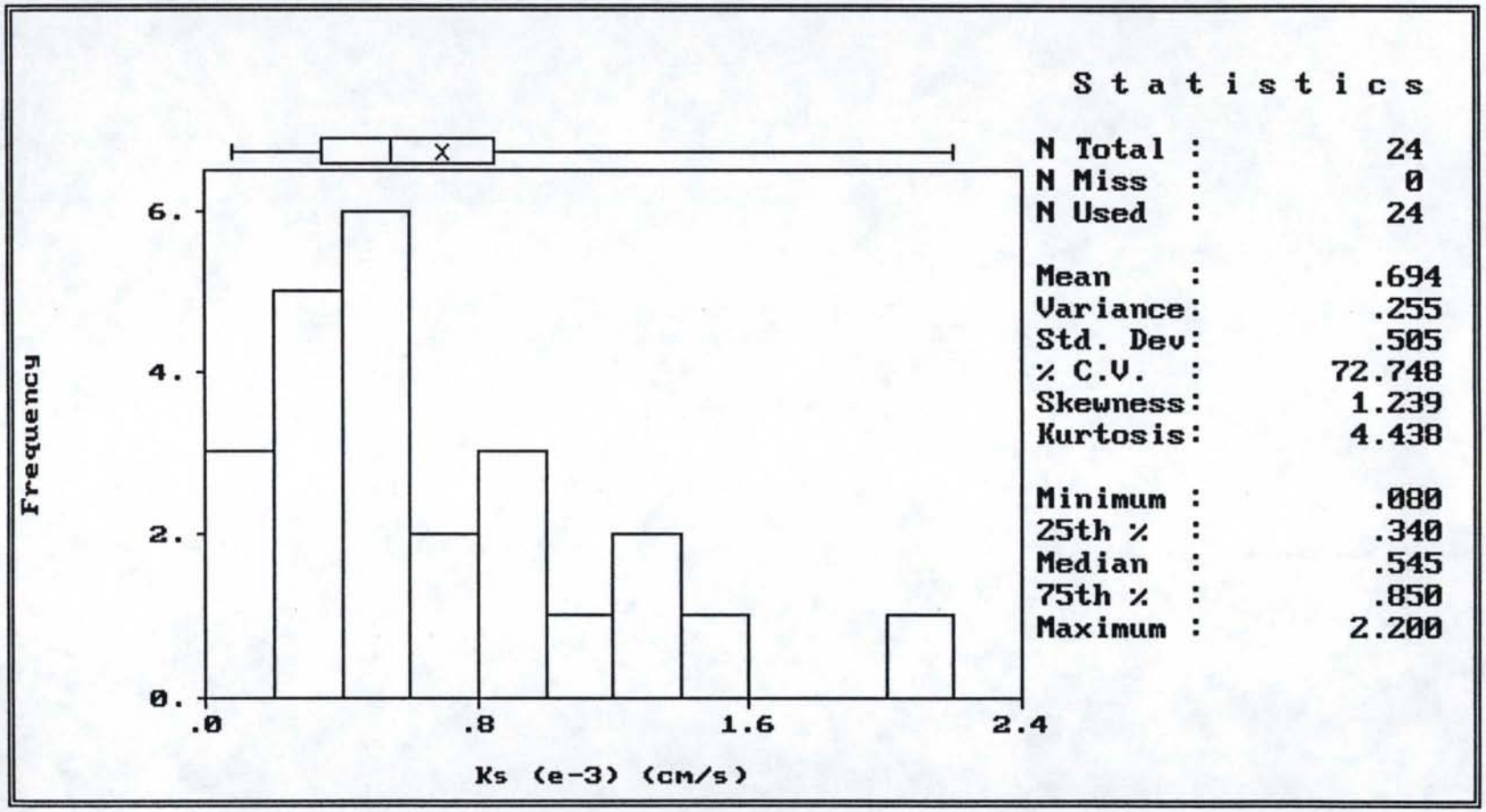

Figure 11. Histogram and univariate statistics for Ks values measured in L2. Twenty-four Ks measurements were analyzed for spatial dependency using GEOEAS software. The skewed distribution suggested that a logarithmic data transform was appropriate.

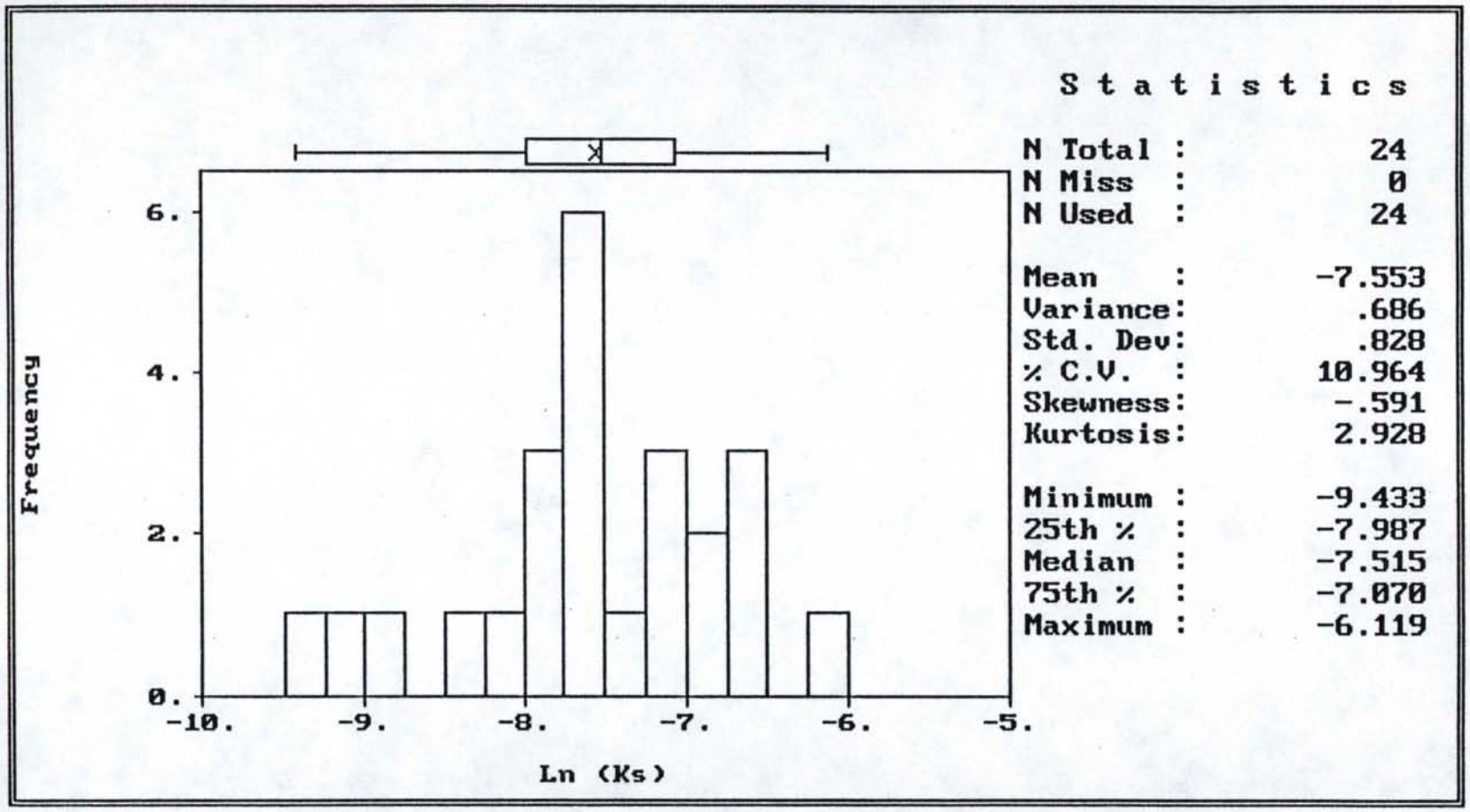

Figure 12. Histogram and univariate statistics for natural log transformed Ks values measured in L2. The log transform resulted in a more symmetric distribution that was approximately Gaussian.

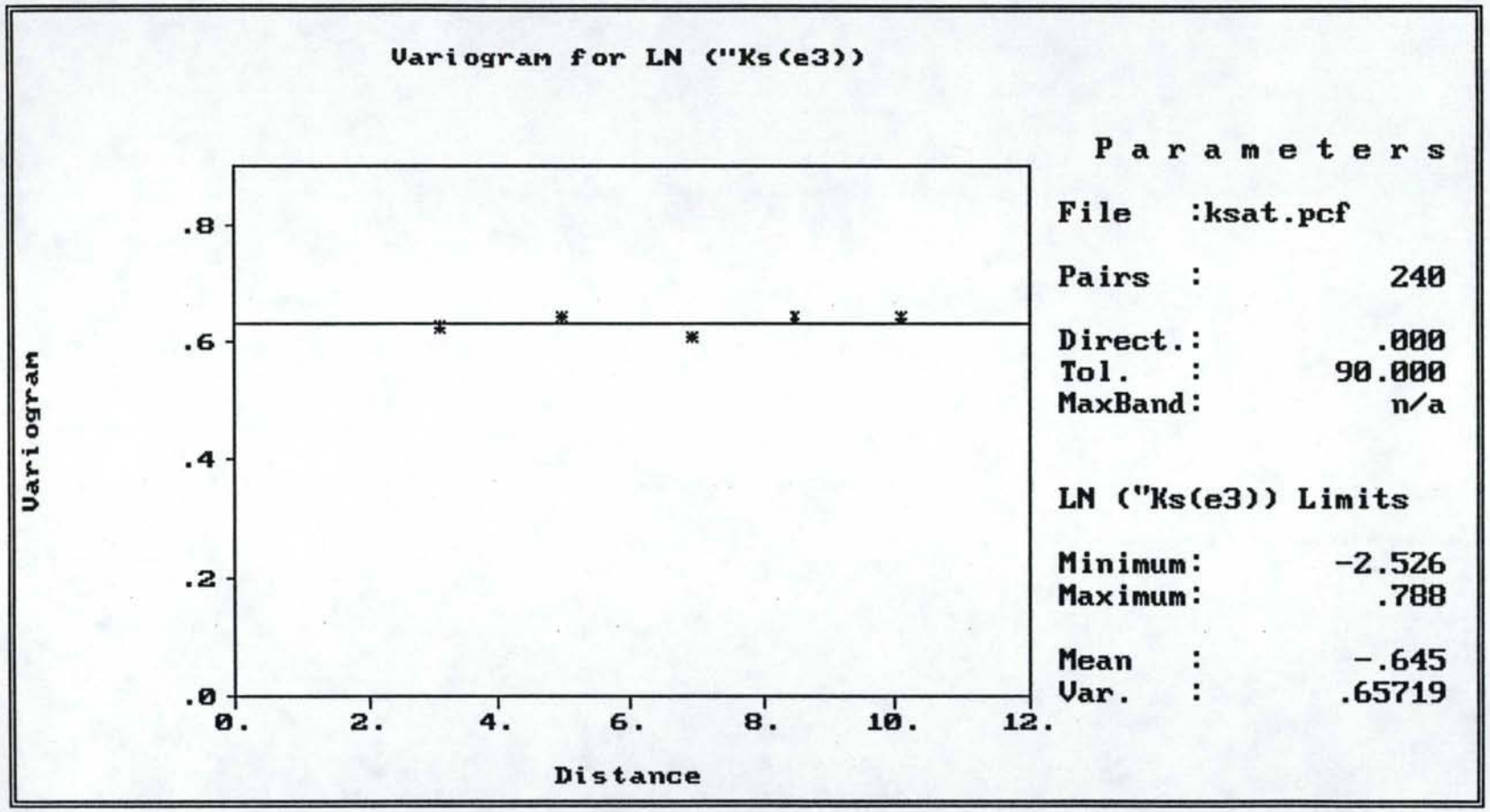

Figure 13. A variogram of natural log-transformed Ks data from the base of L2. Invariance of the value of the variogram over the entire range of lag distances suggests that Ks is not spatially dependent.

masked by measurement uncertainty (Miller, 1992). To further assess these possibilities a non-parametric kriging analysis was employed.

Indicator kriging was utilized for this analysis because, unlike standard kriging methods, it used the magnitude of the data values when calculating its kriged estimates. More specifically, indicator kriging allowed the variability in K_S to be treated as a function of the local mean value of K_S. The indicator transformed data behaved as expected, with means increasing as thresholds increased, and variances increasing to the 0.5 quantile threshold, then decreasing (Appendix A). A number of different indicator thresholds were applied to the data to optimize the resulting distribution of indicator values. The results of the best set of thresholds are reported in Appendix A. Analysis of indicator variograms showed that a properly configured variogram model could be produced by limiting minimum lag spacing to 3 m (Appendix A). This procedure simply removed from the analysis all interactions between K_S sampling locations that were less than 3 m apart. Considering that the study area was 6 m by 12 m, this procedure imposed too much restriction to make the model believable. The need for the 3 m minimum lag spacing further suggests that the distribution of K_S on the landfill base was purely random, or that any spatial variation was masked by measurement error. In addition to the minimum lag spacing restriction, the small number of data used in this preliminary analysis resulted in too few comparisons of K_S in each region of interest (lag bin). In full geostatistical analyses, 50 pairs of attribute values are often compared (Miller, 1992). Too few comparisons in a lag bin allowed small changes in lag bin boundaries to create large jumps in the associated variogram. These two limitations produced variograms that look deceivingly acceptable (Appendix A). In reality, the inverted covariance models that appear in the analysis output in Appendix A are of little practical use.

The morphology and genesis of the laminated lacustrine sediments suggest that saturated hydraulic conductivity depends on location within the cull onion landfills. The

preliminary geostatistical evaluation indicated that spatial dependency was not a factor in the distribution of Ks values in L2. On the scale of the experimental cull onion landfill, variability of saturated hydraulic conductivity across the landfill's base appeared to be independent of location, therefore, best described by univariate statistics.

3.7 Unsaturated Hydraulic Conductivity of Landfill Sediments

Matric suction data have provided evidence that the sediments directly underlying the experimental cull onion landfills were unsaturated during the first three years of operation. A parametric water retention model was used in order to calculate realistic leachate velocities that accounted for the unsaturated condition. The RETention Curve (RETC) model (van Genuchten *et al.*, 1991) was employed to evaluate empirical retention parameters used in an analytical soil-water retention equation and to apply the parameters to an estimation of the hydraulic conductivity function for the landfill matrix.

The water retention function given by van Genuchten (1980) is:

$$S_e=1/[1+(\alpha h)^n]^m \qquad [3]$$

where S_e is the effective degree of saturation, h is the pressure head, and α, n, and m are empirical parameters that affect the shape of the retention curve. The value $1/\alpha$ is often interpreted as the air-entry pressure head for the matrix of interest. The effective degree of saturation (S_e) is commonly referred to as the reduced water content given by

$$S_e=(\theta-\theta_r)/(\theta_s-\theta_r) \qquad [4]$$

where θ_r is the residual water content, θ_s is the saturated water content and θ is the existing water content of a soil. The physical meanings of θ_r and θ_s are limited in the context of the RETC model and are effectively used as empirical parameters in the curve fitting procedure (van Genuchten and Nielsen, 1985; Luckner *et al.*, 1989; van Genuchten *et al.*, 1991).

In accordance with the procedure suggested by van Genuchten *et al.* (1991), initial modeling runs were conducted that attempted to fit all of the parameters simultaneously. Subsequent runs were conducted with fixed values for those parameters that exhibited high correlation, or whose values were well known a priori. Van Genuchten *et al.* (1991) suggested that poorly defined data sets, such as those commonly obtained in the field, exhibit values of m and n that are often strongly correlated. This situation was reported to lead to poor convergence and ill-defined parameter values with large confidence intervals. According to van Genuchten *et al.* (1991), more stable results have been obtained for incomplete data sets by restricting the values of m and n. The water retention measurements for the repacked landfill sediments were typical of such incomplete data because they defined the retention function only in the wet region. The RETC model was run with the limitation m=1-(1/n), as suggested by van Genuchten *et al.* (1991). In addition, θ_r was set to 0.01. The RETC model was designed to fix the value of θ_r at zero if iteration forced the parameter below 0.01. This situation occurred repeatedly during the various modeling runs. A review of published values of θ_r for similar materials indicated that a value between 0.01 and 0.03 is more appropriate for the repacked sediments (van Genuchten *et al.*, 1991). The model was run with α, θ_s and n variable. The model forced α to very small values, while allowing the values of θ_s and n to converge on reasonable values, regardless of their initial parameter estimates. A review of published values of α and n suggested that α ranges from 0.02 /cm for loamy soils to 0.15 /cm for unconsolidated sands, and n ranges from 1.2 to 2.7 for the same materials (Rawls *et al.*, 1982; Carsel and Parrish, 1988). These ranges are not well established; however, they are adequate upper and lower limits for the repacked landfill sediments.

The model was run for a number of α values within the published range. It was found that the value of α that produced the most reasonable combination of θ_s and n was 0.02 (Figure 14). The resultant least-squares curve fit converged on θ_s=0.40 and n=1.15, and had an R^2 value of 0.97. This convergence was considered adequate because the value of θ_s was

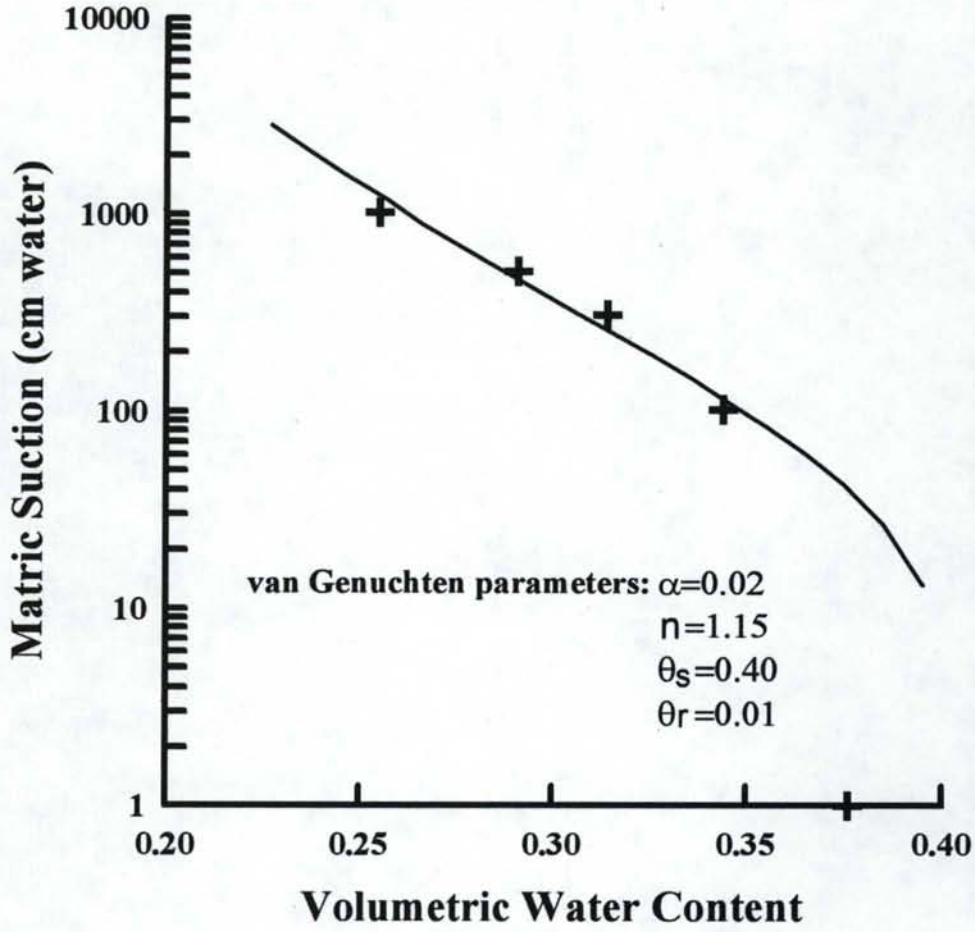

Figure 14. Water release data and RETC parameter estimation. The curve is fitted to water release data (+) from repacked landfill sediments using the method of van Genuchten. R-squared for the fitted curve is 0.97.

determined experimentally to be 0.38 +/- 0.02 (Section 3.2). However, the fixed value of α and the convergent value of n were lower than expected. They also corresponded to values associated with the retention functions of clayey material (Rawls *et al.*, 1982; Carsel and Parrish, 1988). Some of this difference may be real. The landfill sediments may have retention characteristics which vary significantly from the limited data base tabulated in the literature. Deviation from the published values may also have been introduced by repacking the landfill sediments. Altering the physical structure of the material so that it no longer represented an intact sample may have caused deviation from reported values, especially if those values were based on structured soils. Unfortunately, resolution of the reasons for lower α and n values was not possible. It was assumed for this thesis that the model parameters provided an adequate representation of the sediments beneath the cull onion landfills.

The unsaturated hydraulic conductivity function was estimated by RETC using the fitted water retention parameters described above. The RETC software used the method of Mualem (1976) to predict the hydraulic conductivity function. The basic relationship is given by

$$K(S_e)=K_S S_e^{l}[f(S_e)/f(1)]^2 \qquad [5]$$

where $f(S_e)= \int_1^{Se} 1/h(x)dx$, K_S is the saturated hydraulic conductivity, S_e is the degree of saturation, l is a pore-connectivity parameter taken to be 0.5 for many soils, and h(x) is the pressure head, a function of the vertical distance parameter x. For the restricted case in which m=1-(1/n), where m and n are the same empirical parameters described above, and K_S is measured at a known θ_S, equation [5] becomes (Luckner *et al.*, 1989).

$$K(S_e)=K_S S_e^{l}[1-(1-S_e^{1/m})^m]^2 \qquad [6]$$

This predictive equation is valid assuming that K_S is a well-defined soil parameter; however, with the exception of some coarse-textured soils, repacked cores, or other soils characterized by relatively narrow pore-size distributions, saturated conductivity is generally difficult to

quantify (van Genuchten *et al.*, 1991). An unsaturated hydraulic conductivity value should be associated with a known water content because the location and shape of the conductivity curve is sensitive to small changes in K_s (Stephens and Rehfeldt, 1985; van Genuchten *et al.*, 1991).

Saturated conductivity was the only matched data point available in this analysis; therefore, equation [2] was solved. The use of K_s as the match point was justified by three observations: (1) calculated values of $K(S_e)$ were probably not significantly affected by small changes in K_s because the range of matric suctions of interest was relatively small, between 0 and 1000 cm of water, (2) the matrix for which equation [2] was solved consisted of the coarse textured, repacked soil cores described in Section 3.2, and (3) the values of K_s for the repacked cores exhibited relatively small variability and the values of K_s measured in situ were also reasonable.

Results from the RETC estimation of the hydraulic conductivity function using the range of K_s values measured in situ are shown as curves A and B in Figure 15. Considering curve A, the hydraulic conductivity is 1×10^{-3} cm/s at a matric suction from 0.1 cm of water . Increasing the matric suction to 10 cm of water, the hydraulic conductivity decreases to 5×10^{-4} cm/s. At a matric suction of 500 cm (the matric suction measured in the shallow matric suction sensors beneath L1), the hydraulic conductivity is approximately 7×10^{-7} cm/s.

3.8 Calculated Flow Velocities

Darcy's equation for flow through porous media can be extended to unsaturated conditions as

$$q = -K(\psi)dh/dz \qquad [7]$$

where q is the specific discharge (flux), $K(\psi)$ is the hydraulic conductivity as a function of matric suction, and dh/dz is the hydraulic gradient (Hillel, 1982). Flow velocities are typically

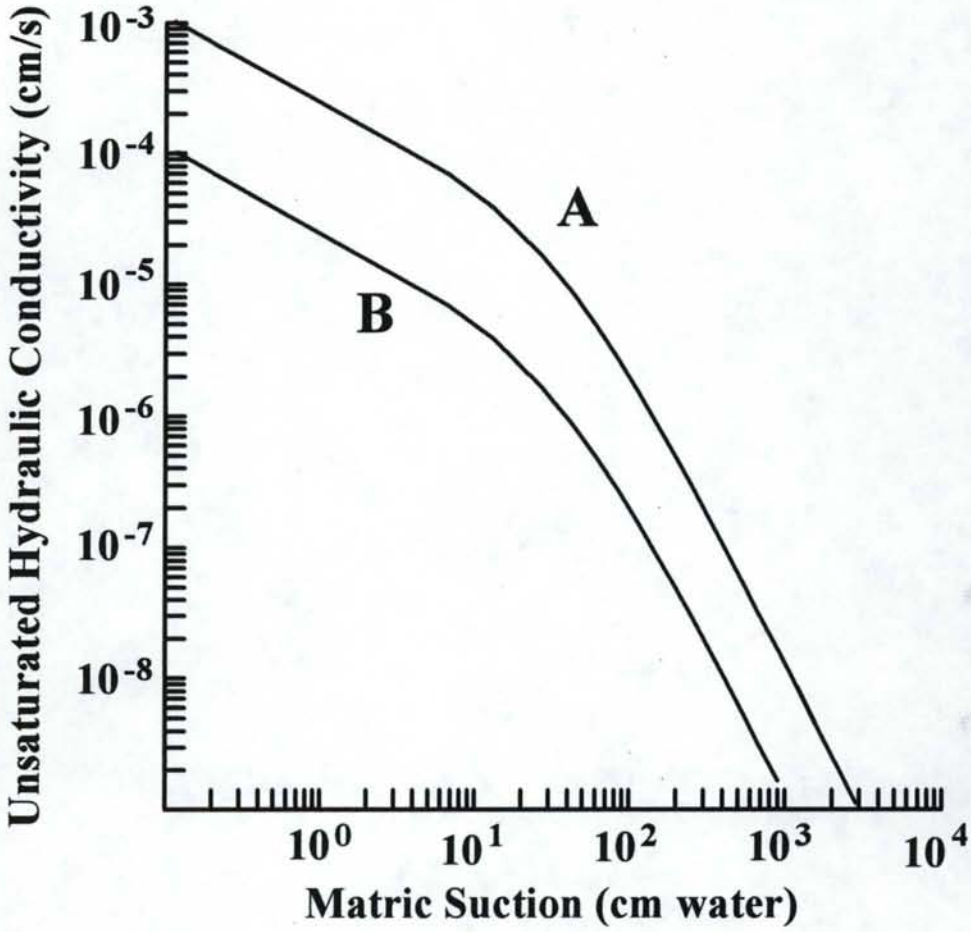

Figure 15. Unsaturated hydraulic conductivity functions calculated by RETC. Parameters estimated by a water retention curve fitting procedure were used to calculate hydraulic conductivity functions for the range of Ks values measured in situ.

evaluated by scaling q by the actual cross-sectional area available for flow, so that $\upsilon=q/\eta e$, where υ is the average linear pore velocity, η is the total porosity, and e is the fraction of the total porosity available for flow. The product ηe is commonly referred to as the effective porosity and has a value less than or equal to total porosity. The difference between total porosity and effective porosity depends largely on the structure of the porous media. Effective and total porosities are usually within 80 percent of each other in coarse-grained unconsolidated sediments (Norton and Knapp, 1977; Domenico and Schwartz, 1990). However, in some fine-grained sediments, effective porosity can be up to an order of magnitude less than the total porosity (Domenico and Schwartz, 1990).

No experimental data were available from which to estimate effective porosity for the landfill sediments. Therefore, a conservative estimate of one-half saturated porosity was assumed. Saturated water content in the landfill sediments ranged between 0.36 and 0.40 (Section 3.2), so that effective porosity was estimated to be 0.2. Calculations for steady state gravitational drainage through sediments were based on the range of K_S values measured in situ ($1x10^{-4}$ to $1x10^{-3}$ cm/s). The assumptions for these calculations are: (1) a saturated matrix, and (2) no ponding, in which case dh/dz is unity and equation [7] becomes $q=K_S$. The value for effective porosity of 0.2 predicted gravitational drainage through the landfill base at a rate of $5x10^{-4}$ cm/s to $5x10^{-3}$ cm/s.

This range of flow velocities is equivalent to between 3 and 30 m of water leaving the landfill per month. Ponded water above the surface of the landfill base would increase the rate of drainage. These estimates show that the cull onion landfill matrix has the potential to leach considerably more water per month than the 2 m that was supplied by the decaying onions and surface runoff. Furthermore, the calculations suggest that water would not remain ponded in the landfill, but would move downward through the underlying profile very quickly.

The results of the RETC model were used to estimate unsaturated hydraulic conductivity values and solve the flow equation for unsaturated conditions because L1 matric suction data showed that landfill sediments remained unsaturated over the period of study. Gradients beneath Sections A and B of L1 during the initial wetting were estimated to be 0.44 and 0.84, respectively (Section 3.4). The range of ψ at the time of greatest saturation during the wetting event was 75 to 200 cm of water (Figure 7). These values were used to obtain $K(\psi)$ estimates from the RETC curves. Unsaturated hydraulic conductivity values were estimated to be between $5x10^{-7}$ and $4x10^{-6}$ cm/s (Figure 15). Although effective porosity is a function of moisture content, no information was available about the functionality for the cull onion landfill sediments; therefore, values of ηe associated with saturation were used. Assuming an effective porosity between 0.2 and 0.4, flow velocities during the initial wetting event were between $5x10^{-7}$ and $2x10^{-5}$ cm/s. Assuming a wetting period of 60 days, these estimated flow velocities suggest that leachate traveled between 3 and 100 cm during the initial wetting event.

Gradients measured beneath Sections A and B of L1 during the steady-state period reached after approximately one year of landfill operation, were close to zero. These gradients were significantly less than those measured during the initial wetting period (Figure 8). When the gradients approached zero, flow velocities became very small. In addition, the range of ψ during the steady-state period was 200 to 500 cm of water. These values yield $K(\psi)$ values of $6x10^{-8}$ to $5x10^{-7}$ cm/s, approximately an order of magnitude less than those estimated for the initial wetting (Figure 15). Assuming that both $K(\psi)$ and the hydraulic gradient were one order of magnitude less than the values for the initial wetting period, the average linear flow velocity of water movement through the profile was reduced by approximately two orders of magnitude.

3.9 Estimation of Flow Velocities Using a Bromide Tracer

Bromide is an anion that has been used as a conservative tracer to model the movement of water through porous media (e.g. Wild and Babiker, 1976; Davis *et al.*, 1980; Bowman, 1984; Gish *et al.*, 1986; Jury *et al.*, 1986). Bromide was particularly useful with respect to leaching beneath cull onion landfills because it mimicked the transport of nitrate through the system. The goal of the Br^- tracer study was to evaluate the travel time of the wetting front through the sedimentary profile underlying the cull onion landfills.

Water samples drawn from within the landfills were dark brown in color and contained large amounts of organic matter that rendered the samples opaque. Cull onion leachate samples drawn from the underlying matrix were also colored by organic matter. Samples appeared to contain less organic material at greater depths below the landfills. Leachate samples drawn from beneath the cull onion landfills were dark yellow and opaque at 0.5 m, light yellow and less opaque at 1 m, and relatively clear at 2 m and below. The "rotten onion" odor of the leachate also decreased in intensity as depth below the landfills increased. Both of these qualitative observations suggest that organic matter was being physically filtered out and/or microbially degraded as the leachate moved through the matrix.

The influence of variable concentrations of organic material on the activity of Br- ions in the leachate solution has not been determined. Two calibration procedures were developed in order to accurately calibrate the selective Br^- electrode used to measure Br^- ion activity in the leachate. Calibration standards were prepared in both millipore filtered distilled water and in a highly organic landfill water sample to determine if the organic matter affected electrode response. These solutions were selected because they bracketed the range of organic matter concentrations found in the leachate. Potassium bromide standard solutions were prepared at 8, 80 and 1000 mg/l in the distilled water and 1, 50 and 500 mg/l in the landfill water. The ion specific electrode was immersed in the distilled water standard solutions, allowed to stabilize,

and the millivolt readings recorded. The least concentrated distilled water standards were run first, followed consecutively by more concentrated standards. The same procedure was used for the landfill water solutions. These initial calibrations were run a number of times in fresh standard solutions in order to verify the results. The standards were also measured periodically after each set of approximately 30 samples (corresponding to each sampling event). The periodic calibrations were conducted immediately prior to polishing the ion specific electrode and immediately after polishing the electrode to determine if organic matter was coating the chemically active surface of the electrode. Results from the initial and periodic calibrations suggested that: (1) electrode response was greater in distilled water at low Br^- concentrations than in landfill water at low concentrations, and (2) the presence of dissolved/colloidal organics in the landfill water decreased electrode response over time. However, electrode response was restored in the landfill water samples when the electrode was polished. This observation suggested that the concentrations of organics in the samples affected the analysis and that organic material coated the electrode surface and attenuated its response to the Br^- in solution. As a result, the electrode was polished after every 30 samples.

The data from the periodic distilled water and landfill water calibrations were compared statistically to evaluate which data set better represented the total Br^- concentration in the leachate (Figure 16). Visual observation of the landfill water data suggested that the electrode accurately measured Br^- concentrations at high solution concentrations, but significantly underestimated the total amount of Br^- in the leachate at low solution concentrations. At high solution concentrations (500 mg/l and 50 mg/l), Br^- probably overwhelmed the effect of the organic matter in solution, while at 1 mg/l the organic matter significantly affected the measured Br^- concentration. Linear regressions of the distilled water standard data and the landfill water data had high R^2 values (0.97 and 0.91 respectively). Results of a statistical comparison of the regressed data indicated that the regression lines were not statistically coincident at the 0.5 significance level. The landfill water data regression

was probably weighted by the underestimated Br^- concentrations measured in the 1 mg/l standard and, therefore, too steeply sloped. When both the distilled water and landfill water calibration regressions were applied to the actual data, the Br^- concentrations given by the distilled water calibration were much more reasonable than those given by the landfill water calibration. The regressions and statistical comparisons were repeated without the 1 mg/l landfill water data. Results of this comparison showed the regressed lines were coincident at the 0.5 significance level. It was decided that although high concentrations of dissolved and colloidal organic material affected the measurement of Br^- concentration in the landfill leachate, the distilled water calibration procedure adequately described the total concentration of Br^- in the leachate.

Leachate samples from sampler locations in the base of L1 that were not tagged with tracer were analyzed to establish background levels of Br^- in the solution. Background concentrations measured in the landfill leachate ranged from 15 to 25 mg/L. These relatively high concentrations were probably the result of electrode interference from naturally occurring chloride in the matrix. The lowest concentration of Br^- that was measured in all of the samplers below the base of L1 was 2.5 mg/l. Overestimation of the background Br^- concentration was not significant in the analysis because peak concentrations of the tracer in the leachate were at least an order of magnitude greater. Background Br^- concentrations were assumed to be the same for L2.

Analysis of tracer concentrations in the leachate shows that Br^- passed through the profiles beneath L1 and L2 over the period of the study (Figure 17). Of the three samplers located at 0.5 m beneath the base of L1, only the Section C sampler had concentrations significantly above the background level. It is likely that the breakthrough curve moved past the other two units at 0.5 m before sampling began in January 1992. The same situation was apparent in Sections A and B at the 1 m depth. The concentration of Br^- at 0.5 m and 1 m

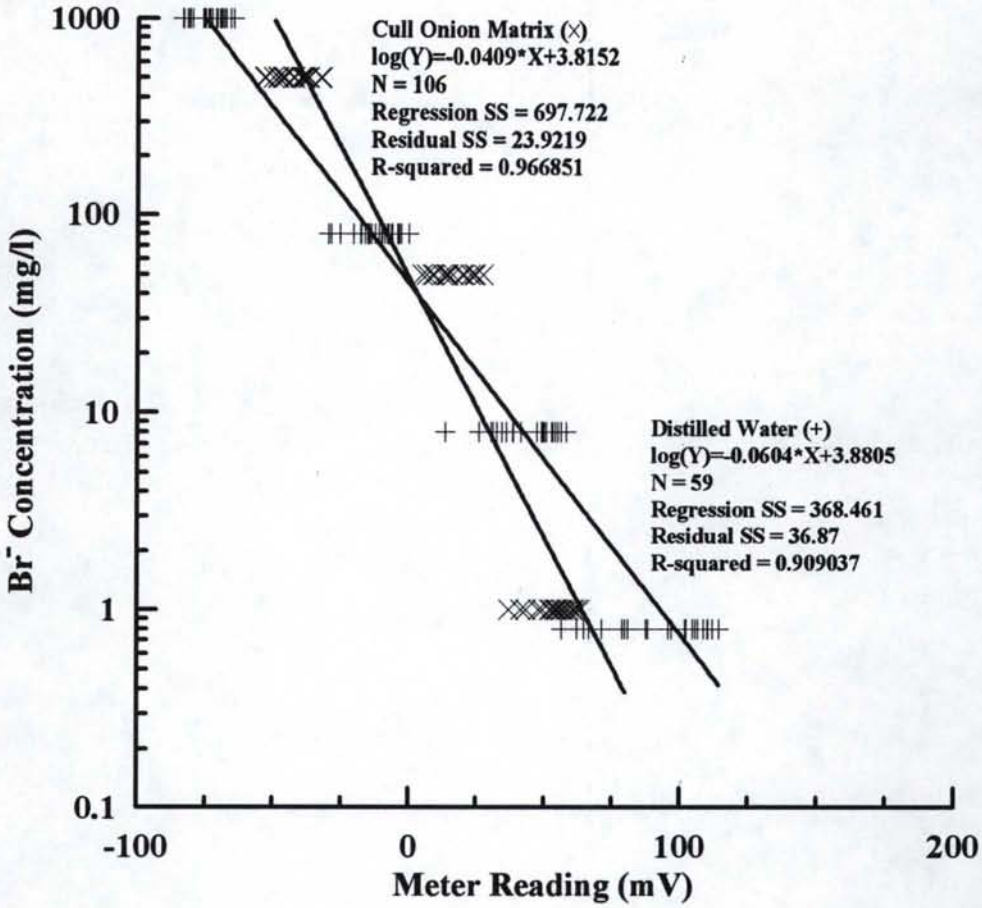

Figure 16. Standard curves for measurement of bromide tracer concentrations. The effect of organic material in the leachate was evaluated by comparing electrode response to bromide standards in the cull onion leachate and in distilled water. The data were compared statistically.

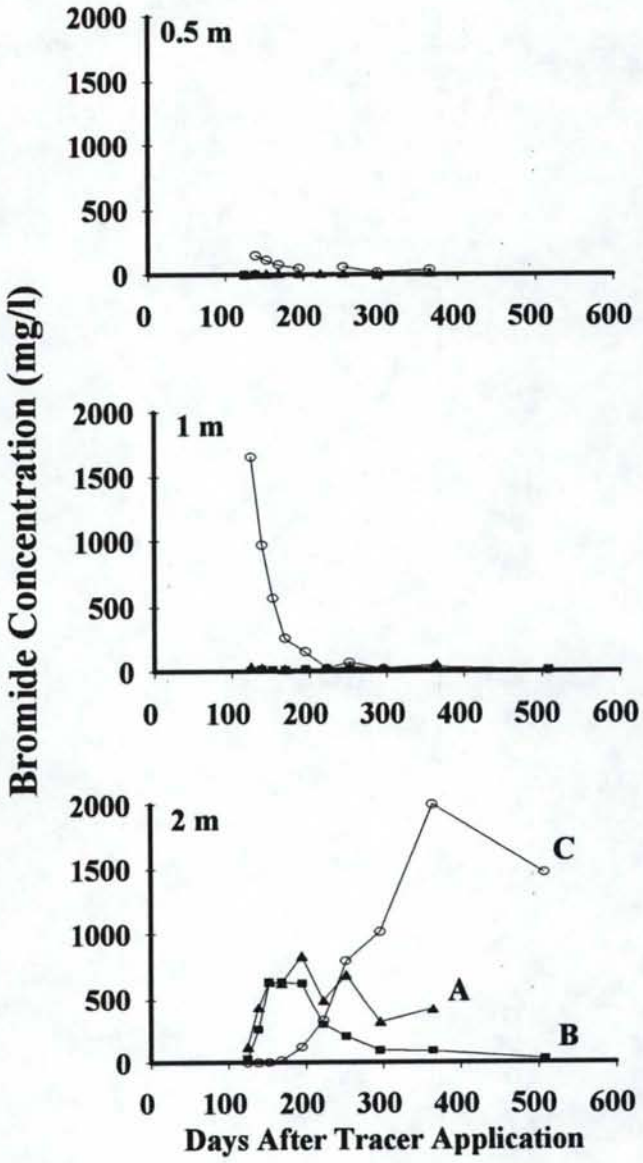

Figure 17. Bromide tracer concentrations at 0.5, 1 and 2 m beneath sections A, B, and C of L1. Disconnected data indicate when samples were not obtained.

beneath Section C decreased with each successive sampling period, indicating that indeed the maximum tracer concentration had already passed the sampling point. The downward gradient that was coincident with an increase in saturation of the profile beneath L1 occurred during the first two to three months after cull onion loading. This gradient was probably responsible for the rapid movement of tracer past the 0.5 m and 1 m samplers. In contrast, all sampling locations at 2 m showed definite increases and subsequent decreases in tracer concentration that were attributed to tracer breakthrough at two meters beneath L1. Maximum concentrations appeared at the 2 m depth 150 to 390 days after application of the tracer in September, 1991. Average flow velocities were calculated by dividing the measurement depth by the time of travel (the lapsed time between tracer application and the occurrence of the maximum tracer concentration). The range of average flow velocities calculated from the 2 m breakthrough curves range from $1x10^{-6}$ to $6x10^{-5}$ cm/s.

Analysis of tracer concentrations in the leachate from beneath L2 suggested similar general patterns of leachate movement; however, tracer concentrations were more variable than those beneath L1 (Figure 18). The reason for the higher degree of variability in L2 Br^- concentrations was probably that the samplers were distributed throughout the experimental grid in L2 rather than arranged in banks as in L1. Unlike the L1 tracer data, L2 leachate sampling began before the wetting front passed through the shallow soil-water samplers. As a result, maximum tracer breakthrough was evident in some of the 0.5 m samplers (2A, 7B, 12A and 12B); however, the values were inconsistent and widely scattered in time. Tracer travel times appeared to be less than 200 days at the 0.5 m depth. The minimum average flow velocity was calculated using the greatest Br- travel time at 0.5 m. The calculated flow velocity was $3x10^{-6}$ cm/s. Breakthrough curves are not well defined in any of the L2 data obtained from samplers deeper than 0.5 m, because samples were not analyzed for an adequate length of time to capture the passing of the peak tracer concentration.

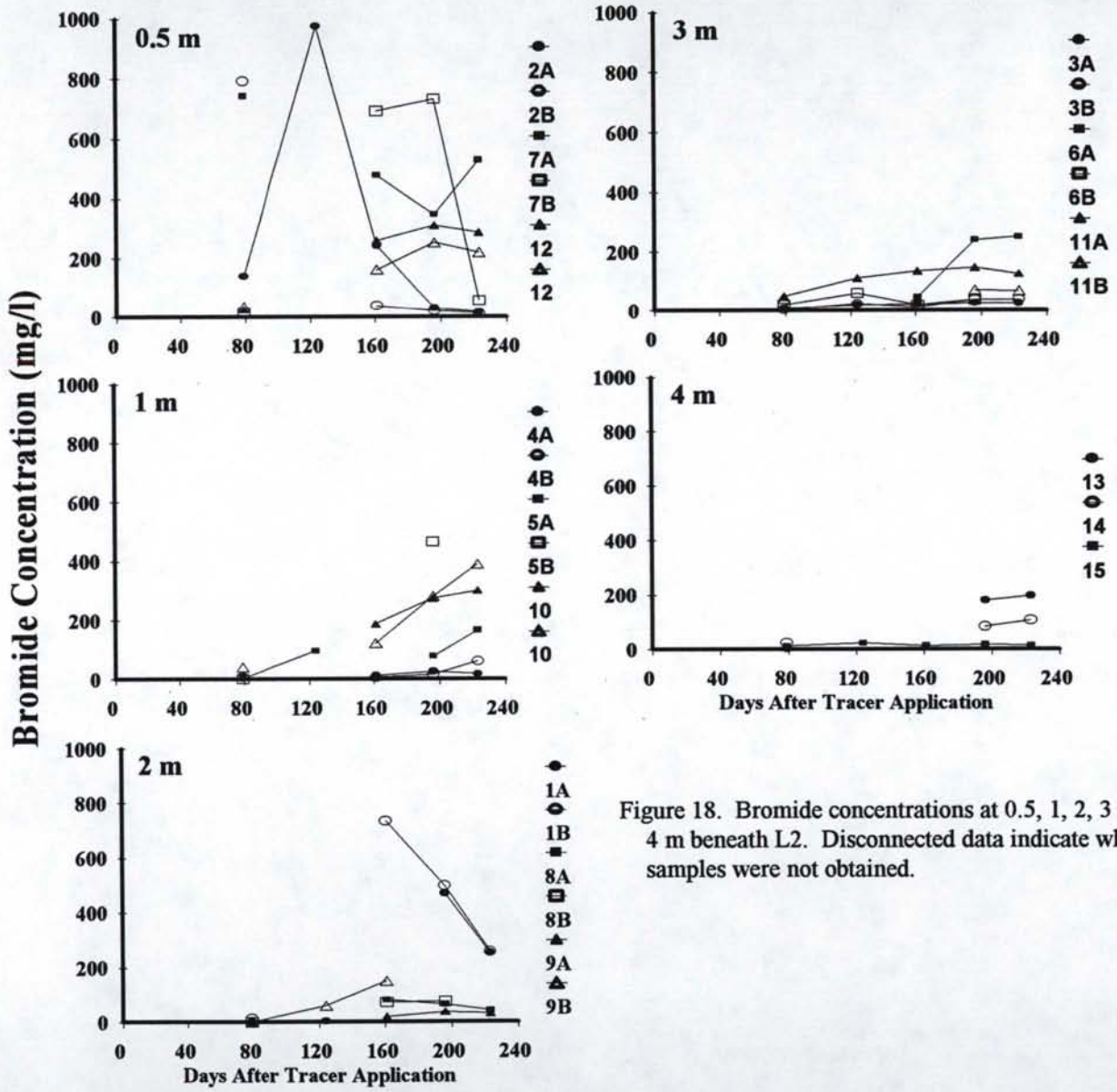

Figure 18. Bromide concentrations at 0.5, 1, 2, 3 and 4 m beneath L2. Disconnected data indicate when samples were not obtained.

3.10 Analysis of Leachate Flow Velocities

This tracer analysis shows: (1) the potential existed for nitrate to move at least 2 m beneath the base of the cull onion landfills during the first five months of landfill operation, (2) average flow velocities for the first five to 13 months of landfill operation were on the order of 10^{-6} to 10^{-5} cm/s in the upper 2 m of the profile, and (3) significant variability in flow velocities was a factor in the transport of tracer in the profile. The tracer data from both landfills suggests that if significant concentrations of nitrate existed in the water that was retained in the landfills, measurable nitrate would have leached through the instrumented volume at some point during the experiment. The variability in peak concentrations and the temporal inconsistency in the tracer concentrations, especially in the L2 data, suggests a complicated distribution of flow paths in the profile. The first two results are in agreement with the estimated leachate flow velocities and travel distances calculated using the unsaturated conductivity values from the RETC model (Section 3.8). Unsaturated conductivities calculated using the RETC model indicate that leachate probably moved 3 to 100 cm below the landfills during the two to three month initial wetting event. This range of values is broad enough to account for the variability in the tracer concentrations that were measured beneath the landfills.

CHAPTER 4

ANALYSIS OF LANDFILL NITROGEN DATA

4.1 Introduction

The purpose of this thesis, to determine whether cull onion landfills are a potential source of nitrate contamination of groundwater, made characterization of the nitrogen composition of landfill leachate an important focus of the study. The nitrogen characterization consisted of: (1) evaluation of nitrogen loading in the landfills with respect to NO_3-N, NH_4-N and TKN, (2) periodic analysis of the nitrogen distribution of the contents of the landfills, and (3) periodic analysis of the nitrogen distribution of the landfill leachate. Leachate drawn from stand-pipes and soil-water samplers was analyzed for NO_3-N, NH_4-N and TKN in order to ascertain the fate of nitrogen. Organic nitrogen was targeted because most of the nitrogen in vegetable matter is retained in the organic form, and because it is the primary nitrate precursor in concentrated agricultural waste. Ammonium was targeted because it is the secondary nitrate precursor in concentrated agricultural waste; therefore, ammonium is a potential storage form of nitrogen.

4.2 Nitrogen Loading

The relative amount of dry matter in cull onions has been reported to depend on the particular variety (Levi *et al.*, 1990). Red and white onions have been reported to consist of 20 to 25 percent dry matter, while Spanish Sweets and other yellow varieties contain 8 to 10 percent dry matter by weight (Levi *et al.*, 1990). Assuming that the average percent dry matter in cull onions ranges from 10 to 20 percent and applying the bulk density estimate described in Section 2.4, the landfills were estimated to contained 58 to 132 kg dry onion material per m^3 of landfill.

Typically, onions contain about 2.8 percent nitrogen on a dry weight basis (Warncke, 1991). Applying this value to the previous result gives a total elemental nitrogen loading of 1.7 to 3.7 kg/m^3. If this nitrogen was distributed homogeneously throughout the approximately 2 m of water that remained in the landfills over the period of study, the landfill water would contain between 10 and 45 mg/l elemental nitrogen. However, this approximation does not consider any nitrogen sinks, such as the microbial population, which may reduce the amount of nitrogen availalable for nitrification, nor does it consider the time required for nitrification.

A more realistic conception of nitrogen loading is one in which a fraction of the total available nitrogen is initially in the nitrate form. Richardson (1907) and Jackson *et al.* (1965) reported that less than 0.01 percent of the total nitrogen in onions is originally in the NO_3-N form. Based on this value, the landfill water would initially contain less than 0.5 mg/l NO_3-N. This initial NO_3-N loading would be compounded as the onions degraded if mineralization and nitrification of organic nitrogen occurred in the landfills. Concentrations of NO_3-N in the landfill water would, in this case, be expected to increase. If nitrification were inhibited in the landfills, then the concentration of NO_3-N would be expected to remain low. A combination of these two scenarios could result in a situation in which nitrate concentrations increase for a period of time and then, if nitrification ceased and microbial metabolism of nitrate continued, NO_3-N concentrations would subsequently decrease.

4.3 Fate of Nitrogen

Analysis of leachate below L1 showed high concentrations of NH_4-N and TKN, but very little NO_3-N. Nitrate-N concentrations within the landfill did not exceed 0.8 mg/l and NO_3-N concentrations below the landfill did not exceed 0.4 mg/l below L1 (Table 4). While NO_3-N levels remained low, NH_4-N concentrations below L1 increased to relatively high

Table 4: Nitrate-N Concentrations Beneath L1*

date		1/5/92	2/8	2/22	3/10	4/5	5/3	6/1	7/15	9/21	2/12/93	5/24	10/8	3/18/94
0.5 m														
A	1	---	---	---	---	---	---	---	---	---	---	1.0	---	---
	2	---	---	---	---	---	---	---	---	---	---	---	---	---
	3	---	---	---	---	---	---	---	---	---	---	---	---	
B	1	---	---	---	---	---	---	---	---					
	2	---	---	---	---	---	---	---	---		---			
	3	---	---	---	---	---	---	0.4	---	---	---			
C	1	0.2	0.1	---	---	---	---	0.2	---	---	---	---		---
	2	0.2	0.2	---	---	---	---	0.4	---	---	0.2	0.1	---	---
	3	0.3	0.3	---	---	---	---		---	---	0.1	---		---
1.0 m														
A	1	---	---	---	---	---	---	---	---	---		---		---
	2	---	---	---	---	---	---	---	---	---		0.1		
	3	---	---	---	---	---	---	---	---	---	---	0.7	---	---
B	1	---		---	---				---	---				
	2	---	---	---	---	---	---	0.1	---	---	---	---	---	---
	3	---	---	---	---	---	---	0.3	---	---	---	---	---	---
C	1	0.1	---	---	---	---		0.4	---	---	---	---	---	
	2	0.1	0.2	---	---	---	---	0.4	---	---	---	---		---
	3	0.2	0.3	---	---	0.2	---	---	---	---	0.2	---	---	---
2.0 m														
A	1	---	---	---	---	---	---	---	---	---	---	---	---	
	2	---	---	---	---	---	---	---	---	---	---	---	---	---
	3	---	---	---	---	---	---	---	---	---	---			
B	1	---	---	---	---		---	0.1	---	---	---	---	---	---
	2	---	---	---	---	---	---	0.2	---	---	---	---	---	---
	3	---	---	---	---	---	---	0.3	---	---	---	---	---	---
C	1	---	---	---	---	---	---	0.4	---	---	---	---	---	---
	2	0.2	0.2	---	---	---	---	0.4	---	---	---	---	---	---
	3	0.2	0.3	---	---	0.2	---	0.4	---	---	0.2	0.1	---	---

*Measurements below detection limit are denoted ---, detection limit is 0.1 mg/l. No entry indicates that soil-water sample was not obtained.

values. Ammonium-N levels as high as 390 mg/l and TKN levels as high as 340 mg/l were observed at 0.5 m beneath L1 (Figures 19 and 20). Ammonium-N concentrations as high as 12 mg/l and TKN concentrations as high as 150 mg/l were measured at 1 m (Figures 21 and 22). Ammonium-N concentrations did not exceed 1.8 mg/l and TKN concentrations did not exceed 38 mg/l at 2.0 m (Figures 23 and 24).

Trends in NH_4-N and TKN concentrations in and beneath L2 were very similar to those observed in and beneath L1, with concentrations increasing over time and decreasing with depth (Figure 25, Appendix B). Unlike L1, however, analyses of leachate below L2 showed that a pulse of NO_3-N passed through the system while the landfill was being filled (Figure 26). The timing of the pulse corresponded approximately to the initial wetting observed in the matric suction and tracer data, and may have been missed in L1 because of the two month lag time between filling of the landfill and the beginning of soil-water sampling. Four samples associated with the pulse had concentrations greater than 20 mg/l. Three of these samples were taken on November 23, 1992, over a month before the landfill was filled. The maximum NO_3-N concentration measured beneath L2 was 41 mg/l (Appendix B). This sample, obtained on January 7, 1993, was taken approximately one and a half months after the commencement of cull onion disposal (November, 1992) and shortly before the landfill was covered. A number of other samples, taken on both of these sampling dates, contained NO_3-N levels between 10 and 20 mg/l. These relatively high NO_3-N samples were collected from various depths and were probably associated with the initial flush of leachate. The nitrate in this leachate flush probably resulted from washing of onion waste by rainfall before the landfill was capped and from nitrification associated with early aerobic decomposition of the cull onions. No samples after January, 1993 contained greater than 5 mg/l NO_3-N. After February 13, 1993, NO3-N values decreased to less than 2 mg/l. Background NO_3-N levels obtained from the soil samples collected during instrumentation did not exceed 2.5 ug/g. Background NH_4-N levels did not

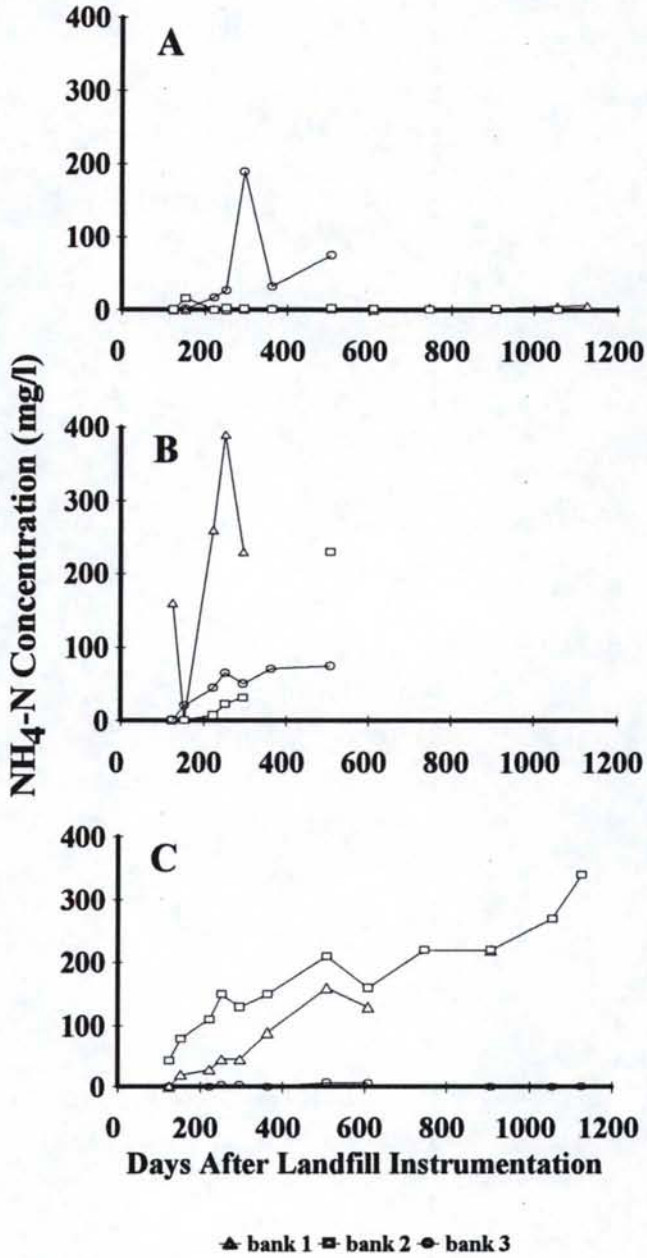

Figure 19. Ammonium concentrations at 0.5 m beneath sections A, B and C of L1 from January, 1992 through October, 1993. Disconnected data indicate when a sample was not obtained.

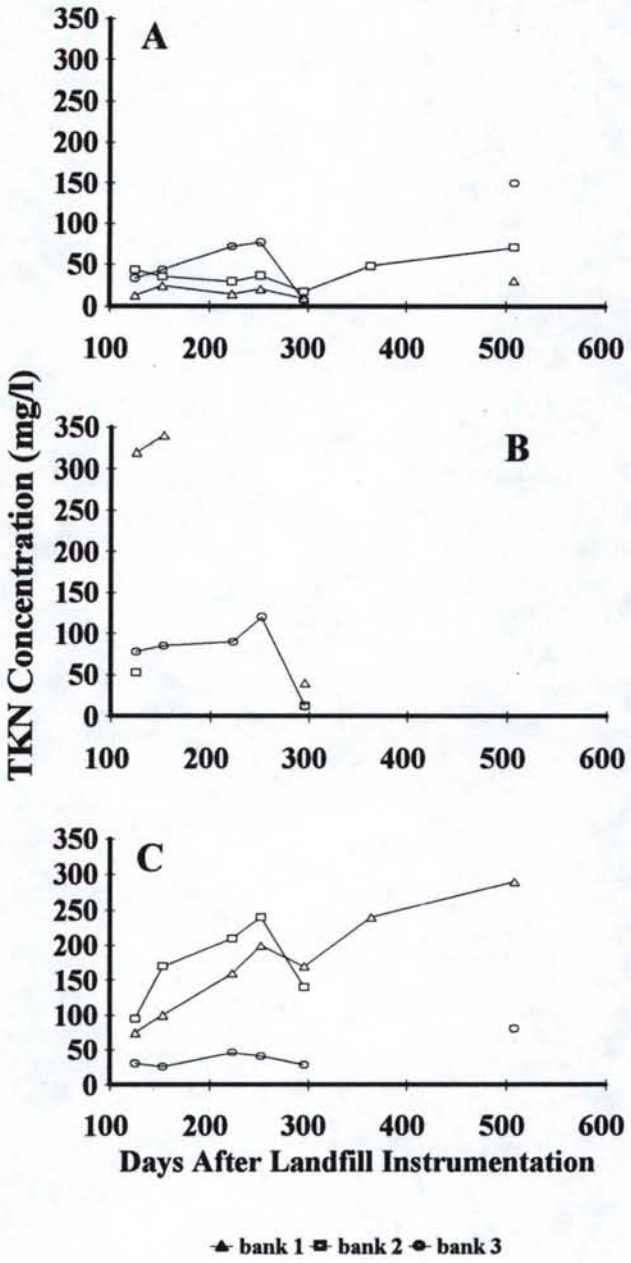

Figure 20. Total Kjeldahl concentrations at 0.5 m beneath sections A, B and C of L1 from January, 1992 through February, 1993. Disconnected data indicate when a sample was not obtained.

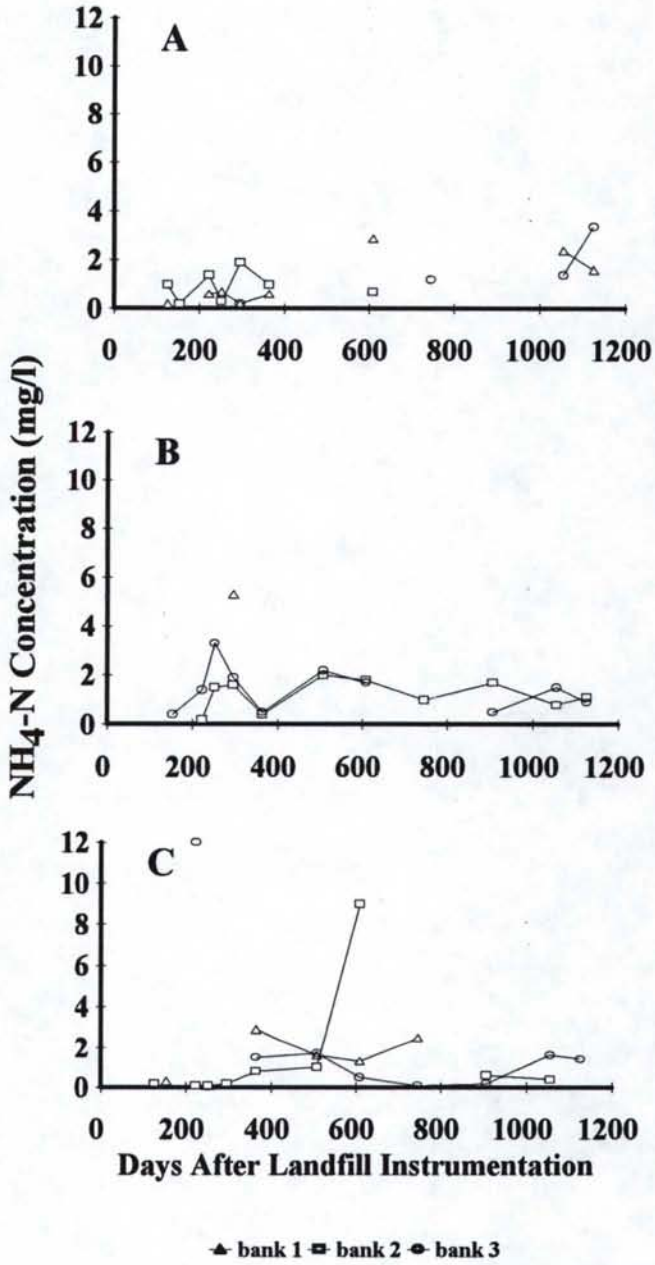

Figure 21. Ammonium concentrations at 1 m beneath sections A, B and C of L1 from January, 1992 through October, 1993. Disconnected data indicate when a sample was not obtained.

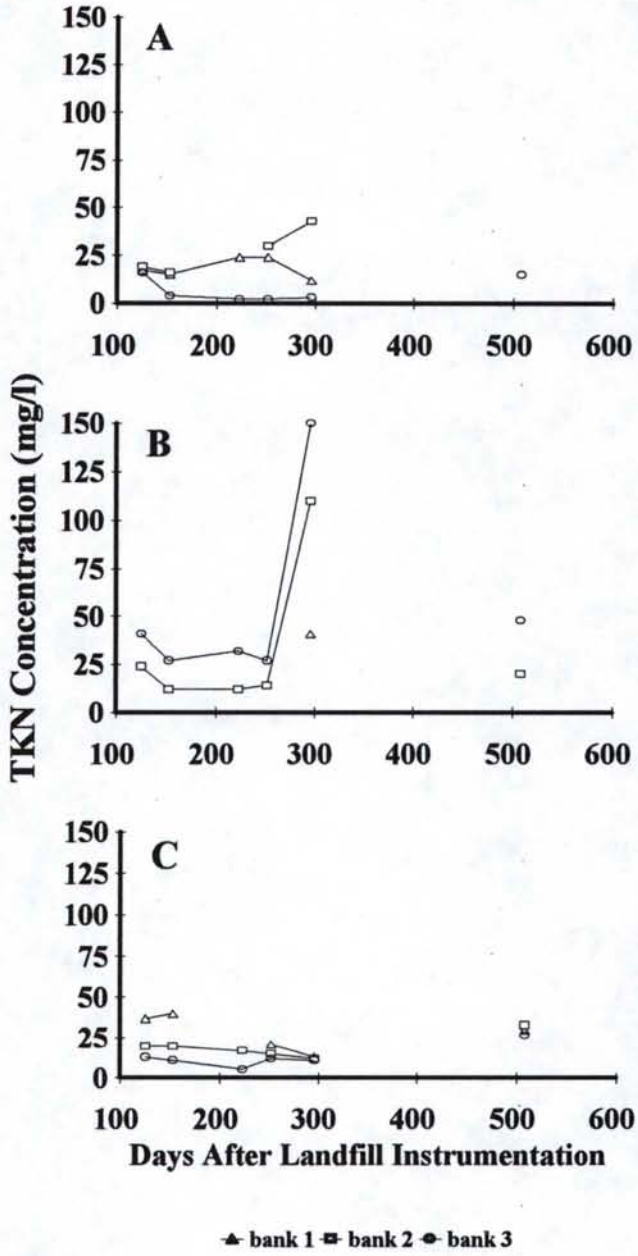

Figure 22. Total Kjeldahl concentrations at 1 m beneath sections A, B and C of L1 from January, 1992 through February, 1993. Disconnected data indicate when a sample was not obtained.

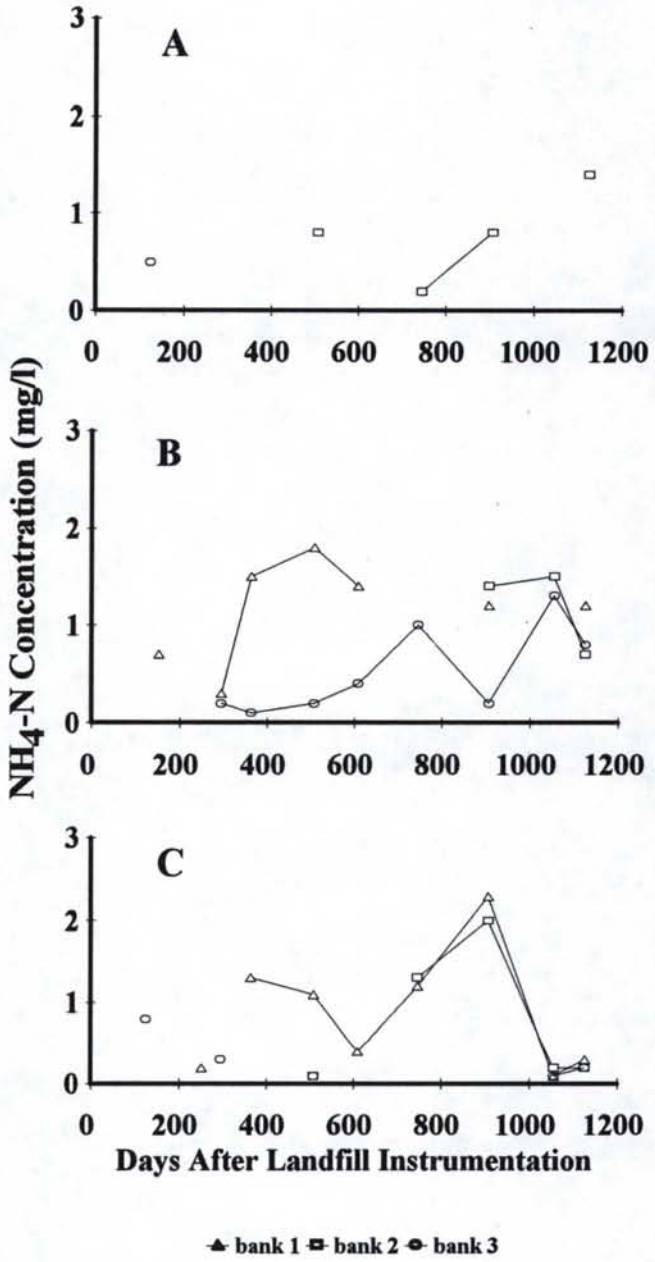

Figure 23. Ammonium concentrations at 2 m beneath sections A, B and C of L1 from January, 1992 through October, 1993. Disconnected data indicate when a sample was not obtained.

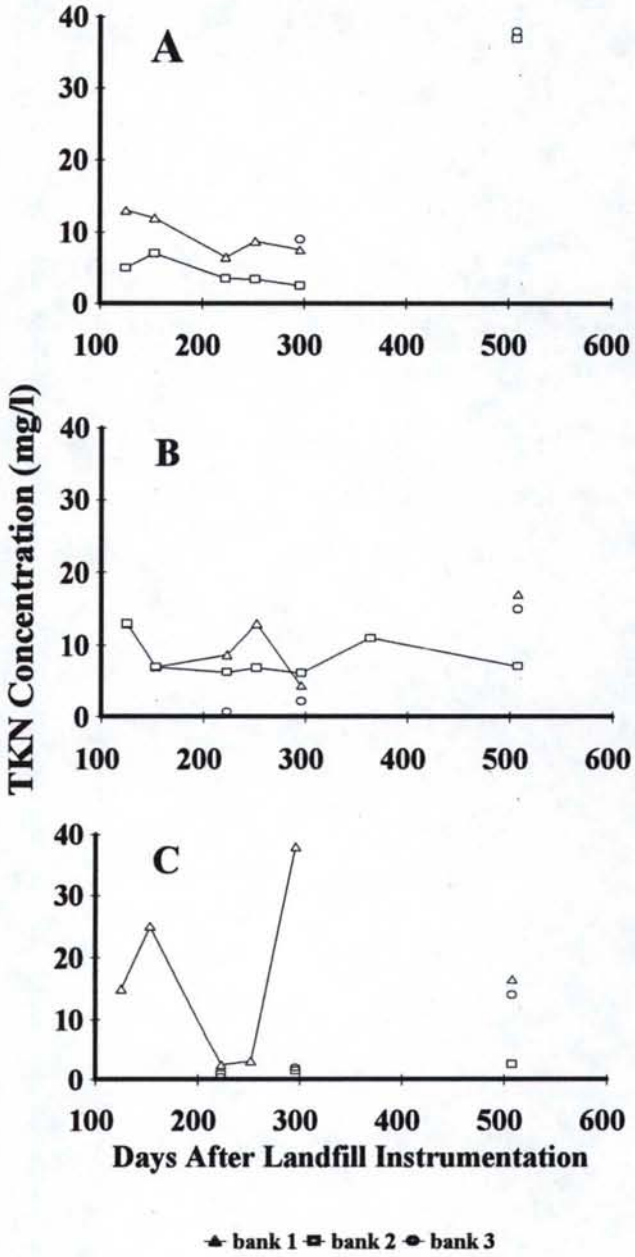

Figure 24. Total Kjeldahl concentrations at 2 m beneath sections A, B and C of L1 from January, 1992 through February, 1993. Disconnected data indicate when a sample was not obtained.

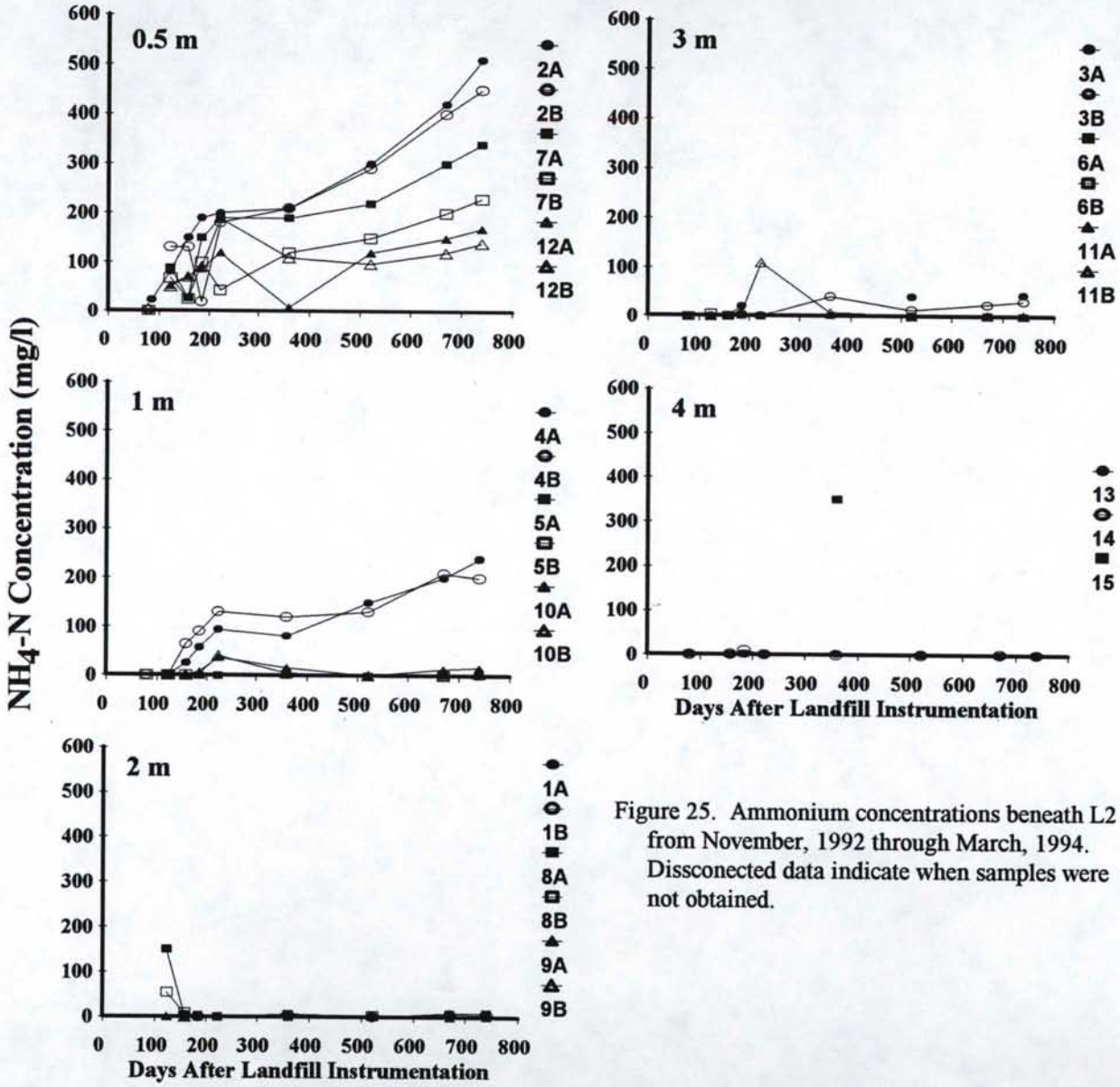

Figure 25. Ammonium concentrations beneath L2 from November, 1992 through March, 1994. Dissconected data indicate when samples were not obtained.

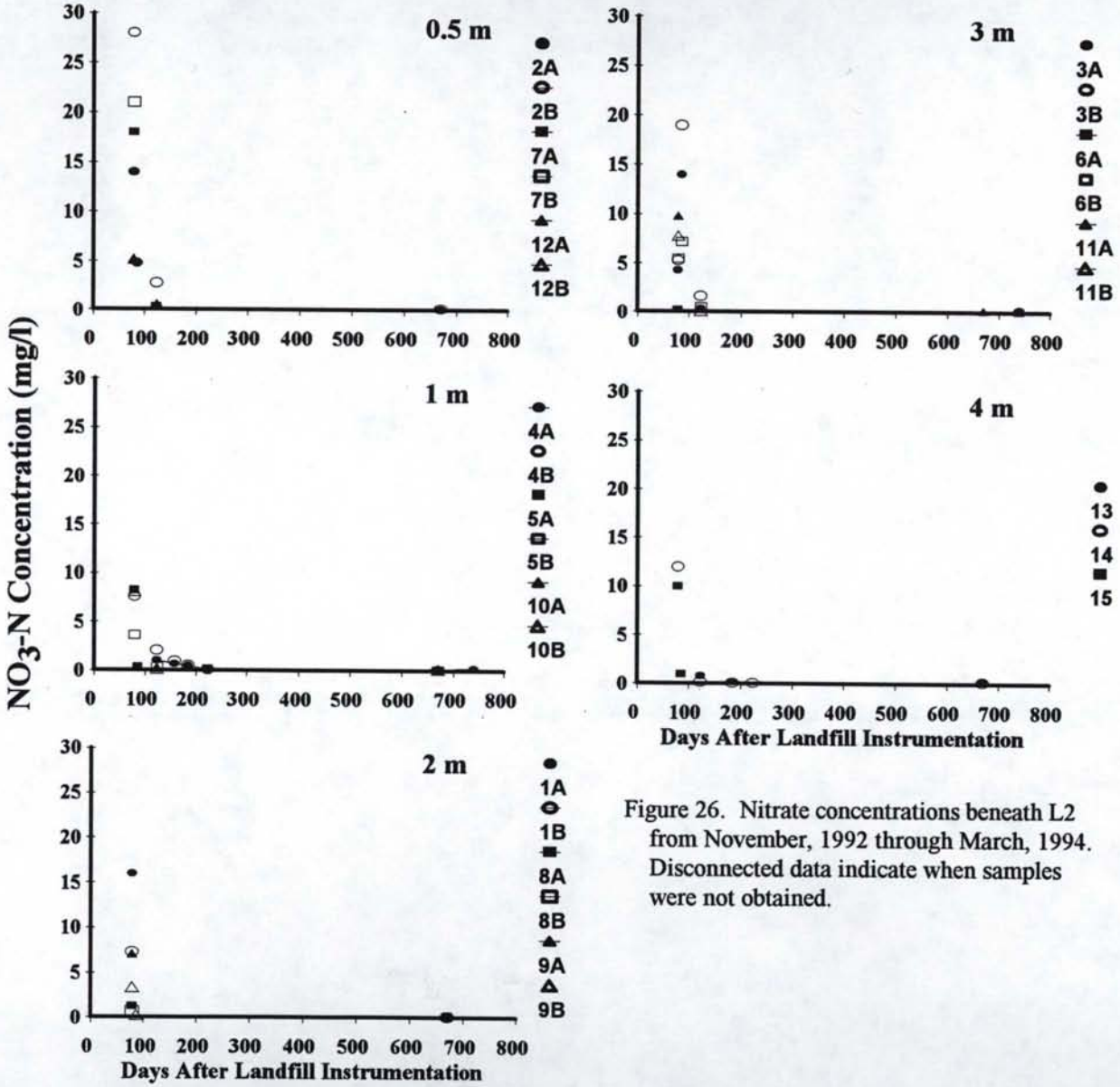

Figure 26. Nitrate concentrations beneath L2 from November, 1992 through March, 1994. Disconnected data indicate when samples were not obtained.

exceed 1.0 ug/g. In a saturated loamy sand soil, this corresponds to approximately 3.3 mg/l, assuming all mineral nitrogen was in the soil solution. This calculation gives a rough upper limit to the amount of background NO_3-N and NH_4-N in sediments beneath the landfill, and suggests that resident nitrogen did not have a significant effect on the measured leachate nitrogen concentrations.

Three important observations were apparent in the relative distributions of NO_3-N, NH_4-N, and TKN in and below the cull onion landfills: (1) NO_3-N concentrations decreased to background levels within the first two months of landfill operation and did not recover during the remainder of the study, (2) NH_4-N and TKN concentrations increased concurrently during the study, and (3) concentrations of NH_4-N and TKN decreased dramatically with depth beneath the landfill bases.

Low concentrations of NO_3-N observed in the cull onion landfill environment after two months of operation are explained by the anaerobic nature of the landfill contents. Oxygen diffusion is an important factor limiting nitrification in saturated soils and submerged environments (Alexander, 1977). In fact, nitrification is reported to be completely inhibited in submerged soils if no free oxygen exists (Patrick, 1982). Nitrification is inhibited in the landfills because oxygen diffusion is minimal in the range of temperatures and pressures found in the saturated landfill environment. Similar inhibition of nitrate production is reported in cattle feedlots and dairy lagoons in which organic nitrogen rich material was inhibited from forming nitrate (Mielke *et al.*, 1974; Schuman and McCalla, 1975; SCS, 1979; SCS, 1990).

The increases of NH_4-N in the landfill solution and in the leachate suggest that a significant pool of mineralizable organic nitrogen was distributed throughout the representative volume. Keeney (1982) reported that mineralization of organic nitrogen is not impeded by anaerobic conditions and that ammonium typically exists where the organic

precursor is found. The general decrease in NH_4-N and TKN concentrations with depth can be explained by the same physical, biological and chemical phenomena used to explain the formation of flow boundaries in other organic waste systems (Schuman and McCalla, 1975, SCS, 1990). First, larger molecular weight organics were believed to be screened out of the leachate by the sedimentary matrix beneath the landfills. Second, smaller molecules were believed to be screened out and an organic layer was formed. In addition, biochemical activity may have fortified the organic layer by binding the organic and sedimentary matrix into a matlike structure (Schuman and McCalla, 1975, SCS, 1990). It is probable that flow velocities through the landfill bases were decreased by several orders of magnitude when the "self-lining" process was complete. Movement of ammonium was further retarded because it is a cation and was, therefore, partitioned into the organic and mineral fraction of the landfill sediments (Keeney, 1982).

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

Landfill disposal of cull onions was studied to determine whether cull onion landfills are a potential source of nitrate contamination of groundwater in the western Treasure Valley. The following conclusions are based on the findings of this study.

1) Well constructed and maintained cull onion landfills are not major source of nitrate contamination to regional groundwater resources while the cull onion material is saturated and anaerobic.

2) Large quantities of ammonium stored in the landfills may be converted to nitrate after the landfills dry and the residual cull onion material becomes aerobic. If this conversion occurs, the landfills may become a source for nitrate contamination of the underlying aquifer.

3) A low conductivity organic lining forms at the base of cull onion landfills which are constructed in environments similar to those of the experimental landfills. Matric suction data and hydraulic gradient analysis suggest that it takes two to three months for an organic lining to become sufficiently developed to restrict the downward movement of water and retain ponded water in cull onion landfills.

4) Nitrate concentrations below cull onion landfills increase during the initial two to three months of landfill operation (a maximum concentration of 41 mg NO_3-N/l was measured), but decrease rapidly to below the detection limit after the formation of the low

conductivity layer. Nitrate concentrations remain below detection limit for at least four years because nitrification is inhibited in saturated (anaerobic) cull onion waste.

5) Ammonium concentrations below cull onion landfills increase from zero to as much as 400 mg NH_4-N/l after the formation of the low conductivity layer and subsequent inhibition of nitrification, because mineralization is not inhibited in the saturated cull onion waste.

6) Analyses of hydraulic heads and tracer breakthrough curves indicate that leachate movement in the sediments below cull onion landfills is restricted to unsaturated hydraulic conductivities. An initial leachate flux containing up to 41 mg NO_3-N/l is hypothesized to move into the profile at approximately 10^{-5} to10^{-7} cm/s before the low conductivity layer is formed. Leachate movement is reduced to approximately 10^{-9} to10^{-7} cm/s after the formation of the low conductivity layer.

5.2 Recommendations

The large pool of organic nitrogen and ammonium that is retained in the saturated cull onion landfills may become a source of nitrate when the residual cull onion material dries. As a result, three recommendations are presented which are aimed at mitigating potential long-term occurrences of nitrate leaching.

1) Continue to monitor NO_3-N and NH_4-N concentrations and matric suctions below the cull onion landfills. In addition, monitor the oxygen condition in the landfills, during and after desaturation to evaluate the rate and magnitude of long-term nitrate leaching.

2) Excavate residual cull onion material in all cull onion landfills after a sufficient degradation time to remove the source of potential nitrate contamination, and land apply the

residual material at acceptable nitrogen loading rates. An additional benefit of this proposed management practice is that it minimizes the land area required for future landfills.

3) Locate future cull onion landfills in hydrogeologic settings similar to the experimental landfills. Water tables should be at least 25 m below the base of landfills in sediments which are composed of fine-grained lacustrine deposits. Shallow water tables and coarse grained sediments should be avoided.

REFERENCES

Alexander, M., 1977, "Introduction to Soil Microbiology," Wiley and Sons, New York, 467 p.

Barrington, S., and P.J. Jutras, 1983, Soil sealing by manure in various soil types, ASAE Pap. 83-4571, ASAE, St. Joseph, MI.

Barrington, S., and R. Broughton, 1990, Literature review of sealing mechanisms, in Design and Construction Guidelines for Considering Seepage from Agricultural Waste Storage Ponds and Treatment Lagoons, Technical Note 716, South National Technical Center, Soil Conservation Service, Fort Worth, TX.

Bowman, R.S., 1984, Evaluation of some new tracers for water studies, Soil Sci. Soc. Am. J., v. 48, p. 987-993.

Brune, D., (1990), Literature review of sealing mechanisms, in Design and Construction Guidelines for Considering Seepage from Agricultural Waste Storage Ponds and Treatment Lagoons, Technical Note 716, South National Technical Center, Soil Conservation Service, Fort Worth, TX.

Carsel, R.F., and R.S. Parrish, 1988, Developing joint probablility distributions of soil water retention characteristics, Water Resour. Res., v. 24, p. 755-769.

Davis, S.N., G.M. Thompson, H.W. Bentley, and G. Stiles, 1980, Groundwater tracers-a short review, Groundwater, v. 18, p. 14-23.

Domenico, P.A., and F.W. Schwartz, 1990, "Physical and Chemical Hydrogeology", John Wiley and Sons, Inc., New York, 824 p.

Elrick, D.E. and W.D. Reynolds, 1992, Infiltration from constant-head well permeameters and infiltrometers, In G.C. Topp, W.D. Reynolds, and R.E. Green (eds.), Advances in measurement of soil physical properties: bringing theory into practice, SSSA Spec. Publ. No. 30, p. 1-24, SSSA, Madison, WI.

Fetter, C.W., 1988, "Applied Hydrology," Merrill Publishing, Colombus, Ohio, 592 p.

Figueroa, E.E., 1989, The competitiveness of New York state onions during the 1987-88 marketing year, AE-Ext. 89-1, Cornell University, Ithaca, New York.

Gee, G.W. and J.W. Bauder, 1986, Partical-size analysis, p. 383-409, In A. Klute (ed.), Methods of soil analysis, Part 1, 2nd ed. Agron. Monogr. 9, ASA, Madison, WI.

Gish, T.J., W. Zhuang, C.S. Helling, and P.C. Kearney, 1986, Chemical transport under no-till field conditions, Geoderma, v. 38, p. 251-259.

Hazen, A, 1911, Discussion: dams on sand foundations, Trans. Am. Soc. Civil Engineers. v. 73(199), p. 199-203.

Hillel, D., 1982, "Introduction to Soil Physics," Academic Press, San Diego, California, 364 p.

Jackson, W.A., J.S. Steel, and V.R. Boswell, 1965, Nitrates in edible vegetables and vegetable products, J. Amer. Soc. Hort. Sci., v. 90, p. 349-352.

Jansson, S.L. and J. Persson, 1982, Mineralization and immobilization of soil nitrogen, In F.J. Stevenson (ed.), Nitrogen in agricultural soils, Agronomy, v. 22, p. 229-248, ASA, CSSA, and SSSA, Madison, WI.

Jensen, L., 1992, Malheur County Extension Agent, Oregon State University, Personal Comm.

Jury, W.A., H. Elabd, and M. Resketo, 1986, Field study of naproamide movement through unsaturated soil, Water Resour. Res., vol. 22, p. 749-755.

Keeney, D.R., 1982, Nitrogen management for maximum efficiency and minimum pollution, In F.J. Stevenson (ed.), Nitrogen in agricultural soils, Agronomy, v. 22, p. 605-649, ASA, CSSA, and SSSA, Madison, WI.

Klute, A., and C. Dirksen, 1986, Hydraulic conductivity and diffusivity: laboratory methods, p. 687-734, In A. Klute (ed.), Methods of soil analysis, Part 1, 2nd ed. Agron. Monogr. 9, ASA, Madison, WI.

Korom, S.F., and R.W. Jeppson, 1994, Nitrate contamination from dairy lagoons constructed in coarse alluvial deposits, J. Environ. Qual., vol. 23(5), p. 973-976.

Levi, A.E., J.K. Fellman, J.F. Guenthner, L.D. Makus, and M.K. Thornton, 1990, University of Idaho, Agricultural Experiment Station Bulletin No. 730, 14 p.

Luckner, L., M.Th. van Genuchten, and D.R. Nielsen, 1989, A consistent set of parametric models for the two phase flow of immicible fluids in the subsurface, Water Resour. Res., vol. 25, p. 2187-2193.

Malde, H.E., 1991, Quaternary geology and structural history of the Snake River Plain, Idaho and Oregon, In Morrison, R.B. (ed.), Quaternary Nonglacial Geology: Conterminous U.S., Boulder, Colorado, Geologic Society of America, The Geology of North America, v. K-2, 671 p.

Mahler, R.L. and M. Gardner, 1991, Idaho Wellhead Sampling Program: Nitrate Sampling, Gem and Payette Counties, March, 1991, University of Idaho Cooperative Extension System Publication WQ-7.

Mielke, L.N., N.P. Swanson, and T.M. McCalla, 1974, Soil profile conditions of cattle feedlots, J. Environ. Qual., v. 3, p. 14-17.

Mielke, L.N., and J.R. Ellis, 1976, Nitrogen in soil cores and groundwater under abandoned cattle feedlots, J. Environ. Qual., v. 5(1), p. 71-75.

Miller, S., 1992, Associate Professor, Geology and Geological Engineering, University of Idaho, Personal Comm.

Mualem, Y., 1976, A new model for predicting the hydraulic conductivity of unsaturated porous media, Water Resour. Res., v. 12, p. 513-522.

Norton, D., and R. Knapp, 1977, Transport phenomenon in hydrothermal systems: nature of porosity, Amer. J. Sci., v. 27, p. 913-936.

Othberg, K.L., 1994, Geology and geomorphology of the Boise Valley and adjoining areas, western Snake River Plain, Idaho, Idaho Geological Survey Bulletin 29, 53 p.

Patrick, W.H., 1982, Nitrogen transformations in submerged soils, In F.J. Stevenson (ed.), Nitrogen in agricultural soils, Agronomy, v. 22, p. 449-462, ASA, CSSA, and SSSA, Madison, WI.

Patrick, R., E. Ford and J. Quarles, 1987, "Groundwater Contamination in the United States", University of Pennsylvania Press, Philadelphia, 513 p.

Paul, E. A. and F. E. Clark., 1989, "Soil Microbiology and Biochemistry", Academic Press, Inc., New York, 273 p.

Rawls, W.J., D.L. Brakensiek, and K.E. Saxton, 1982, Estimating soil water properties, Trans. Am. Soc. Agric. Engineers, v. 25(5), p. 1316-1320 and 1328.

Reigert, L.E., 1992, Anaerobic digestion and mass transfer in the unsaturated landfill environment: University of Idaho M.S. thesis, 154 p.

Richardson, W.W., 1907, The occurrence of nitrates in vegetable foods, in cured meats and elsewhere, J. Amer. Chem. Soc., v. 29, p. 1757-1767.

Schmidt, E.L., 1982, Nitrification in soil, In F.J. Stevenson (ed.), Nitrogen in agricultural soils, Agronomy, v. 22, p. 253-283, ASA, CSSA, and SSSA, Madison, WI.

Schuman, G.E., and T.M. McCalla, 1975, Chemical characteristics of a feedlot soil profile, Soil Science, v. 119(2), p. 113-118.

Sewell, J.I., 1978, Dairy lagoon effects on groundwater quality, Trans. Am. Soc. Agric. Engineers, St Joseph, MI., p. 948-952.

Soil Conservation Service, 1976, Soil Survey of Payette County, Idaho, United States Department of Agriculture, Washington D.C., 97 p.

Soil Conservation Service, 1979, Pond Sealing or Lining, Artificial Gleization: Interim Report 521-F-1, United States Department of Agriculture, Washington D.C., 3 p.

Soil Conservation Service, 1990, Design and Construction Guidelines for Considering Seepage from Agricultural Waste Storage Ponds and Treatment Lagoons, Technical Note 716, South National Technical Center, Soil Conservation Service, Fort Worth, TX.

Stephens, D.B., and K.R. Rehfeldt, 1985, Evaluation of closed-form analytical models to calculate conductivity in a fine sand, Soil Sci. Soc. Am. J., vol. 49, p. 12-19.

Stewart, B.A., F.G. Viets, Jr., G.L. Hutchinson, W.D. Kemper, F. E. Clark, M.L. Fairbourn and F. Strauch, 1967, Distribution of nitrates and other pollutants under fields and corrals in the middle South Platte valley of Colorado, USDA-ARS 41-134, 206 p.

Sweeten, J., 1990, Literature review of sealing mechanisms, in Design and Construction Guidelines for Considering Seepage from Agricultural Waste Storage Ponds and Treatment Lagoons, Technical Note 716, South National Technical Center, Soil Conservation Service, Fort Worth, TX.

U.S. Environmental Protection Agency,1983, Methods for chemical analysis of water and waste water, USEPA Rep. 600/4-79-020, USEPA, Washington D.C.

U.S. Environmental Protection Agency, 1991, GEO-EAS 1.2.1: Geostatistical Environmental Assessment Software, EPA 600/8-91/008, Environmental Monitoring Systems Laboratory, Office of Research and Development, USEPA, Las Vegas, NV 89119.

van Genuchten, M.Th., 1980, A closed-form equation for predicting the hydraulic conductivity of unsaturated soils., Soil Sci. Soc. Am. J., vol. 44, p. 892-898.

van Genuchten, M.Th., and D.R. Nielsen, 1985, On describing and predicting the hydraulic conductivity of unsaturated soils, Ann. Geophys., vol. 3, p. 615-628.

van Genuchten, M.Th., F.J. Leij, and S.R. Yates, 1991, The RETC Code for Quantifying the Hydraulic Functions of Unsaturated Soils, EPA/600/2-91/065, U.S. Environmental Protection Agency, 85 p.

Warncke, D.D., 1991, Department of Crop and Soil Science, Michigan State University, personal comm.

Wild, A., and I.A. Babiker, 1976, The asymmetric leaching pattern of nitrate and chloride in a loamy sand under field conditions, Soil Sci, vol. 27, p. 460-466.

APPENDIX A

Geostatistical Analysis

```
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|                                                                              |
|                   A program for computing variograms                         |
|  File Prefix:d:\culonion\matrix\kanal\geostat\                               |
|------------------------------------------------------------------------------|
|  File                                  |  Vars :          # Kept    :       |
|                                        |  Data :          # Obs<=0  :       |
|  Pair Comparison File: ksat.pcf        |  Pairs:          # Missing :       |
|----------------------------------------+-------------------------------------|
|  Variable                              |                                     |
|          Variable: "Ks(e3)             |        Variogram Options            |
|        Log Option:  On                 |                                     |
|                                        |  Use this option to display the     |
|----------------------------------------|  Variogram Options Screen and menu. |
|  Limits                                |  Variogram options may be selected  |
|        Minimum:          -2.526        |  and the variogram may be computed. |
|        Maximum:            .788        |                                     |
|------------------------------------------------------------------------------|
|                                                                              |
-------------------------------------------------------------------------------
  Prefix  Data  Variable  Limits  Options/Execute  Quit
  Specify variogram options, compute
```

```
------------------------------- O P T I O N S -------------------------------
| Variable LN ("Ks(e3))                          Pair File      : ksat.pcf     |
| Minimum  :    -2.53                            Min. Distance  :      .300    |
| Maximum  :     .788                            Max. Distance  :      14.4    |
|-----------------------------------------------------------------------------|
|                                 | Lag        Distance        Lag        Distance |
| Direction                       |-----------------------------------------------|
|                                 |  1          4.000          13                |
| Direction     :          .000   |  2          6.000          14                |
| Tolerance     :        90.000   |  3          8.000          15                |
| Max Bandwidth:            MAX   |  4          9.000          16                |
|                                 |  5         12.000          17                |
|-------------------------------  |  6                         18                |
|                                 |  7                         19                |
| Lag Spacing                     |  8                         20                |
|                                 |  9                         21                |
| Minimum       :         2.000   | 10                         22                |
| Maximum       :        10.000   | 11                         23                |
| Increment     :         2.000   | 12                         24                |
|-----------------------------------------------------------------------------|
|                                                                             |
-------------------------------------------------------------------------------
  Direction  New Lags  Change Lags  Post Plot  Execute  Quit
  Specify pair orientation (selection) criteria
```

R E S U L T S

Variable: LN ("Ks(e3))		Direction : .000
Minimum : -2.526	Estimator : Variogram	Tolerance : 90.000
Maximum : .788	Total Pairs : 240	BandWidth : n/a

	Pairs	Avg Distance	Estimate		Pairs	Avg Distance	Estimate
1	77	3.092	.626	13			
2	68	4.983	.644	14			
3	46	6.915	.610	15			
4	20	8.474	.641	16			
5	29	10.100	.641	17			
6				18			
7				19			
8				20			
9				21			
10				22			
11				23			
12				24			

Type Plot Box Plot Lag Results Model Quit
Plot the selected estimator vs. average distance

M O D E L

	Pairs	Avg Distance	Value		Pairs	Avg Distance	Value
1	77	3.092	.626	18			
2	68	4.983	.644	19			
3	46	6.915	.610	20			
4	20	8.474	.641	21			
5	29	10.100	.641	22			
6				23			
7				24			
8							
9				Model			
10						Nugget :	.633
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12					Type	Sill	Range
13							
14					Linear	.633	10000.000
15							
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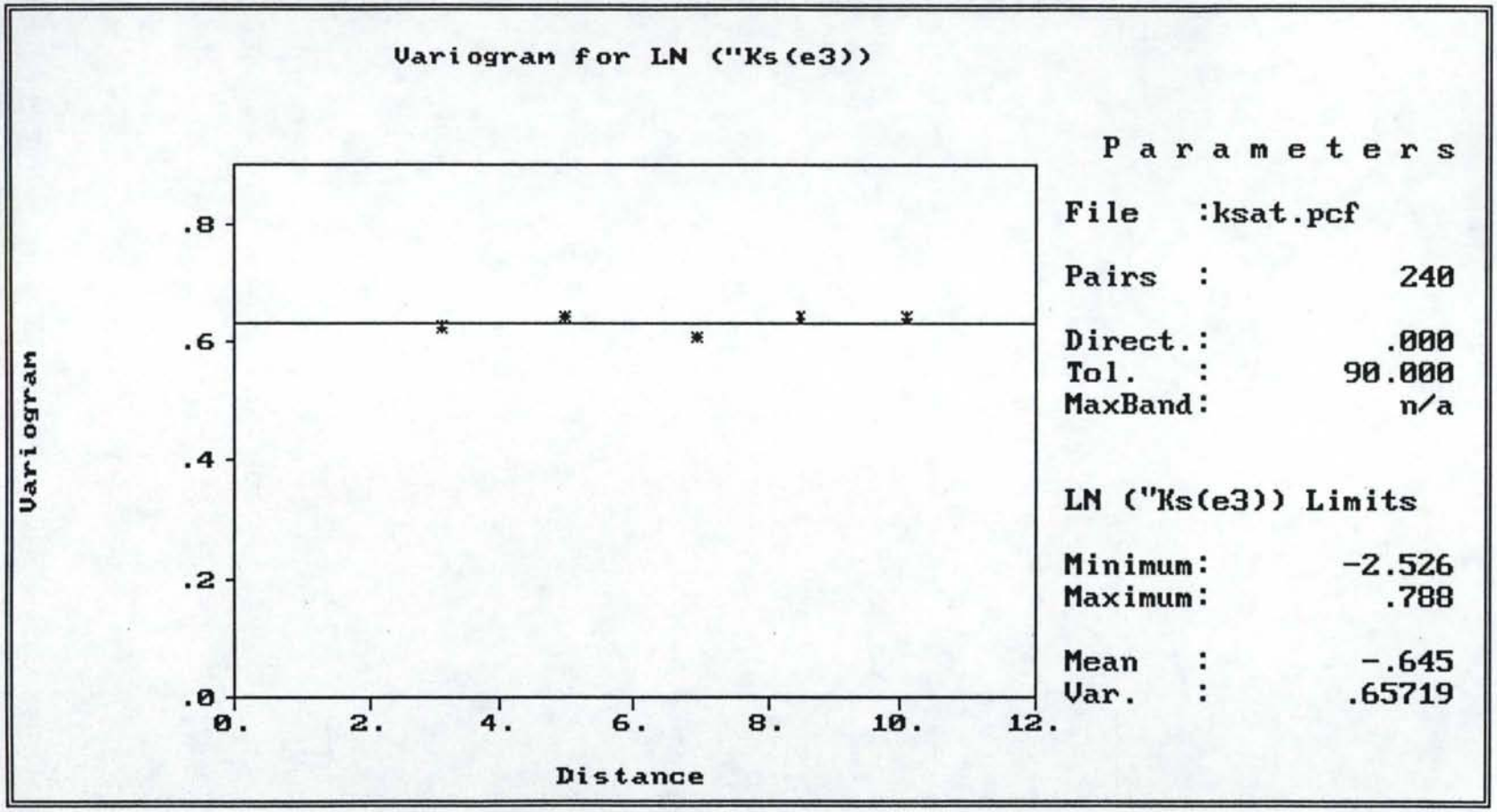
Model Plot Options Quit
Plot the variogram and Model


Variogram for LN ("Ks(e3))
Variogram
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Parameters
File :ksat.pcf
Pairs : 240
Direct.: .000
Tol. : 90.000
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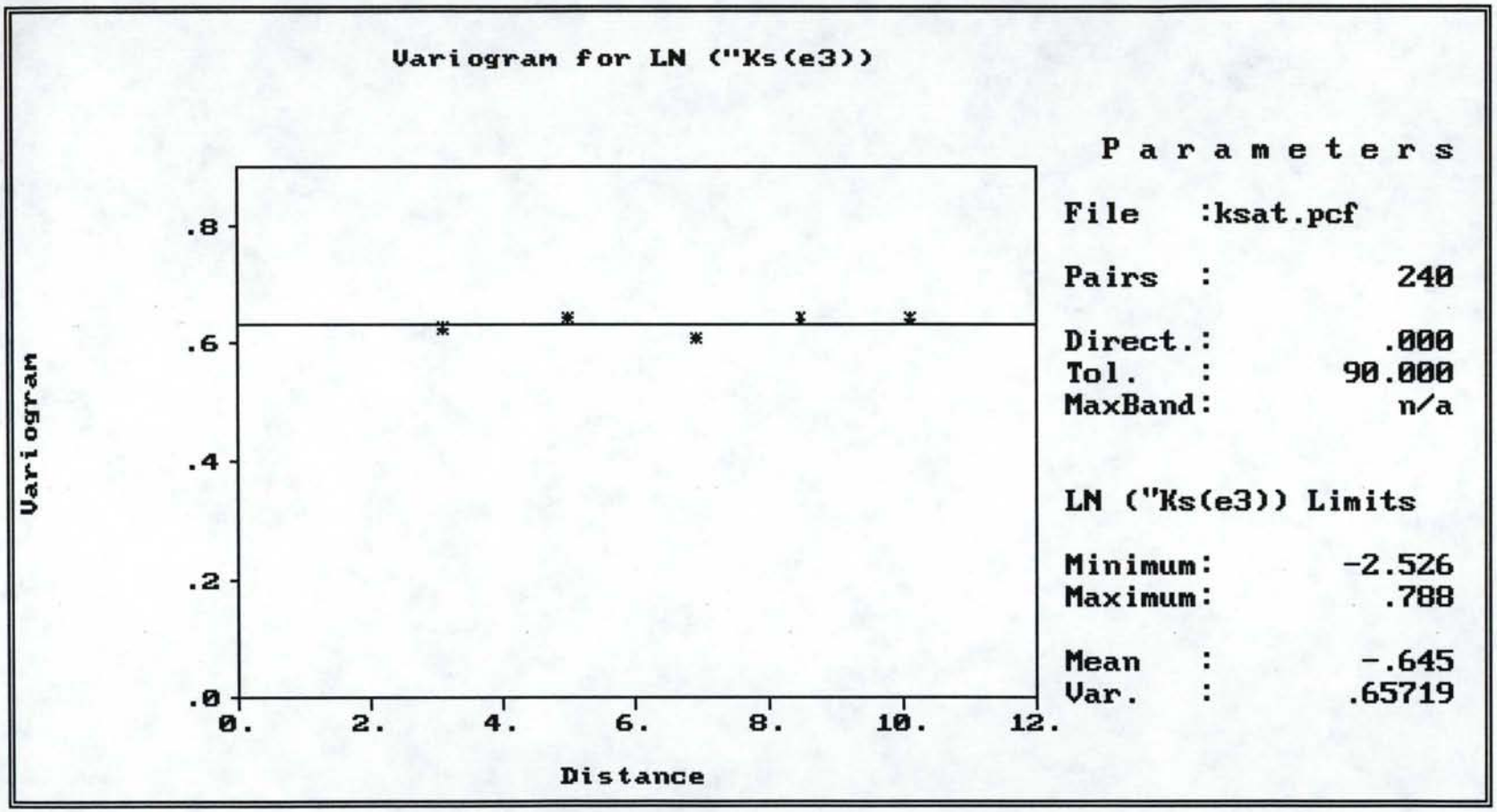
LN ("Ks(e3)) Limits
Minimum: -2.526
Maximum: .788
Mean : -.645
Var. : .65719


Variogram for LN ("Ks(e3))
Variogram
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.6
.4
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2.
4.
6.
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10.
12.
Distance
Parameters
File :ksat.pcf
Pairs : 240
Direct.: .000
Tol. : 90.000
MaxBand: n/a
LN ("Ks(e3)) Limits
Minimum: -2.526
Maximum: .788
Mean : -.645
Var. : .65719

Cull Onion Landfill 2 Hydraulic Conductivity Measurements

9								
Easting	m	G7.2						
Northing	m	G7.2						
Ks	cm/s	G7.2						
Ks (e-3)	cm/s	F7.2						
x1		G8.1						
x2		G8.1						
x3		G8.1						
x4		G8.1						
x5		G8.1						
8.0	4.0	.20E-04	.02	1.	1.	1.	1.	1.
4.9	.50	.50E-04	.05	1.	1.	1.	1.	1.
9.5	.50	.60E-04	.06	1.	1.	1.	1.	1.
4.2	3.5	.80E-04	.08	1.	1.	1.	1.	1.
1.3	5.6	.11E-03	.11	.0	1.	1.	1.	1.
7.5	3.0	.13E-03	.13	.0	1.	1.	1.	1.
11.	5.3	.22E-03	.22	.0	1.	1.	1.	1.
.00	5.7	.31E-03	.31	.0	.0	1.	1.	1.
9.7	.30	.34E-03	.34	.0	.0	1.	1.	1.
9.6	4.6	.38E-03	.38	.0	.0	1.	1.	1.
2.1	1.6	.40E-03	.40	.0	.0	1.	1.	1.
9.5	2.5	.47E-03	.47	.0	.0	1.	1.	1.
3.1	1.4	.52E-03	.52	.0	.0	.0	1.	1.
2.1	1.3	.52E-03	.52	.0	.0	.0	1.	1.
5.4	2.6	.54E-03	.54	.0	.0	.0	1.	1.
10.	3.5	.55E-03	.55	.0	.0	.0	1.	1.
.40	.30	.55E-03	.55	.0	.0	.0	1.	1.
.80	4.0	.66E-03	.66	.0	.0	.0	1.	1.
2.1	4.5	.79E-03	.79	.0	.0	.0	.0	1.
6.9	.60	.83E-03	.83	.0	.0	.0	.0	1.
4.2	2.0	.85E-03	.85	.0	.0	.0	.0	1.
4.6	4.9	.10E-02	1.00	.0	.0	.0	.0	1.
7.4	1.3	.11E-02	1.10	.0	.0	.0	.0	.0
6.0	4.0	.13E-02	1.30	.0	.0	.0	.0	.0
14.	4.5	.13E-02	1.30	.0	.0	.0	.0	.0
8.0	4.7	.15E-02	1.50	.0	.0	.0	.0	.0
12.	6.0	.22E-02	2.20	.0	.0	.0	.0	.0

B A T C H S T A T I S T I C S

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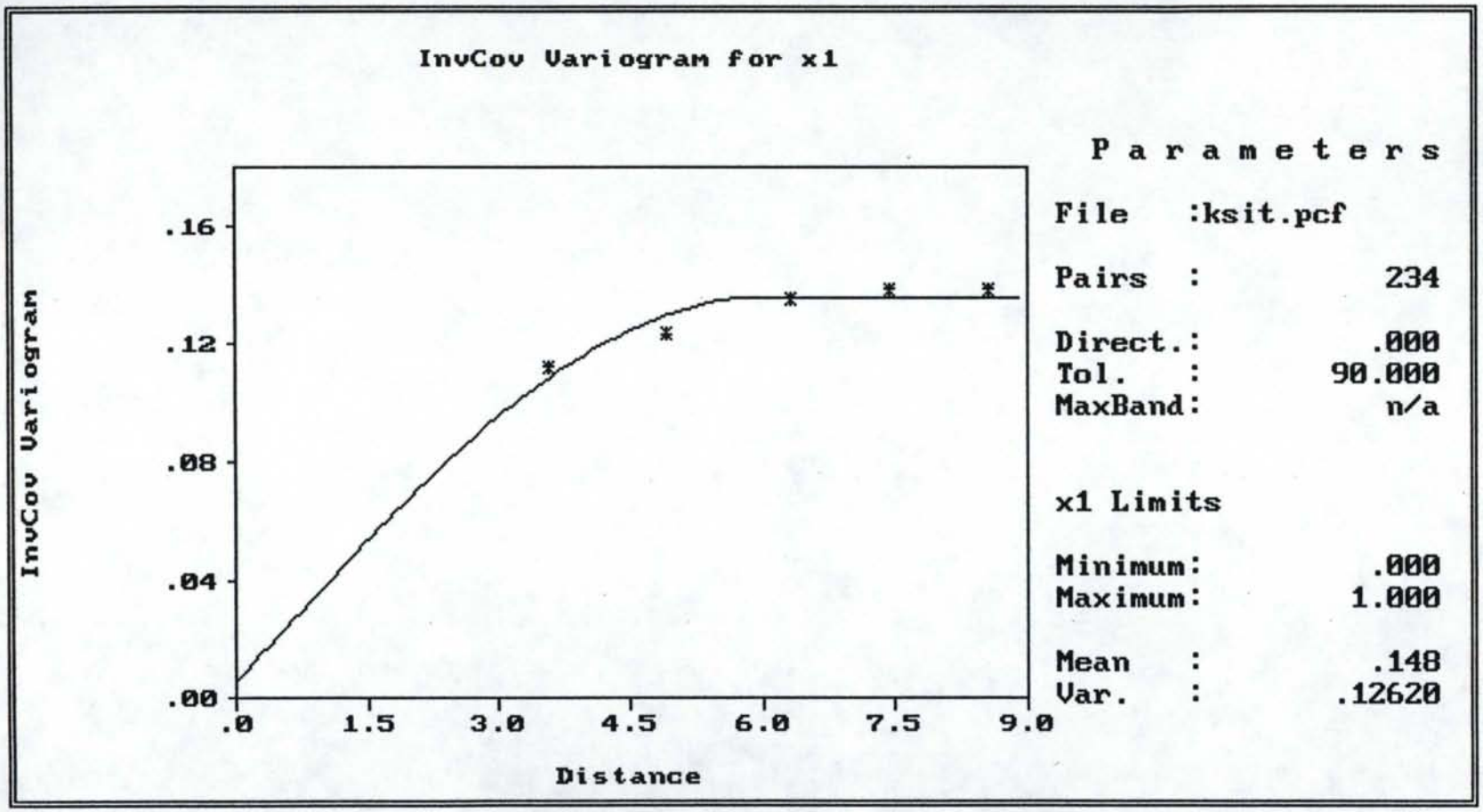
	Easting	Northing	Ks	Ks (e-3)	x1
N used :	27	27	27	27	27
N missing :	0	0	0	0	0
N .LE. 0 :	1	0	0	0	23
Mean :	6.085	3.063	.001	.621	.148
Variance :	14.827	3.455	.000	.269	.131
Std. Dev. :	3.851	1.859	.001	.518	.362
Coef. Var. :	63.279	60.690	83.414	83.414	244.359
Skewness :	.117	-.104	1.205	1.205	1.981
Kurtosis :	2.032	1.662	4.371	4.371	4.924
Minimum :	.000	.300	.000	.020	.000
25th %tile :	2.100	1.300	.000	.197	.000
Median :	6.000	3.500	.001	.520	.000
75th %tile :	9.500	4.525	.001	.835	.000
Maximum :	14.000	6.000	.002	2.200	1.000

	x2	x3	x4	x5
N used :	27	27	27	27
N missing :	0	0	0	0
N .LE. 0 :	20	15	9	5
Mean :	.259	.444	.667	.815
Variance :	.199	.256	.231	.157
Std. Dev. :	.447	.506	.480	.396
Coef. Var. :	172.251	113.933	72.058	48.581
Skewness :	1.099	.224	-.707	-1.621
Kurtosis :	2.207	1.050	1.500	3.627
Minimum :	.000	.000	.000	.000
25th %tile :	.000	.000	.000	1.000
Median :	.000	.000	1.000	1.000
75th %tile :	.250	1.000	1.000	1.000
Maximum :	1.000	1.000	1.000	1.000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.08
.04
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4.5
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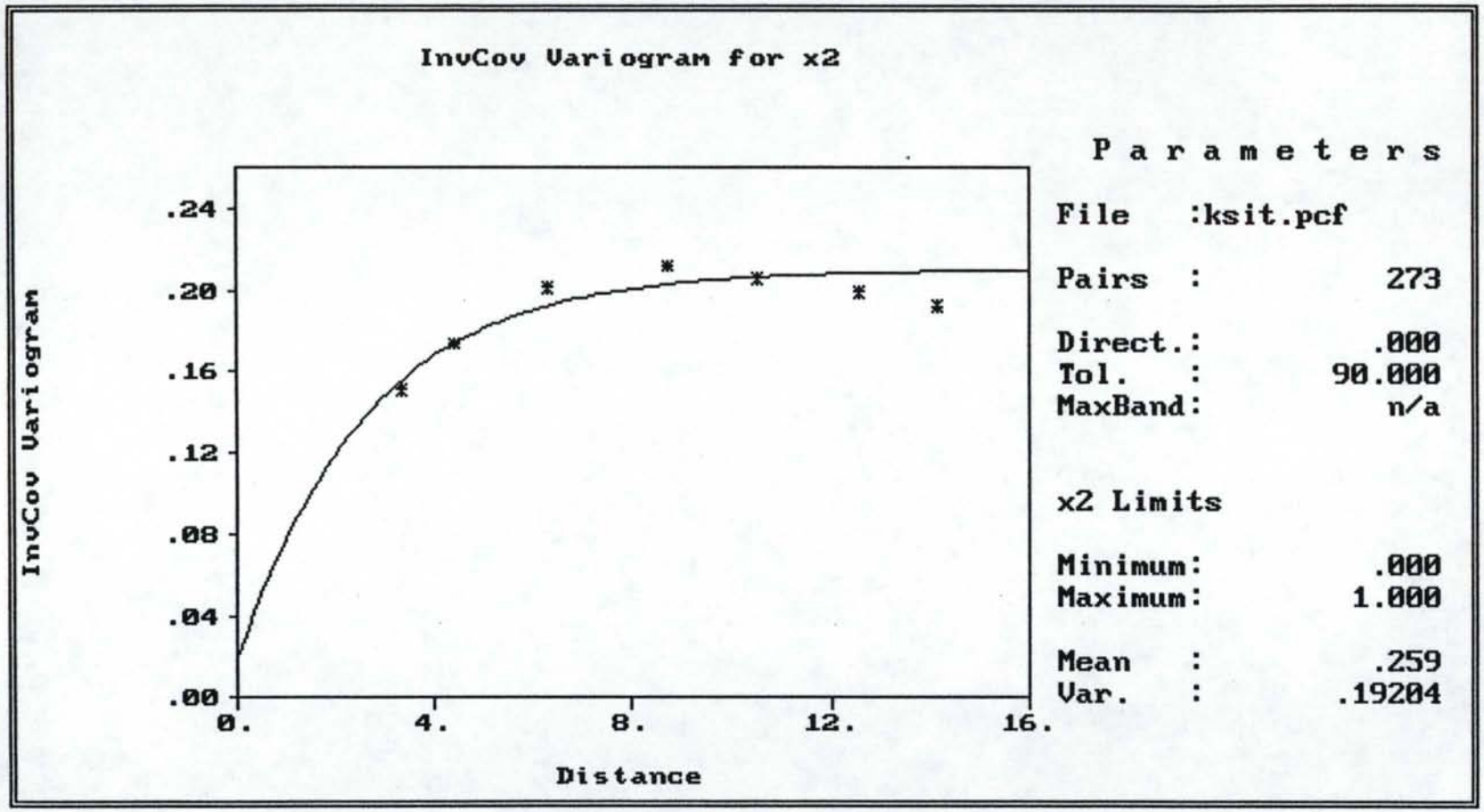
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8.
12.
16.
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x2 Limits
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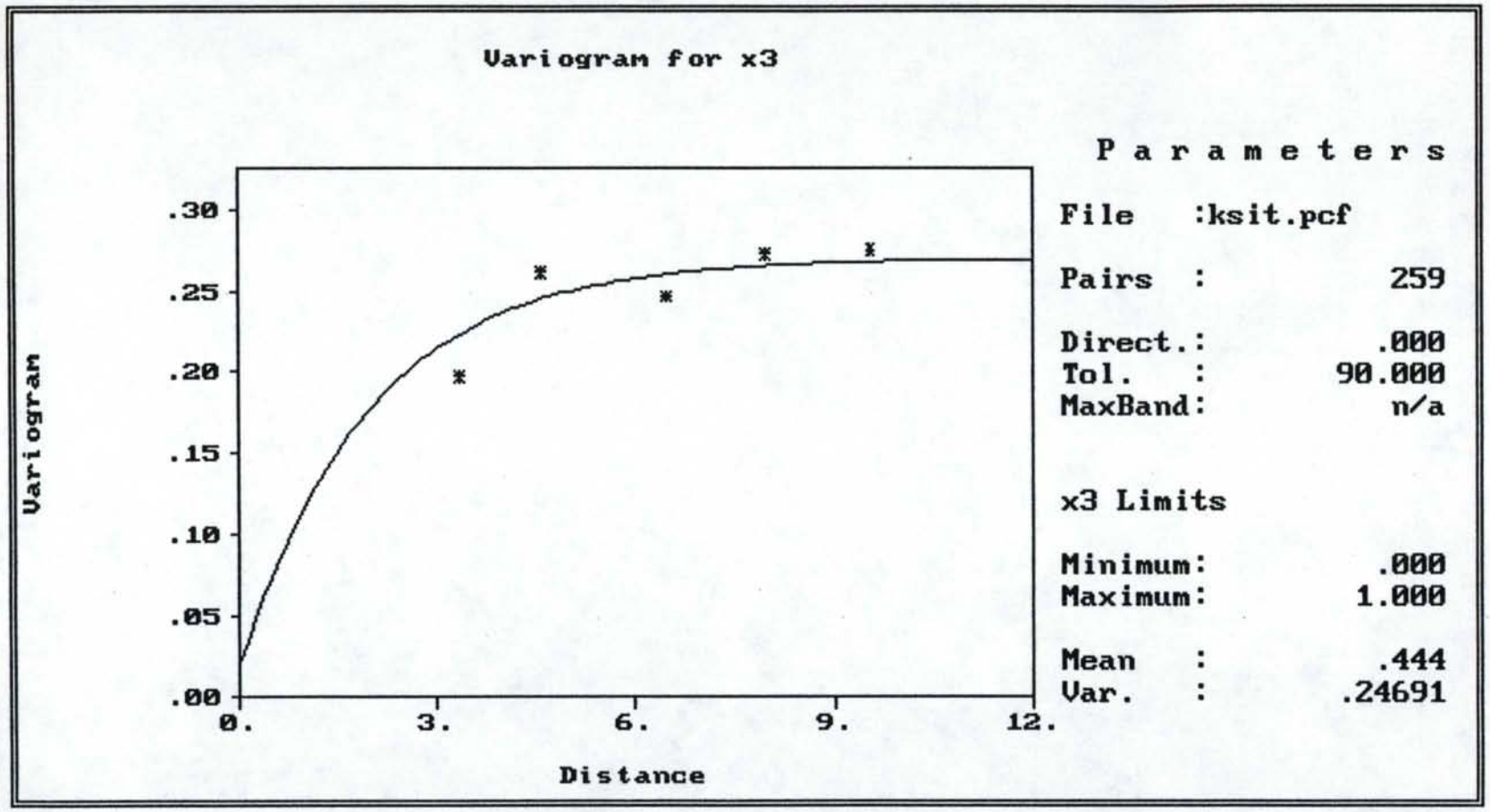
Var. : .19204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.15
.10
.05
.00
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6.
9.
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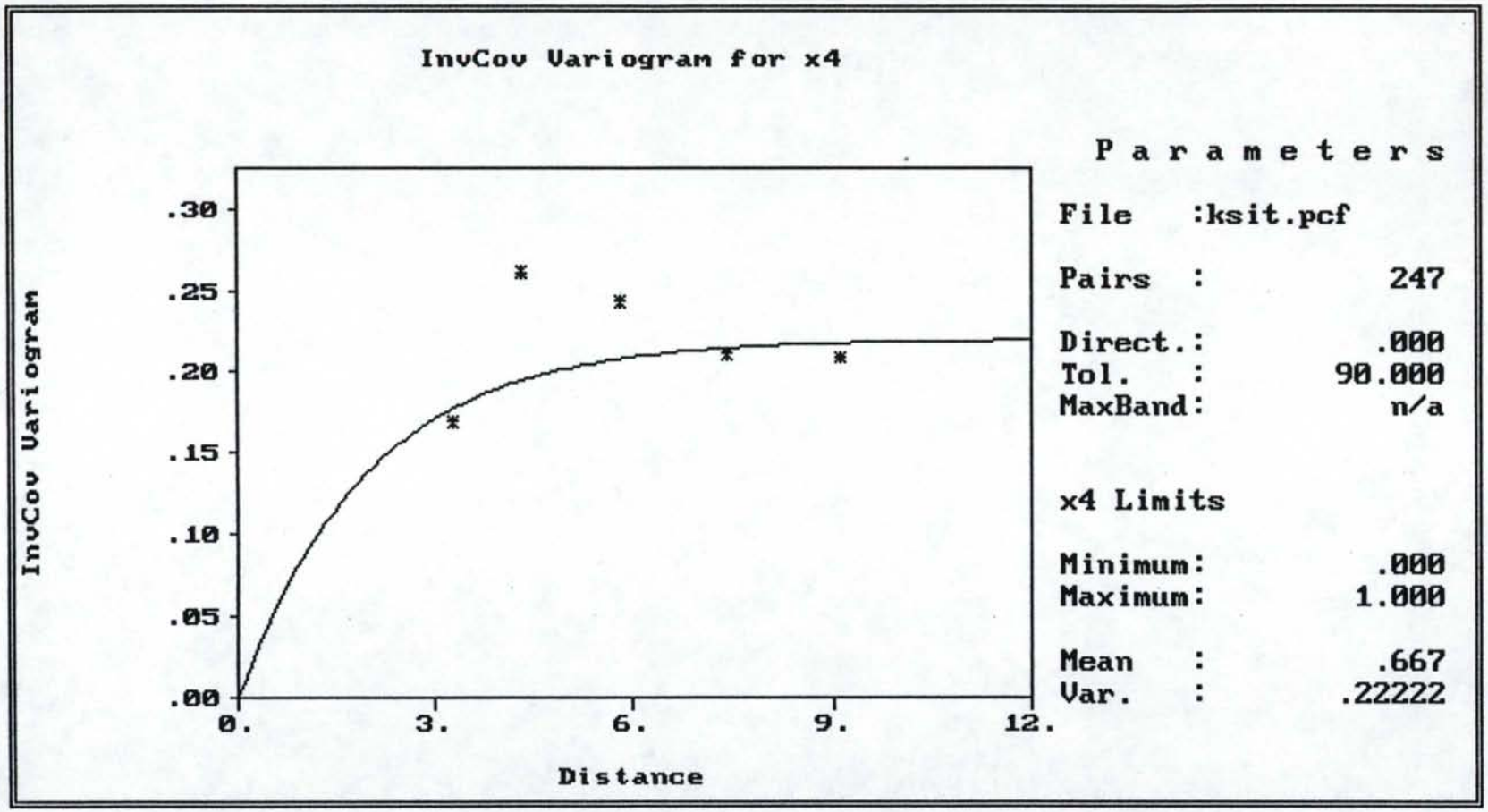
Distance
Parameters
File :ksit.pcf
Pairs : 259
Direct.: .000
Tol. : 90.000
MaxBand: n/a
x3 Limits
Minimum: .000
Maximum: 1.000
Mean : .444
Var. : .24691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.25
.20
.15
.10
.05
.00
0.
3.
6.
9.
12.
Distance
Parameters
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Pairs : 247
Direct.: .000
Tol. : 90.000
MaxBand: n/a
x4 Limits
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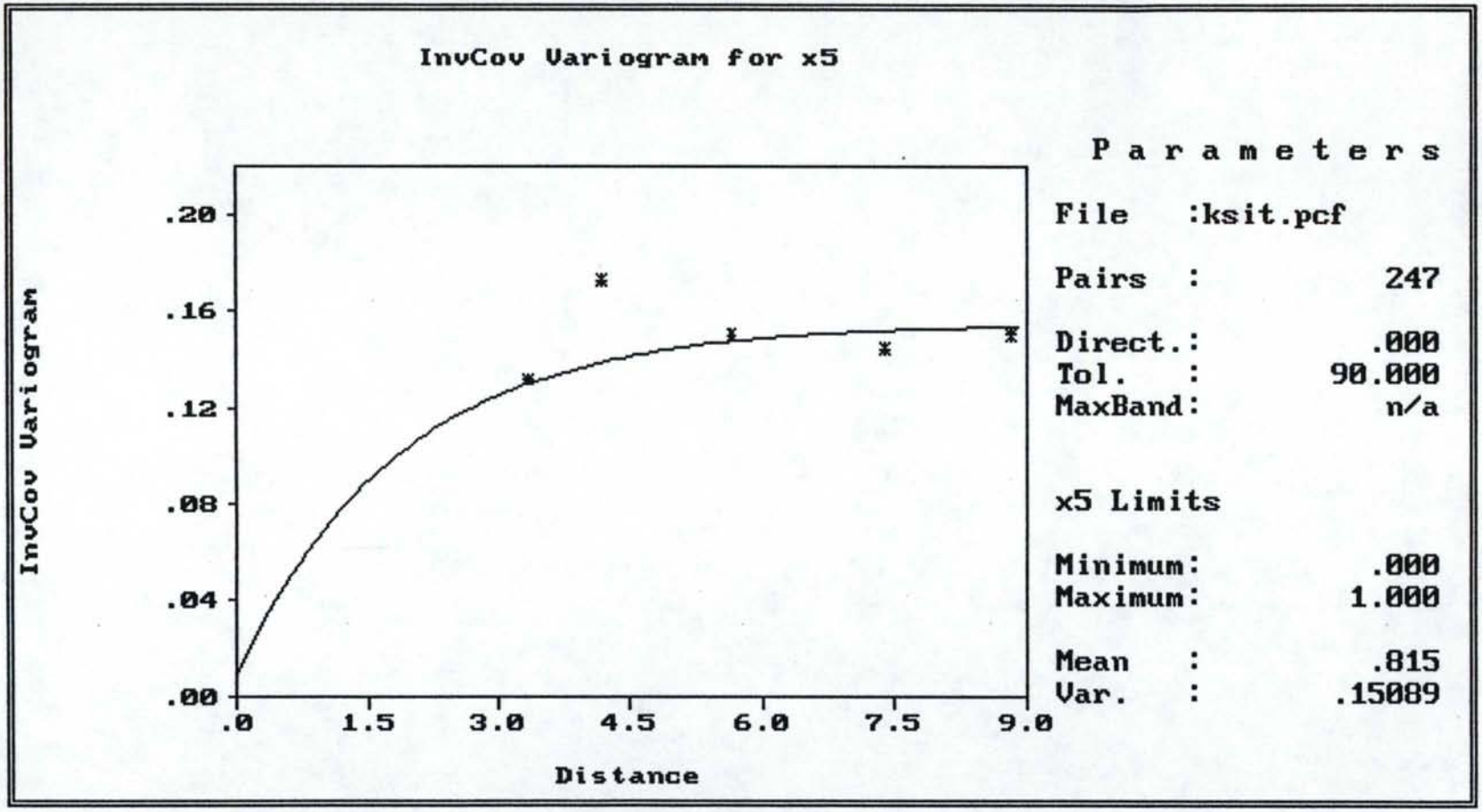
Maximum: 1.000
Mean : .667
Var. : .22222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6.0
7.5
9.0
Distance
Parameters
File :ksit.pcf
Pairs : 247
Direct.: .000
Tol. : 90.000
MaxBand: n/a
x5 Limits
Minimum: .000
Maximum: 1.000
Mean : .815
Var. : .15089

APPENDIX B

Chemical Data

Landfill L2 Kjeldahl-N Concentrations (mg/l)

date	03/20/93	04/16/93
xfer blank		
1	7.4	0.6
2	7.8	2.2
0.5 m		
2A	300	510
2B	300	370
7A	110	130
7B	89	120
12A	49	140
12B	210	170
1.0 m		
4A	180	280
4B	180	290
5A	21	28
5B	32	48
10A	70	90
10B	47	50
2.0m		
1A		6.1
1B	10	21
8A	41	45
8B	29	
9A	15	12
9B		
3.0m		
3A	75	
3B	130	170
6A	36	22
6B	33	
11A	16	13
11B	28	50
4.0m		
13	9.1	9.3
14	45	53
15	19	18
1.0m wall		
16	120	48
18	170	430
20	170	360
.5m wall		
17	170	350
19	19	590
21	190	400

Landfill L2 Ammonium-N Concentrations (mg/L)

date	11/23/92	01/07/93	02/13/93	03/20/93	04/16/93	05/24/93	10/08/93	03/18/94
trip blank	BDL	BDL	BDL	0.1	BDL	BDL	BDL	BDL
xfer blank								
1	BDL		BDL	BDL	BDL	BDL	BDL	BDL
2	BDL		BDL	BDL	BDL	BDL	BDL	
0.5 m								
2A	BDL	23	81	150	190	200	210	300
2B	0.1		130	130	20	180	210	290
7A	BDL		89	27	150	190	190	220
7B	0.2		68	26	100	43	120	150
12A	BDL		53	72	87	120	9.2	120
12B	4.75		51	71	92	190	110	98
1.0 m								
4A			1.1	26	58	94	81	150
4B	0.1		0.7	64	91	130	120	130
5A	BDL	BDL	0.2	BDL	0.4	0.2	1.7	0.1
5B	0.2		0.8	0.9				
10A	BDL		2.2	0.6	7.3	38	16	0.1
10B	BDL		0.8	2.4	3.6	41	6	0.4
2.0 m								
1A	BDL	BDL		BDL	BDL	BDL	1	0.3
1B	BDL		BDL	BDL	0.2	BDL	2.9	0.6
8A	BDL		150	5.1	2.9	0.3	6.2	1
8B	BDL		55	9.1				3.2
9A	BDL		0.1	0.4	0.3	0.2	0.4	0.5
9B	BDL	BDL	BDL	0.1				
3.0 m								
3A	BDL	BDL	BDL	1.1	21			41
3B	BDL	BDL	0.1	BDL	4.2	0.6	41	14
6A	0.2		BDL	0.7	1.6	0.2		0.5
6B	0.3	BDL	5.5	0.9				
11A	0.1	BDL	0.3	1.7	3.6	1	2.6	0.5
11B	BDL		0.2	1.9	9.3	110	8	1.1
4.0 m								
13	BDL		BDL	0.2	0.3	0.2	2.1	BDL
14	0.1		BDL	1.3	9.9	0.5	0.3	0.2
15	0.1	BDL	BDL	0.4	1.9	0.1	350	0.1
1.0 m wall								
16	0.6		0.3	30	110	180	300	
18	0.1		0.2	31	120	170	320	460
20	BDL		0.2	20	95		280	320
0.5 m wall								
17			0.2	23	87	170	190	380
19	0.1		BDL	110	160	210	360	530
21			0.5	31	120	160	370	420

Landfill L2 Nitrate-N Concentrations (mg/L)

date	11/23/92	01/07/93	02/13/93	03/20/93	04/16/93	05/24/93	10/08/93	03/18/94
trip blank	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
xfer blank								
1	BDL		BDL	BDL	BDL	BDL	BDL	BDL
2	BDL		BDL	BDL	BDL	BDL	BDL	
0.5 m								
2A	14	4.8	BDL	BDL	BDL	BDL	BDL	BDL
2B	28		2.8	BDL	BDL	BDL	BDL	BDL
7A	18		0.4	BDL	BDL	BDL	BDL	BDL
7B	21		BDL	BDL	BDL	BDL	BDL	BDL
12A	5.2		0.7	BDL	BDL	BDL	BDL	BDL
12B	5.2		0.6	BDL	BDL	BDL	BDL	BDL
1.0 m								
4A			1.05	0.7	0.5	0.1	BDL	BDL
4B	7.6		2.1	1	0.6	0.1	BDL	BDL
5A	8.3	0.4	BDL	BDL	BDL	0.2	BDL	BDL
5B	3.7		0.4	BDL				
10A	8.3		0.2	BDL	BDL	BDL	BDL	BDL
10B	8.1		0.2	BDL	BDL	BDL	BDL	BDL
2.0 m								
1A	16	41		BDL	BDL	BDL	BDL	BDL
1B	7.3		BDL	BDL	BDL	BDL	BDL	BDL
8A	1.3		BDL	BDL	BDL	BDL	BDL	BDL
8B	0.7		BDL	BDL				BDL
9A	7.1		BDL	BDL	BDL	BDL	BDL	BDL
9B	3.4	0.3	BDL	BDL				
3.0 m								
3A	4.3	14	0.4	BDL	BDL			BDL
3B	5.3	19	1.7	BDL	BDL	BDL	BDL	BDL
6A	0.3		BDL	BDL	BDL	BDL		BDL
6B	5.5	7.2	0.6	BDL				
11A	9.8	BDL	BDL	BDL	BDL	BDL	BDL	BDL
11B	7.8		0.1	BDL	BDL	BDL	BDL	BDL
4.0m								
13	35		0.8	BDL	0.2	BDL	BDL	BDL
14	12		0.2	BDL	0.1	0.1	BDL	BDL
15	10	1	BDL	BDL	BDL	BDL	BDL	BDL
1.0 m wall								
16	0.3		3.7	1.3	0.3	BDL	BDL	
18	0.3		0.5	1.1	0.5	BDL	BDL	BDL
20	0.6		BDL	1.9	0.4		BDL	BDL
0.5 m wall								
17			0.5	BDL	BDL	BDL	BDL	BDL
19	2.2		3.6	1.5	1	BDL	BDL	BDL
21			BDL	1.4	BDL	BDL	BDL	BDL

Landfill L1 Nitrate-N Concentrations (mg/l)

date		1/25/92	2/8	2/22	3/10	4/5	5/3	6/1	7/15	9/21	2/12/93
0.5 m											
A	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	3	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL		
	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL		BDL
	3	BDL	BDL	BDL	BDL	BDL	BDL	0.4	BDL	BDL	BDL
C	1	0.2	0.1	BDL	BDL	BDL	BDL	0.2	BDL	BDL	BDL
	2	0.2	0.2	BDL	BDL	BDL	BDL	0.4	BDL	BDL	0.2
	3	0.3	0.3	BDL	BDL	BDL	BDL		BDL	BDL	0.1
1.0 m											
A	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	
	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	
	3	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	1	BDL		BDL	BDL				BDL	BDL	
	2	BDL	BDL	BDL	BDL	BDL	BDL	0.1	BDL	BDL	BDL
	3	BDL	BDL	BDL	BDL	BDL	BDL	0.3	BDL	BDL	BDL
C	1	0.1	BDL	BDL	BDL	BDL		0.4	BDL	BDL	BDL
	2	0.1	0.2	BDL	BDL	BDL	BDL	0.4	BDL	BDL	BDL
	3	0.2	0.3	BDL	BDL	0.2	BDL	BDL	BDL	BDL	0.2
2.0 m											
A	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	3	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	1	BDL	BDL	BDL	BDL		BDL	0.1	BDL	BDL	BDL
	2	BDL	BDL	BDL	BDL	BDL	BDL	0.2	BDL	BDL	BDL
	3	BDL	BDL	BDL	BDL	BDL	BDL	0.3	BDL	BDL	BDL
C	1	BDL	BDL	BDL	BDL	BDL	BDL	0.4	BDL	BDL	BDL
	2	0.2	0.2	BDL	BDL	BDL	BDL	0.4	BDL	BDL	BDL
	3	0.2	0.3	BDL	BDL	0.2	BDL	0.4	BDL	BDL	0.2

Landfill L1 Kjeldahl-N Concentrations (mg/l)

date		1/25/92	2/22	5/3	6/1	7/15	9/21	2/12/93
0.5 m								
A	1	13.00	25.00	15.00	21.00	9.50		31.00
	2	44.00	36.00	30.00	37.00	18.00	48.50	71.00
	3	34.00	44.00	72.00	77.00	7.90		150.00
B	1	320.00	340.00			40.00		
	2	53.00				13.00		
	3	78.00	85.00	90.00	120.00	12.00		
C	1	76.00	100.00	160.00	200.00	170.00	240.00	290.00
	2	95.00	170.00	210.00	240.00	140.00		
	3	31.00	26.00	46.00	41.00	29.00		81.00
1.0 m								
A	1	17.00	15.00	24.00	24.00	12.00		
	2	19.00	16.00		30.00	43.00		
	3	16.00	4.00	2.10	2.30	3.10		15.00
B	1					41.00		
	2	24.00	12.00	12.00	14.00	110.00		20.00
	3	41.00	27.00	32.00	27.00	150.00		48.00
C	1	37.00	40.00		21.00	13.00		30.00
	2	20.00	20.00	17.00	15.00	12.00		33.00
	3	13.00	11.00	5.40	12.00	11.00		27.00
2.0 m								
A	1	13.00	12.00	6.50	8.70	7.50		
	2	5.00	7.00	3.50	3.40	2.50		37.00
	3					9.00		38.00
B	1	13.00	7.00	8.70	13.00	4.40		17.00
	2	13.00	7.00	6.30	6.90	6.20	11.00	7.10
	3			0.70	BDL	2.20		15.00
C	1	15.00	25.00	2.50	3.10	38.00		16.50
	2			0.90	BDL	1.50		2.60
	3			0.40	BDL	2.00		14.00
pit								
	1		57.00	220.00	601.00	600.00	650.00	
	2		68.00	770.00	746.00	370.00	970.00	

Landfill L1 Ammonium-N Concentrations (mg/l)

date		1/25/92	2/22	5/3	6/1	7/15	9/21	2/12/93	5/24	10/8	3/18/94
0.5 m											
A	1	0.1	0.5	0.7	0.2	0.8	1.4	2.1	0.4	3	1.8
	2	0.5	16.0	0.1	2.9	1.9	1.1	2.4	1.5	1.2	1.9
	3	BDL	1.3	17.0	27.0	190.0	32.0	75			
B	1	160.0	2.0	260.0	390.0	230.0					
	2	0.6	0.8	7.5	23.0	32.0		230			
	3	1.3	21.0	45.0	66.0	51.0	71.0	75			
C	1	1.1	19.8	28.0	45.0	45.0	90.0	160	130		220
	2	43.0	80.0	110.0	150.0	130.0	150.0	210	160	220	220
	3	0.2	BDL	0.1	3.2	3.5	0.5	6.9	6.5		0.6
1.0 m											
A	1	0.2	BDL	0.6	0.7	0.2	0.6		2.9		BDL
	2	1.0	0.2	1.4	0.3	1.9	1.0		0.7		
	3	BDL	BDL	BDL	BDL	0.2	BDL	BDL	BDL	1.2	BDL
B	1					5.3					
	2	BDL	BDL	0.2	1.5	1.6	0.4	2	1.8	1	1.7
	3	BDL	0.4	1.4	3.3	1.9	0.5	2.2	1.7	BDL	0.5
C	1	BDL	0.3		0.1	BDL	2.9	1.6	1.3	2.5	
	2	0.2	BDL	0.1	0.1	0.2	0.8	1	9		0.6
	3	BDL	BDL	12.0	BDL	BDL	1.5	1.7	0.5	0.1	0.2
2.0 m											
A	1	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	
	2	BDL	BDL	BDL	BDL	BDL	BDL	0.8	BDL	0.2	0.8
	3	0.5	BDL	BDL	BDL	BDL		BDL			
B	1	BDL	0.7	BDL	BDL	0.3	1.5	1.8	1.4	BDL	1.2
	2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	1.4
	3	BDL	BDL	BDL	BDL	0.2	0.1	0.2	0.4	1	0.2
C	1	BDL	BDL	BDL	0.2	BDL	1.3	1.1	0.4	1.2	2.3
	2	BDL	BDL	BDL	BDL	BDL	BDL	0.1	BDL	1.3	2
	3	0.8	BDL	BDL	BDL	0.3	BDL	0.1	BDL	BDL	BDL
pit											
	1		127.0	160.0	290.0	190.0	440.0			490	420
	2		2.7	210.0	350.0	170.0	440.0		200	150	430