GROUNDWATER FLOW

AND

CONTAMINANT TRANSPORT ANALYSIS

OF THE

KITZVILLE DUMP,

ST. LOUIS COUNTY, MINNESOTA

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ABSTRACT OF THESIS

GROUNDWATER FLOW AND CONTAMINANT TRANSPORT ANALYSIS OF THE KITZVILLE DUMP, ST. LOUIS COUNTY, MINNESOTA

by Scott L. Turner

The Kitzville Dump was used as a municipal and industrial solid waste disposal site for approximately 35 years. After the site was closed and capped in 1981, chemical analyses of monitoring wells installed at the site indicated leachate was entering the groundwater.

Two confined glacial outwash aquifers overlie Precambrian bedrock at the site. Hydrogeologic characteristics determined at the site indicate the general groundwater flow direction to be from the northwest to the southeast at an average rate of 20 ft/yr.

A three-layer groundwater flow model provided the basis for analysis of solute transport at the site. The solute transport model was initially calibrated to chloride. Additional contaminants analyzed were benzene, toluene, cadmium, and lead. Retardation factors were determined for each of these chemical species.

Results of the solute transport model indicate that the contaminant plume is moving in the general direction of groundwater flow. The migration rate varies among the chemical species analyzed, with chloride having the highest and cadmium the lowest rate.

The extent of the contaminant plume does not pose a threat to any existing water supplies; however, future development in the area would necessitate further testing to ensure that health standards are maintained.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS ii

ABSTRACT iii

LIST OF TABLES vi

LIST OF FIGURES vii

SECTION PAGE

I. INTRODUCTION 1

Site Location 2
Objectives of Research 2
Research Methods 4
Previous Work 4

II. REGIONAL AND SITE GEOLOGY 8

Bedrock Geology 8
Geology of Glacial Sediments 11
Geomorphology 16

III. REGIONAL AND SITE HYDROGEOLOGY 18

Groundwater Elevations 20
Hydraulic Conductivity 20
Hvorslev Method 20
In-situ Permeameter Tests 23
Groundwater Flow Velocities and Gradients 26

IV. NUMERICAL MODEL OF SITE GROUNDWATER FLOW 28

Introduction 28
Data Preparation 31
HELP Model 36
Model Simulation 38
Model Calibration 39
Parameter Uncertainty 40

Sensitivity Analysis 42

V. REGIONAL AND SITE GROUNDWATER QUALITY 45

Water Quality Standards 46
Leachate Characteristics 47

VI. NUMERICAL MODEL OF SITE CONTAMINANT TRANSPORT . . 51

Introduction 51
Preliminary Geophysical Investigation 55
Electrical Resistivity Survey 55
Seismic Survey 56
Data Preparation 57
Contaminant Selection Criteria 58
Model Simulation 59
Model Calibration 59
Sensitivity Analysis 62
Organics 63
Benzene 64
Toluene 65
Metals 66
Lead 67
Cadmium 67
Predictive Simulation 67

VII. SUMMARY AND CONCLUSIONS 75

REFERENCES 81

APPENDICES 85

Appendix A Soil Boring Logs 85
Appendix B Hydraulic Conductivity Data 91
Appendix C Permeameter Field Data 100
Appendix D Static Water Level Data 105
Appendix E Chemical Analytical Data 111
Chloride 111
Sulfate 112
Ammonia 113
Iron 114
Nitrate 115
COD 116
Field Specific Conductance . . 117
Field/Lab pH Data 118
Field Water Temperature Data . 120
Organics and Metals Analysis . 121

LIST OF TABLES

TABLE 1 Aquifer Hydraulic Conductivity Values 24

TABLE 2 Comparison of Field and Lab Hydraulic Conductivity Values 25

TABLE 3 Average Annual Totals for HELP Model Simulation of Operating Kitzville Dump . . 37

TABLE 4 Average Annual Totals for HELP Model Simulation of Closed Kitzville Dump 38

TABLE 5 U.S. EPA Standards for Contaminant Levels . . 47

TABLE 6 Typical Leachate Composition Values of Specified Contaminants 49

TABLE 7 Summary Table of Flow and Transport Parameters as Optimized in this Study . . . 79

LIST OF FIGURES

FIGURE 1 Location Map of Kitzville Dump 3

FIGURE 2 Cross-Section of Kitzville Dump 11

FIGURE 3 Morphology of Kitzville Dump 16

FIGURE 4 Aquifer and Confining Units Cross-Section . . 19

FIGURE 5 Contour Map of Observed Hydraulic Head Values 21

FIGURE 6 Monitoring Well Location Map 22

FIGURE 7 Cross-Section of Site Illustrating Modeled Layers 32

FIGURE 8 Finite-Difference Grid Computer Notation . . . 34

FIGURE 9 Grid Map of Study Area 35

FIGURE 10 Contour Map of Groundwater Model Generated Head Values 41

FIGURE 11 Contour Map of Present Chloride Values 50

FIGURE 12 Contour Map of Apparent Resistivity Values . . 56

FIGURE 13 Contour Map of Pre-Closing Chloride Concentration Values 61

FIGURE 14 Contour Map of Solute Transport Model Generated Chloride Concentration Values (Present) 62

FIGURE 15 Concentration Contour Map of Benzene (1990) . 65

FIGURE 16 Concentration Contour Map of Toluene (1990) . 66

FIGURE 17 Concentration Contour Map of Lead (1990) . . . 68

FIGURE 18 Concentration Contour Map of Cadmium (1990) . 69

FIGURE 19 Concentration Contour Map of Chloride (2000) . 70

FIGURE 20 Concentration Contour Map of Benzene (2010) . 71

FIGURE 21 Concentration Contour Map of Toluene (2010) . 72

FIGURE 22 Concentration Contour Map of Lead (2010) . . . 73

FIGURE 23 Concentration Contour Map of Cadmium (2010) . 74

SECTION I

INTRODUCTION

The growing concern for the environment by the general public, private interest groups, and government agencies has led to an increased awareness of the effects of solid waste disposal. This awareness has manifested itself in the form of increased regulation at all levels of government, in order to lessen the environmental impact of the tremendous volumes of solid waste generated each year. The present primary disposal method of most municipal and industrial solid waste is burial in a landfill (Fetter, 1988). [The term "landfill" has replaced the term "dump" for solid waste disposal sites opened after implementation of Minnesota Pollution Control Agency (MPCA) regulations in 1972. Throughout this paper these terms will be used interchangeably, except where specified.]

The major cause for concern is the potential for surface waters to infiltrate the waste and leach compounds from the solid waste. The resulting liquid or leachate can then move outward to surface waters and/or downward from the landfill, enter the groundwater, and cause contamination. The growing utilization of groundwater in households, agriculture, and industry requires an increased awareness of possible sources of contamination and their potential effects on the environment. The Kitzville Dump is one such source and this paper will examine the environmental impact of leachate from this site. Even though the Kitzville Dump is unique in some respects, the basic assumptions and estimates used in this study may be applied to other solid waste disposal sites in similar hydrogeological environments.

SITE LOCATION

The Kitzville Dump is approximately one mile east of Highway 169 between Hibbing and Chisholm in St. Louis County (Figure 1). The site, which covers about 40 acres, is in the SW 1/4, NW 1/4, Section 3, T. 57 N., R. 20 W. on the United States Geological Survey's Buhl 7 1/2' topographic quadrangle.

OBJECTIVES OF RESEARCH

This site study has four primary objectives: (1) to ascertain the characteristics of the regional and site geology/hydrogeology necessary for analysis of groundwater flow and solute transport at the study site; (2) to determine the direction and rate of groundwater flow at the site through application of a mathematical groundwater flow model; (3) to determine the movement and retardation values of selected contaminants at the site through application of

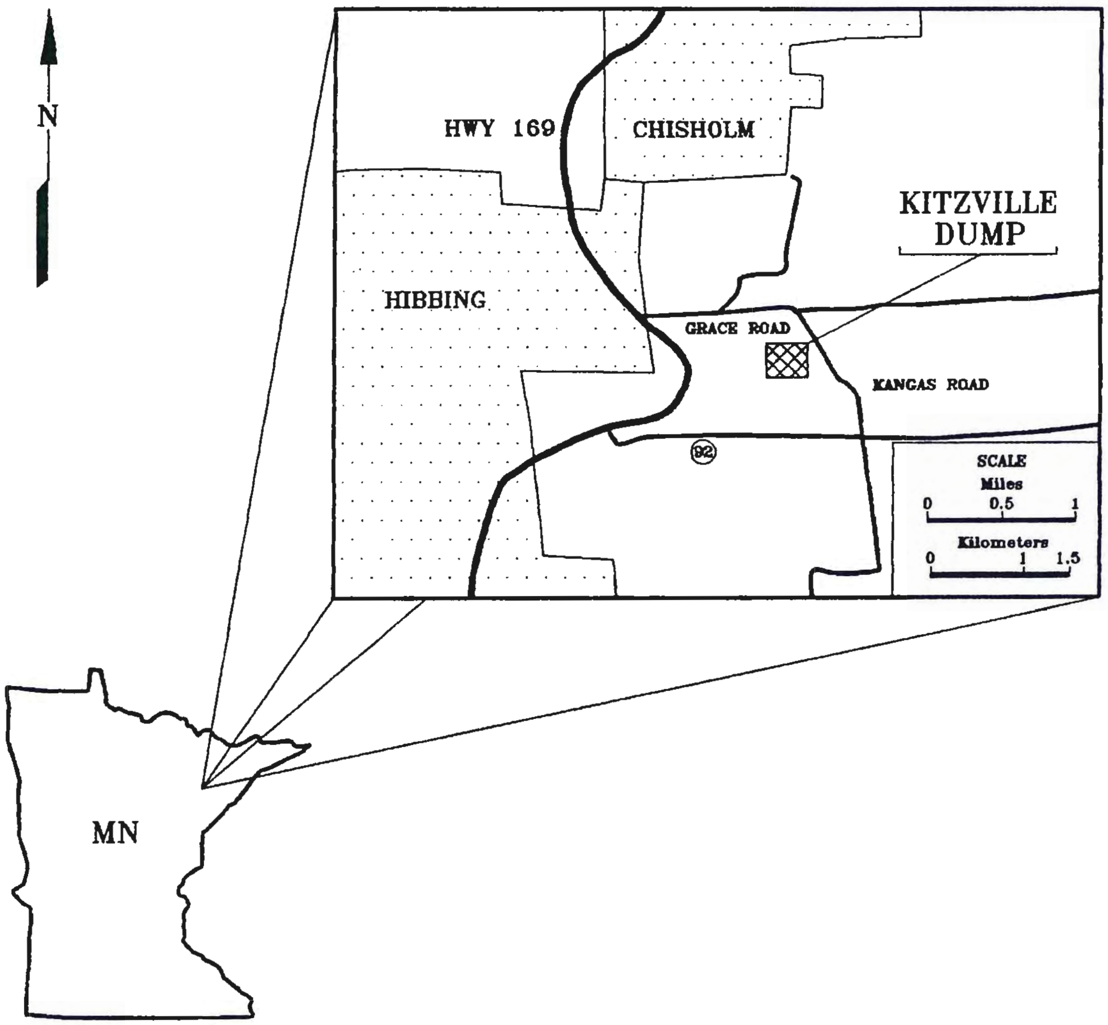

Figure 1 Location Map of Kitzville Dump.

a mathematical solute transport model utilizing flow parameters obtained from the groundwater flow model and other processes affecting transport; and (4) to utilize these models to evaluate the present and future environmental impact of leachate from this site.

RESEARCH METHODS

Groundwater modeling has become one of the foremost methods used in analyzing groundwater systems. Groundwater models have proven to be effective in increasing our understanding of groundwater systems in a variety of hydrogeological environments (Mercer and Faust, 1981). In view of this proven effectiveness, the primary instrument of this site analysis is the mathematical groundwater model. Therefore, the analyses and conclusions put forth in this paper rely predominantly on results obtained from the groundwater flow and solute transport models.

Sections II and III of this report, the regional and site geology/hydrogeology, are based on published papers and field work. The field work consisted of a seismic survey to better delineate the extent of the solid waste; in situ permeameter tests to determine hydraulic conductivity values of surficial materials; aquifer tests to determine hydraulic conductivity values of aquifer materials; and an apparent resistivity survey to define the present extent of the contaminant plume.

Section IV encompasses the application of the numerical groundwater flow model to the study site. A brief introduction to mathematical groundwater flow modeling is presented, followed by a description of the parameters, methods and basic assumptions utilized in this study. Fundamental equations used by the flow model are also explained. This section closes with an analysis of groundwater flow rates and gradients based on the groundwater flow model's computed results.

Section V covers the regional and site groundwater quality and discusses the water quality standards as they apply to the examined chemical species. Application of the solute transport model to the study site is covered in Section VI. This section presents a brief introduction to solute transport modeling, followed by a description of the parameters, selected contaminants, methods, and assumptions utilized in the transport model. The equations used to arrive at the final computed concentrations are also examined. The predictive capabilities of the model are then utilized to examine the future extent of the contaminant plume.

Section VII provides a summary of the site study and contains concluding remarks concerning the results of the groundwater flow and transport models. This section closes with a discussion of the environmental impact of the contaminant plume as identified in this study.

PREVIOUS WORK

The Kitzville Dump was used as a municipal and industrial solid waste disposal site by the city of Hibbing beginning in about 1946. In 1981 the dump was closed and capped with a layer of silty sand. Preliminary hydrogeologic and water quality data were submitted to the St. Louis County Health Department and the Minnesota Pollution Control Agency (MPCA) in June, 1982 by RREM, Inc. of Duluth, MN. RREM, Inc had been contracted by St. Louis County to perform this investigation in preparation for the site closing.

During October, 1982 soil borings were taken and six monitoring wells were installed by Braun Engineering Testing to determine the geologic and hydrogeologic characteristics at the site (Braun, 1982). Sampling for general water quality parameters was completed on November 9, 1982. A second round of sampling was performed in January, 1983, after additional well development. The results of these analyses, which indicated the presence of a contaminant plume, were included in a groundwater study submitted by RREM, Inc. to the St. Louis County Health Department and MPCA in April, 1983 (RREM, 1983).

Beginning in mid-1983, St. Louis County and the MPCA instituted routine groundwater quality sampling procedures, which entail sampling each of the six monitoring wells three times a year for various inorganics, chemical oxygen demand, temperature, pH, and conductivity (MPCA, 1984). These sampling procedures are being conducted on a continuing basis.

SECTION II

REGIONAL AND SITE GEOLOGY

A site analysis of groundwater flow and contaminant transport must begin with a thorough understanding of the local and regional geology. The geology in the region of the Kitzville Dump is complex both in structure and lithology. Glacial drift units rest on bedrock of Lower Proterozoic age. These glacial drift units range up to 300 feet in thickness and are the result of various phases of glaciation which occurred during the Pleistocene Epoch (Winter, Cotter, and Young, 1973).

BEDROCK GEOLOGY

Bedrock in the vicinity of the study site is composed of the Pokegama Quartzite, Biwabik Iron Formation, and the Virginia Formation. These units, also known as the Animikie Group, are bordered on the north by the Giants Range Granite, a long, linear, Archean granitic complex (Morey, 1976).

The most prominent bedrock feature in this region is the Biwabik Iron Formation which crops out for approximately 120 miles in an east-northeasterly direction, comprising the Mesabi Range, to the north of the study area. The Biwabik

Iron Formation is underlain by the Pokegama Quartzite and overlain by the Virginia Formation. The overall structure of the Mesabi Range is a southeasterly dipping homocline with an east-northeast strike (Morey, 1972).

The Virginia Formation, which underlies the study area, is composed of interbedded argillite, argillaceous silt-stone, and graywacke (Morey, 1976). There are no known natural exposures of this formation in the vicinity of the study area.

Within this region there is also evidence of lenses of Cretaceous shale overlying the Animikie Group, but its presence in the study area has not been confirmed (Wright, 1972).

Winter (1973) has described two bedrock valleys in the vicinity of the study area. The Hibbing bedrock valley, approximately five miles to the southwest of the Kitzville Dump, is narrow, steep sided, and has relief of 100-150 feet. This valley extends south from Hibbing. The Virginia bedrock valley, 35 miles to the east of the study area, extends southwest from Virginia, is broad and less than 100 feet deep. The thickness of overlying surficial deposits is greatest in these bedrock-valley areas.

GEOLOGY OF GLACIAL SEDIMENTS

Glacial sediments overlie the Animikie Group throughout the study area and most of the surrounding region. These sediments are highly variable in thickness and lithology. This is due primarily to the various advances and retreats of glacier ice during the Pleistocene (Wright, 1972).

The glacial sediments throughout northeastern Minnesota were deposited by various glacial lobes which were controlled primarily by the preglacial bedrock topography. These lobes were all part of the Wisconsin glaciation which was the last major advance of glacier ice into the region. The geomorphology and surficial deposits for most of the state are primarily the result of this last glaciation (Wright, 1972).

There are three primary till units present in the study area: a lower till, middle till and an upper or surficial till. Various glacial outwash deposits are also intermingled with these till units. The following descriptive summaries of these units are based primarily on published reports by Winter and others (1973) of field investigations and six soil boring logs completed by Braun Engineering Testing at the study site (Appendix A). At the study site, glacial sediments, approximately 170 feet in thickness, rest on a bedrock surface estimated to be 1280 feet above MSL (Figure 2). The deepest soil boring at the site extends to a depth of only 87 feet below the surface; consequently, information below this level is inferred from regional data.

The lower till unit is believed to have been deposited

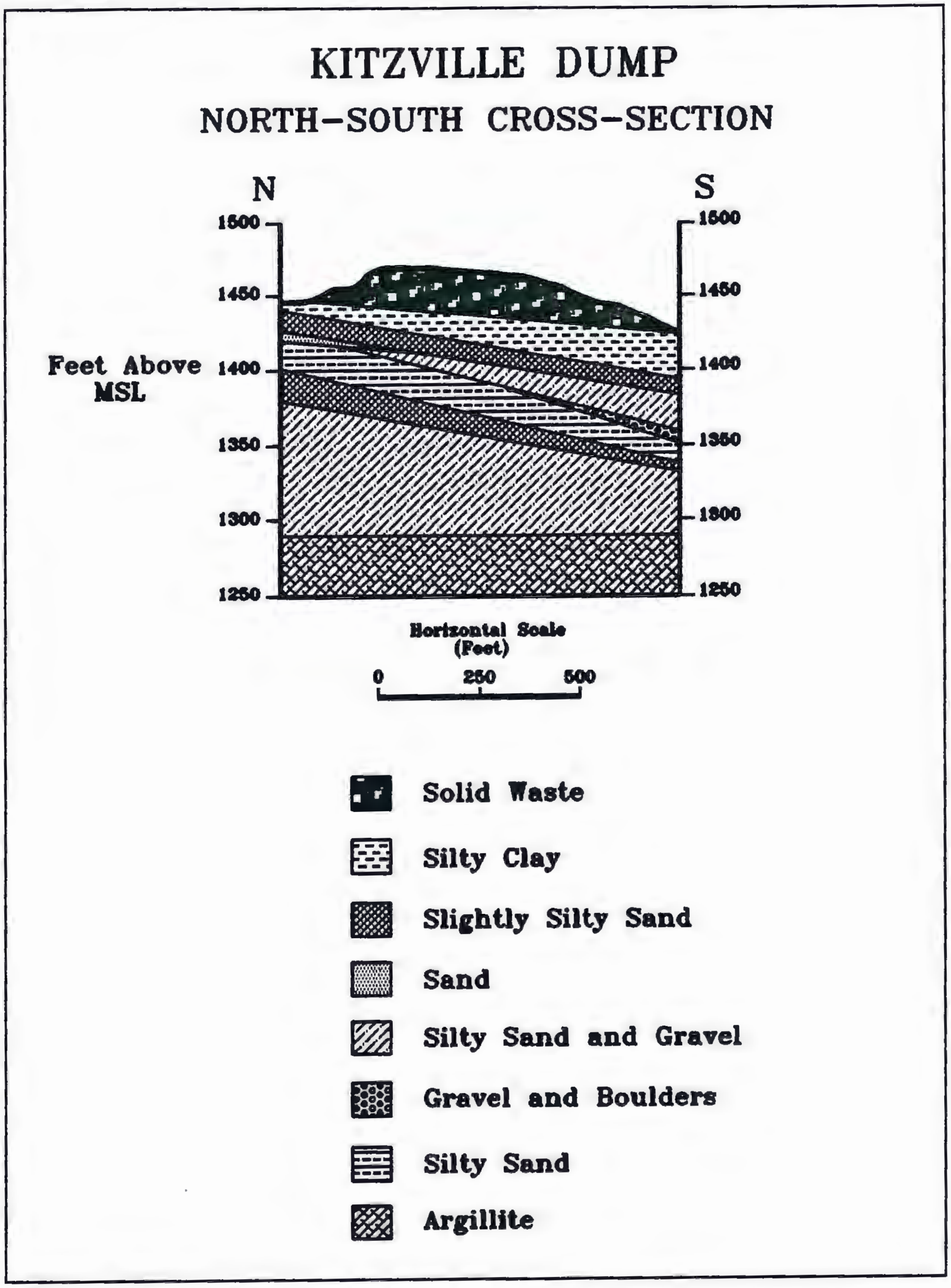

Figure 2 Cross-section of Kitzville Dump, inferred from soil borings and regional data.

by glacier ice moving from the west-northwest during the middle or early Wisconsin (Wright, 1972). The thickness of this till unit varies between 50 and 100 feet, but increases in thickness to well over 100 feet east of Hibbing. This till is calcareous, dark gray to dark greenish and brownish gray, with a sandy to gravelly texture. Cobbles and some boulders are also scattered throughout the till (Winter and others, 1973). The soil boring data provide no additional information concerning this till in the study area.

Little is known about the glacial outwash sediments underlying the lowest till. The only known occurrences of continuous stratified drift underlying the lower till occur near Grand Rapids and Hibbing. Near Hibbing, the lower stratified drift ranges from 6 to 122 feet thick, with the bulk of this being clay and silt. The thickness of sand within the glacial outwash section is generally not more than 30 feet. In general, indications are that glacial outwash sediments underlying the lower till in the Mesabi Iron Range area are not extensive and that most of these sediments are clay and silt (Winter, 1973).

Stratified glacial drift above the lower till and underlying the middle till unit appears to be continuous in the western and central parts of the Mesabi Iron Range. Generally, the glacial outwash deposits at this stratigraphic interval are less than 50 feet thick. Near Hibbing however, this stratified drift is greater than 100 feet thick. Variation in grain size of these deposits is wide, but the deposits consist largely of sand or sand and gravel (Winter and others, 1973). At the study site, soil boring ST-1A penetrated 10 feet into this drift which proved to be a slightly silty sand with some fine gravel.

The middle till unit, sometimes referred to as the bouldery till, has an approximate age of 20,000 to 16,000 years B.P. This till was deposited as the Rainy Lobe moved from the northeast to the southwest across the Giants Range and is relatively continuous across the entire area (Wright, 1972; Goebel and Walton, 1979). The till is absent in the general area of the Virginia bedrock valley and in scattered areas throughout the Iron Range. Thickness of the till is generally less than 50 feet thick, and in much of the area it is less than 25 feet (Winter, 1973). Soil boring ST-1A, at the study site, shows a thickness of 42 feet for this middle till unit.

The middle till is a noncalcareous, gray to yellow, red orange or brown, and is characterized by abundant cobbles and boulders in a sandy, silty matrix. Lenses of stratified glacial outwash deposits are common regionally in the middle till, but none were detected at the study site. The grain size and degree of sorting of the stratified drift is highly variable, but most lenses are local and consist of fine sand and silt (Winter and others, 1973).

The glacial outwash deposits between the middle till and the upper till are the thickest and most continuous of the stratified drift units. Thickness of these sediments is commonly greater than 50 feet. In some areas along the Mesabi Iron Range, the thickness can exceed 100 feet (Winter, 1973). At the study site these deposits range from 10 to 24 feet in thickness. The horizontal variation in grain size is great in this stratigraphic unit. The thicker, coarse-grained bodies of stratified drift are associated with linear bedrock depressions in the Giants Range. These outwash sediments were deposited when the Rainy Lobe ice retreated north of the Giants Range, and sediment-laden meltwaters poured southward through the depressions (Winter and others, 1973).

The upper till was deposited by the St. Louis Sublobe between 14,000 and 12,000 years B.P. The St. Louis Sublobe, part of the larger Des Moines Lobe, moved from the northwest to the southeast across much of northern Minnesota (Wright, 1972). The upper till in the Mesabi Iron Range area is thin, generally less than 25 feet thick, but continuous. Immediately west of both Hibbing and Eveleth, the till is greater than 50 feet thick (Winter and others, 1973). Soil penetrations at the study site indicate a local thickness range of from 4 to 25 feet.

The upper till consists of two subunits (Winter, 1973) distinguished largely by color and to a lesser degree, by grain size of the matrix. In the eastern part of the range, the till is red and clayey and is characterized by a red to reddish-brown clayey, silty, and calcareous matrix. In the western part of the range, the upper till is brown and silty and is characterized by a light to medium-brown sandy, silty, and calcareous matrix (Winter and others, 1973). The difference in these tills is attributed to the variation in sediment picked up by the St. Louis Sublobe as it moved across the region. The red clayey till is believed to have resulted from the red lake clays of Glacial Lake Upham I (Baker, 1964; Wright and Ruhe, 1965).

Glacial outwash and glaciolacustrine sediments cover much of the land surface in the Mesabi Iron Range area. Most of the sediments are associated with glacial lakes Aitkin II and Upham II (Wright, 1972). This stratified drift is generally less than 25 feet thick (Kanivetsky, 1979).

GEOMORPHOLOGY

The study site and surrounding area are characterized by landforms consisting of glacial drift overlying Precambrian bedrock. The glacial drift is bordered on the north by a long linear ridge of hills of Archean rocks that rise 200-400 feet higher than the area to the immediate south. This ridge is known as the Giants Range (Morey, 1976). Common in the areas of till are hummocky topography of morainal and ice-contact landforms that have relief up to 50 feet (Wright, 1972).

Elevations within the study area range from just over 1490 feet (454 m) to just under 1440 feet (439 m). Due to the presence of the solid waste, the topography increases in elevation toward the center of the dump area giving the site

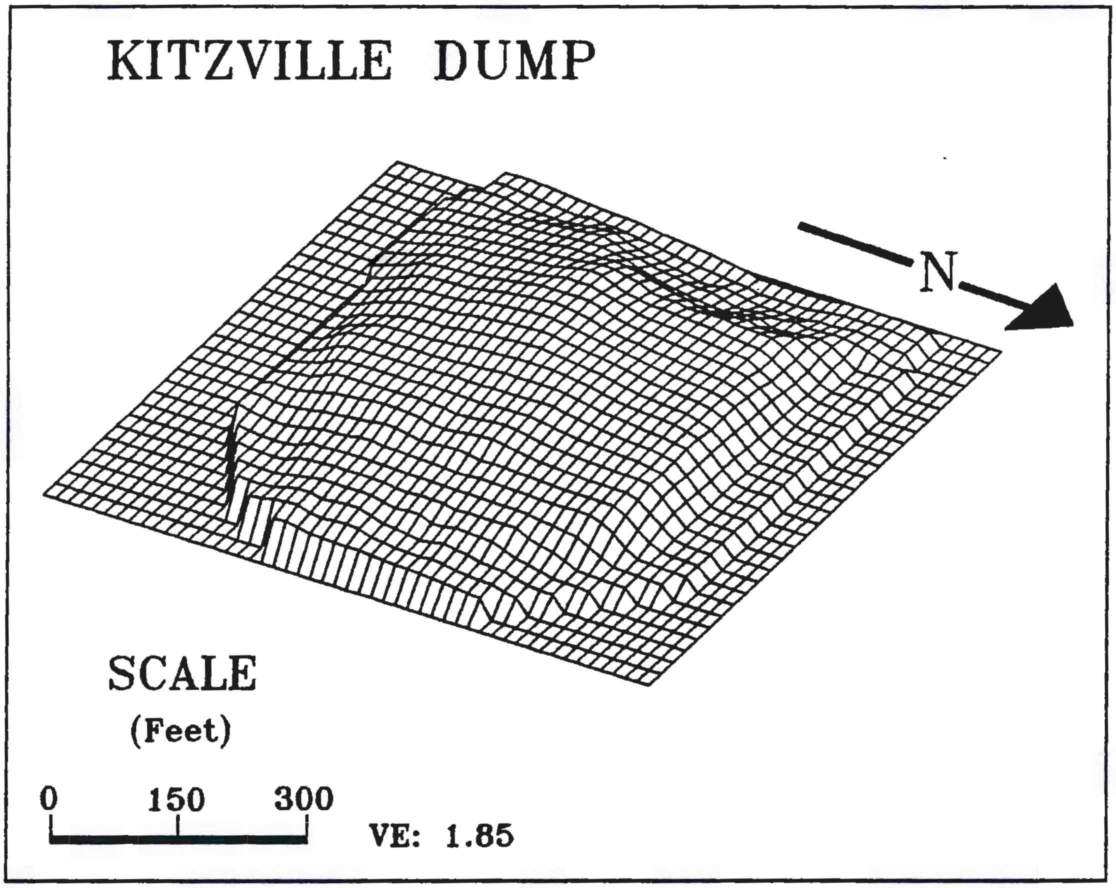

Figure 3 Morphology of Kitzville Dump.

its mounded appearance (Figures 2 and 3). The major topographical features in this area consist of eskers or crevasse fillings, one approximately 1/4 mile (0.4 km) to the north, with two more located approximately one mile (1.6 km) to the south of the site. Each of these eskers is approximately 1/2 mile (0.8 km) in length and trend in a northeast-southwest direction.

Swamps almost completely encircle the site with the largest of these to the south-southeast of the site. The swamps are poorly drained areas which are one of the characteristics of glacial stagnation terrains (Sugden and John, 1976).

In the 10,000 or more years since glacier ice left the area, erosion and deposition have modified the landscape. Deposition of eroded material has been largely confined to lakes and other depressions. Continued deposition of sediment and organic material into the lakes caused many of them to become shallow enough around the margins so vegetation could spread toward the centers, ultimately converting many of the lakes to bogs. Presently, there are more bogs than lakes in northern Minnesota (Wright, 1972).

SECTION III

REGIONAL AND SITE HYDROGEOLOGY

As previously indicated, glacial sediments within the study area are highly variable both in thickness and extent. The major sand and gravel deposits, which are primarily glaciofluvial in origin, constitute the aquifers and occur between the till units (Figure 4). The basal, bouldery and surficial tills, discussed in the previous section, are also termed confining beds in that they inhibit the flow of groundwater from and to the aquifers. These till units generally have hydraulic conductivity (K) values two to three orders of magnitude less than aquifer K values.

There are two principal aquifers located within the study area: a relatively shallow aquifer and a deep aquifer. Both aquifers are confined. The upper, or shallow aquifer is the primary focus of this study because chemical analyses have indicated the presence of a contaminant plume (Appendix E). Chemical analyses performed on well MW-1A, which has its screened interval in the lower aquifer, indicate no contamination has occurred at that level. The thickness of the shallow aquifer ranges from eight feet to more than 23 feet within the study area.

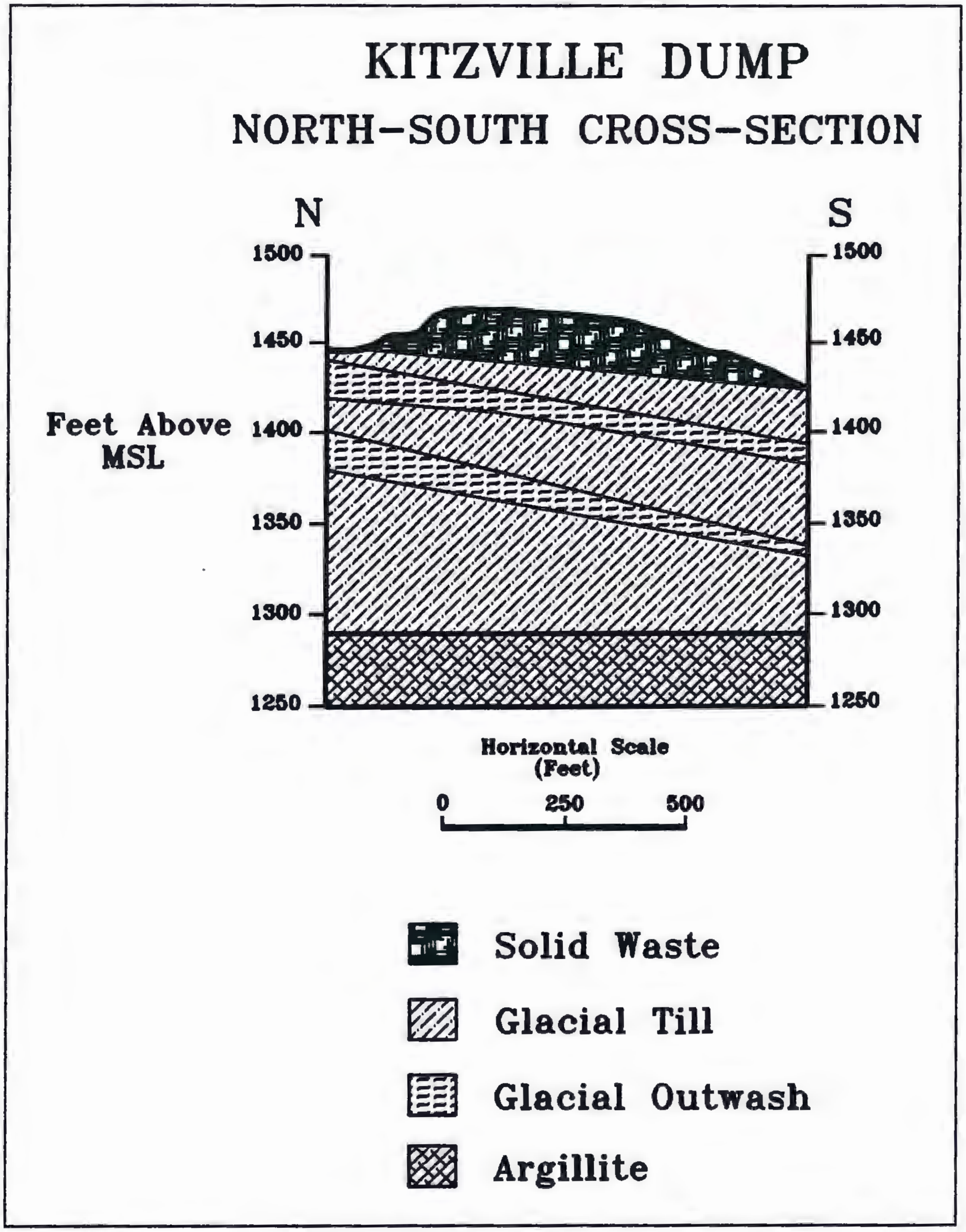

Figure 4 Cross-section of the study site illustrating the glacial outwash aquifers and confining till units.

GROUNDWATER ELEVATIONS

Groundwater elevations in this region range from 1430 to 1450 feet above MSL with regional flow direction from the northwest to the southeast (Lindholm and others, 1979). The groundwater contour map shown in Figure 5 is based on static water level information from October 1989. This contour map indicates that the general groundwater flow direction in the vicinity of the site is from the northwest to the southeast.

HYDRAULIC CONDUCTIVITY

The determination of hydraulic conductivity values of materials at the site involved the utilization of various testing methods. The hydraulic conductivity of the aquifer materials was determined using the Hvorslev method (Freeze and Cherry, 1979). Hydraulic conductivity values of the surficial till and the site cover material were determined by both in-situ and lab permeameter tests.

<u>HVORSLEV METHOD.</u> The hydraulic conductivity of the shallow aquifer was determined by analyzing five of the six monitoring wells located at the site (Figure 6). The remaining monitoring well, MW-1A, has its screened interval in the lower aquifer. A measured volume of water was removed or added to each well, resulting in a decrease or increase in the water level. The level the water rises above or

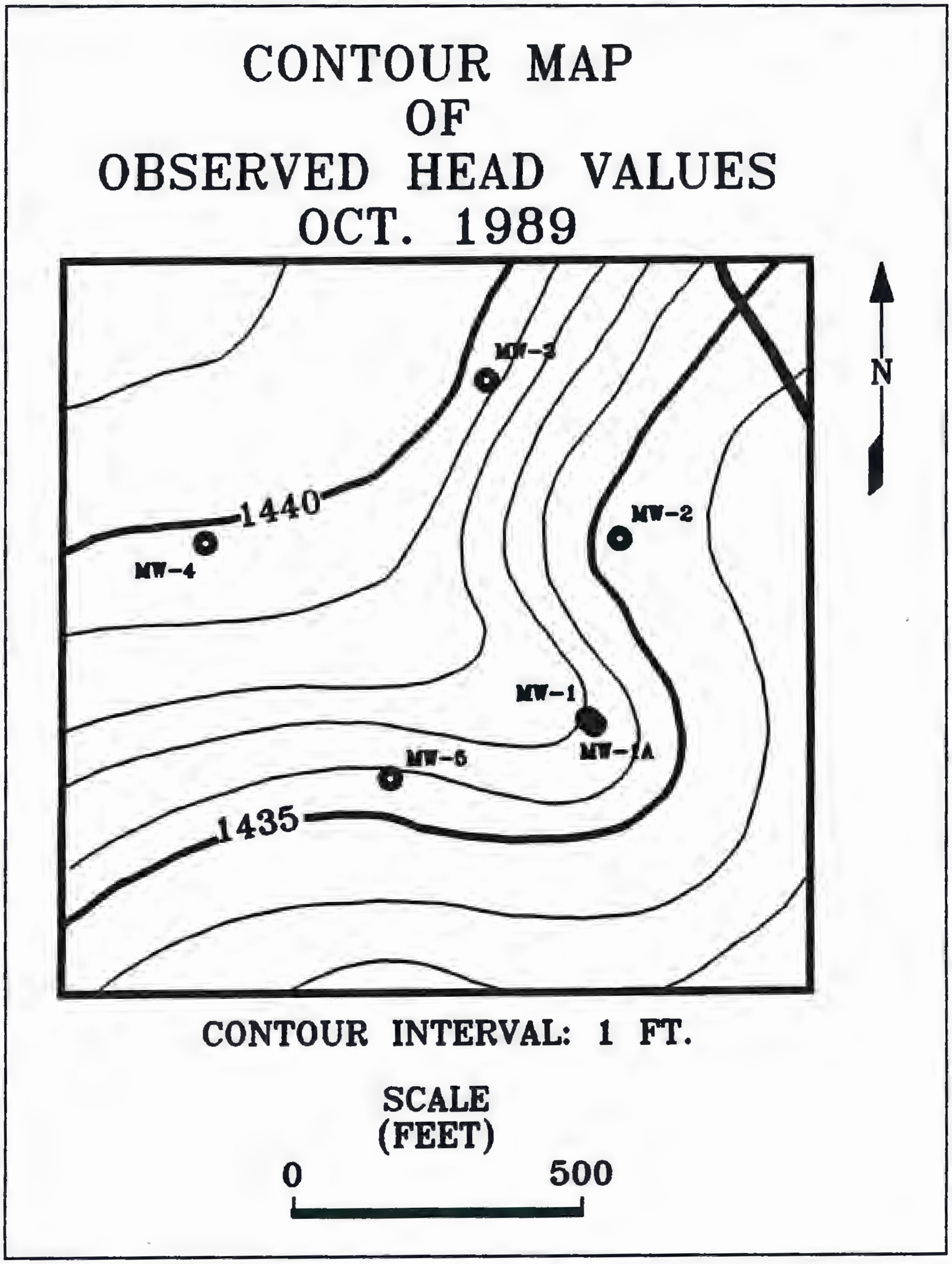

Figure 5 Contour map of observed hydraulic head values (feet), showing well locations.

below static level, immediately upon the addition or removal of this volume of water, is termed H_0. A water level indicator (electronic tape) was then used to measure the water level (H), above or below static level, at timed intervals until it returned to its original elevation. The recovery data was graphed (Appendix B) and K values were determined by the Hvorslev method.

The Hvorslev method plots time on a linear scale and H/H_0 on a logarithmic scale. The data normally plot in a nearly straight line. A best fit line is then applied to the plotted data to determine the basic time lag (t_0). The basic time lag is related to the time required for the water level to return to its original level, assuming the initial rate of inflow was maintained, and corresponds to H/H_0 = 0.37.

The following equation is used to calculate the hydraulic conductivity (K):

$$K = \frac{R^2 \ln(L/R)}{2Lt_o}$$

where R = radius of well casing (cm)
L = length of well screen (cm)
t_0 = basic time lag (sec)

Table 1 lists the hydraulic conductivity values calculated using this equation.

<u>IN SITU PERMEAMETER TESTS.</u> To determine accurately the hydraulic conductivity of surficial materials at the study

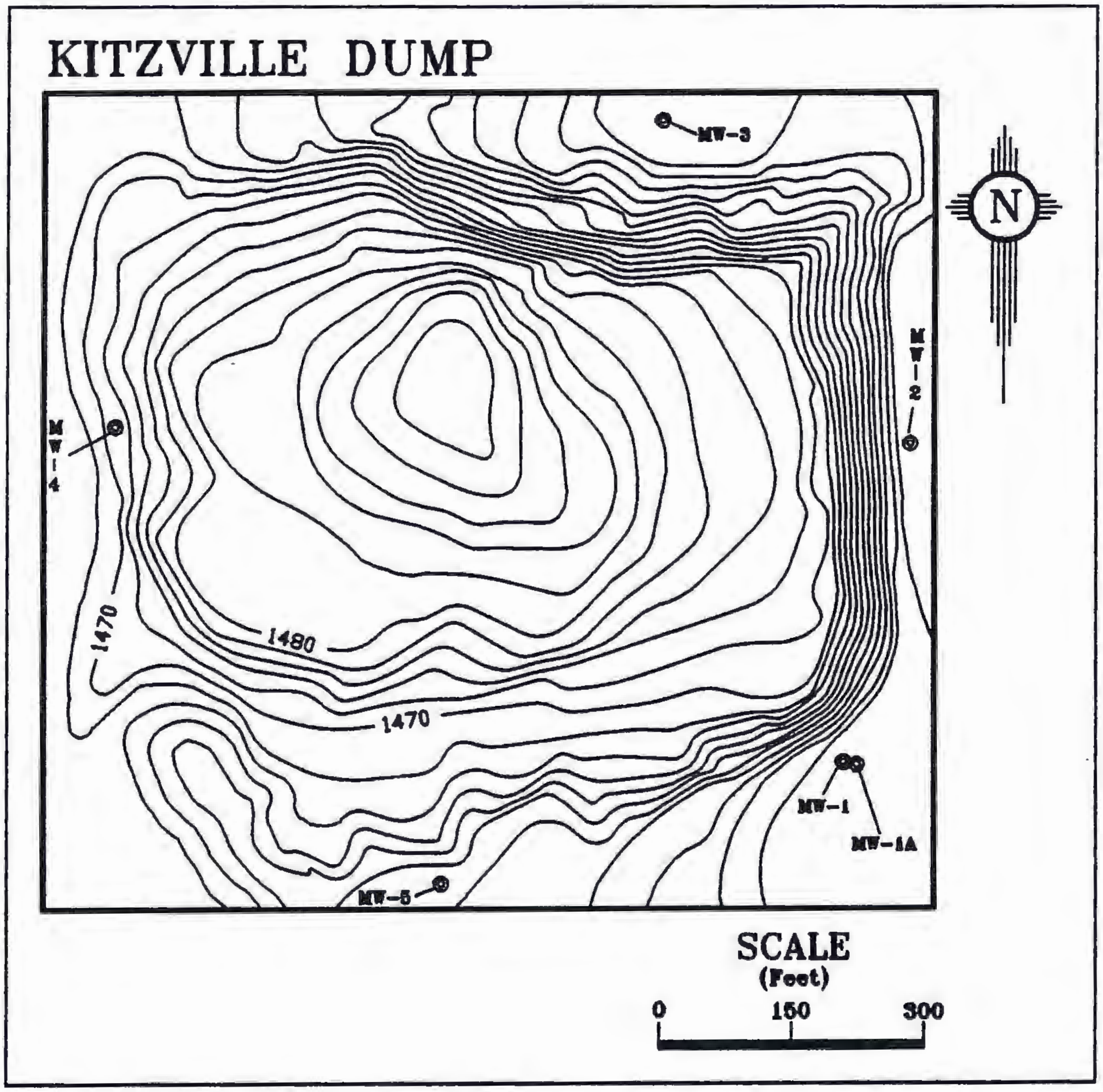

Figure 6 Monitoring well location map.

site the Guelph Permeameter (Model 2800K1) was used. The Guelph Permeameter, a constant-head device which operates on the Mariotte siphon principle, provides a relatively efficient method for determining field saturated hydraulic conductivity in the vadose zone. Hydraulic conductivity calculations are based on the rate of fall of water in the

TABLE 1

AQUIFER HYDRAULIC CONDUCTIVITY VALUES COMPUTED USING THE HVORSLEV METHOD

Well #	Hydraulic Conductivity (K) (cm/sec)
MW-1	5.56×10^{-4}
MW-2	1.31×10^{-3}
MW-3	8.02×10^{-4}
MW-4	2.95×10^{-4}
MW-5	1.44×10^{-3}

permeameter cylinder and other geometric parameters (Soilmoisture, 1987; Elrick and others, 1989). The actual calculations performed in the field for determination of hydraulic conductivity may be seen in Appendix C. Final computed results are listed and compared with lab values in Table 2.

As mentioned previously, the dump was capped with a layer of silty sand in 1981, and this fill was one of the primary materials tested with the in-situ permeameter. Both the north and south sides of the cover material were tested. On the north side the cover material was tested at a depth of 17 cm and provided a K value two orders of magnitude higher than that obtained in the laboratory. The south

TABLE 2

COMPARISON OF IN SITU AND LAB HYDRAULIC CONDUCTIVITY VALUES FOR SURFICIAL MATERIALS

Location	In-Situ K Values (cm/sec)	Lab (Braun) K Values (cm/sec)
North Cover	1.45×10^{-5} (Depth: 17 cm)	5.2×10^{-7}
	1.59×10^{-3} (Depth: 41 cm)	
South Cover	1.75×10^{-5} (Depth: 26 cm)	4.8×10^{-7}
Surficial Silty Clay	1.81×10^{-5} (Depth: 26 cm)	5.5×10^{-8}
	1.03×10^{-7} (Depth: 51 cm)	

cover was tested at a depth of 26 cm and produced a K value comparable to the north cover field value. An additional test was performed on the north cover at a depth of 41 cm and produced a value two orders of magnitude higher than the two other field values. This high value resulted from penetration of the cover material by the test hole into the solid waste below.

The comparison of lab and field hydraulic conductivity values indicate a significant disparity. The difference is most likely due to the difficulty in simulating field density of soil samples in the laboratory. The difference may also reflect the conditions of measurement, in that field tests measure primarily horizontal hydraulic conductivity, while laboratory tests typically measure mainly vertical hydraulic conductivity (U.S. EPA, 1984). This higher field hydraulic conductivity indicates that the actual recharge occurring to underlying layers may be greater than previously thought. This higher recharge translates into a greater amount of leachate being generated due to increased percolation through the solid waste.

In-situ permeameter tests were also performed on the surficial silty clay at the site (Table 2). Tests were performed at depths of 26 cm and 51 cm. The high value obtained at the shallower depth most likely was the result of organic material, such as roots, which were prevalent at this depth. At 51 cm, less of the organic material was present giving a value somewhat more comparable to that obtained in the laboratory.

GROUNDWATER FLOW VELOCITIES AND GRADIENTS

The groundwater gradient, or the change in unit head per unit distance, varies throughout the region. In the vicinity of the site this gradient ranges from as little as 0.28 percent in the northwest region of the site to as much as 0.85 percent toward the southeast region of the dump. This increased gradient is probably due in part to the irregular topography and lack of drainage in this region.

The groundwater gradient can be used to infer the average linear velocity or seepage velocity, which is the rate at which water actually moves through the pore spaces in the aquifer. The average linear velocity is computed as follows:

$$V_s = \frac{K\,dh}{n_e\,dl}$$

where

V_s is the average linear velocity (cm/sec)

K is the hydraulic conductivity (cm/sec)

n_e is the effective porosity

dh/dl is the hydraulic gradient

By using a mean hydraulic conductivity of 8.8×10^{-4} cm/sec (Table 1), an average gradient of 0.56 percent and an assumed effective porosity of 0.25 (Fetter, 1988), the average linear velocity is 2.0×10^{-5} cm/sec. This is equivalent to approximately 20 feet per year. The average linear velocities computed in this study have a probable range from 10 ft/yr to 31 ft/yr.

SECTION IV

NUMERICAL MODEL OF SITE GROUNDWATER FLOW

INTRODUCTION

Models of real hydrogeologic systems are useful in understanding why a flow system is behaving in a particular observed manner and to predict how a flow system will behave in the future. The first step in creating a groundwater flow model is to develop a conceptual model of the flow system (Prickett, 1979). The Kitzville Dump can be described as a groundwater system being contained in deposits of glacial outwash, confined by glacial till units, overlying a nearly level, impermeable bedrock surface. This conceptual model is less complex than the real system but is necessary in understanding the behavior of the flow system. Conceptual models are static; however, they describe the present condition of a system. In order to make predictions of future behavior, it is necessary to have some sort of dynamic model that is capable of manipulation (Mercer and Faust, 1981).

A mathematical model is a type of dynamic model that is simply a set of equations, which, subject to certain assumptions, describes the physical processes active in the aquifer. The derivations of equations applied to groundwater situations are based on the conservation principles dealing with mass, momentum, and energy. These principles require that the net quantity, whether it be mass, momentum or energy, leaving or entering a specified volume of aquifer during a given time interval be equal to the change in the amount of that quantity stored in the volume. The mathematical model for groundwater flow consists of a partial differential equation together with appropriate boundary and initial conditions that express conservation of mass and that describe continuous variables, such as head or concentration, over the region of interest (Pinder and Bredehoeft, 1968).

There are essentially two approaches that can be utilized to obtain a solution for the mathematical model: analytical and numerical. If certain basic assumptions may be made, such as infinite aquifer extent and constant thickness, the groundwater flow equation can be simplified and solved analytically (Mercer and Faust, 1981).

For those situations where the simplified analytical model no longer describes the physics of the situation, the partial differential equation can be approximated numerically. The continuous differential equation, defining hydraulic head everywhere in an aquifer, is replaced by a finite number of algebraic equations that define hydraulic head at specific points. The finite-difference equation for steady-state conditions with recharge, also known as the Poisson Equation (Wang and Anderson, 1982) is:

$$\frac{(h_{i-1,j}-2h_{i,j}+h_{i+1,j})}{(\Delta x)^2} + \frac{(h_{i,j-1}-2h_{i,j}+h_{i,j+1})}{(\Delta y)^2} = -\frac{R}{T}$$

where delta x and y are the distances between nodes in the x and y directions (ft)

R is the recharge (GPM/ft^2)

$h_{i,j}$ is the hydraulic head at a specific point (ft)

T is the aquifer transmissivity

(See Figure 8 for an explanation of the i,j coordinate system.)

The specific finite-difference groundwater flow model used in this study is Intersat, developed by Hydrosoft of Lake Wales, Florida. The matrix equation is solved using the iterative ADI (alternating direction implicit) technique (Hydrosoft, 1985). Iterative methods attempt solution by a process of successive approximation. They involve making an initial guess at the matrix solution, then improving this guess by some iterative process until an error criterion is attained. Therefore, in these techniques, one must be concerned with convergence, and the rate of convergence (Prickett and Lonquist, 1971). A numerical model is most appropriate for the Kitzville Dump study due to the heterogeneities of the glacial sediments and the variable recharge rate of the aquifer. Numerical models, although more difficult to apply, are not limited by many of the simplifying assumptions necessary for the analytical methods (Prickett, 1979).

The final step in modeling a groundwater flow system is to translate the mathematical results back to their physical meanings (Mercer and Faust, 1981). In addition, these results must be interpreted in terms of their agreement with the observed conditions at the study site.

The application of a groundwater model to a field problem entails effort in several areas. These include data collection, data preparation, calibration, and predictive simulation. These tasks should be considered more as a feedback approach, rather than as distinct steps (Mercer and Faust, 1981).

DATA PREPARATION

Data preparation for the groundwater flow model first involves determining the boundaries of the region to be modeled. The boundaries may be physical (impermeable or no flow, recharge, and constant head), or merely convenient (small area of a large aquifer) (Prickett, 1979). The boundary conditions used at the study site are artificial constant head boundaries oriented with the regional flow direction (northwest to southeast).

A three-layer system is modeled with the first layer being the upper confining layer of glacial till and the second layer being the shallow glacial outwash aquifer directly beneath (Figure 7). The third layer is the lower confining unit of glacial till directly beneath the shallow aquifer and directly above the deeper aquifer.

With the vertical boundaries of the aquifer determined it is necessary to subdivide the region into a grid pattern. There are two types of common grids: mesh-centered and block centered. Associated with the grids are node points that represent the position at which the solution of the unknown value (head or concentration) is obtained. In the mesh-centered grid the nodes are located on the intersection of grid lines, whereas in the block-centered grid the nodes are centered between grid lines. Typical computer grid notation for the finite difference grid is shown in Figure 8. The grid network spacing is typically smaller in those regions where the physical characteristics of the system are better known and larger in those regions where less is known (Mercer and Faust, 1981).

The Kitzville Dump site study uses a mesh-centered grid composed of 21 columns and rows with a variable grid spacing. The grid network is composed of a central region with a spacing of 100 feet between nodes. Outward from this central region the grid spacing increases non-linearly to 275 feet node spacing (Figure 9). The total side length of

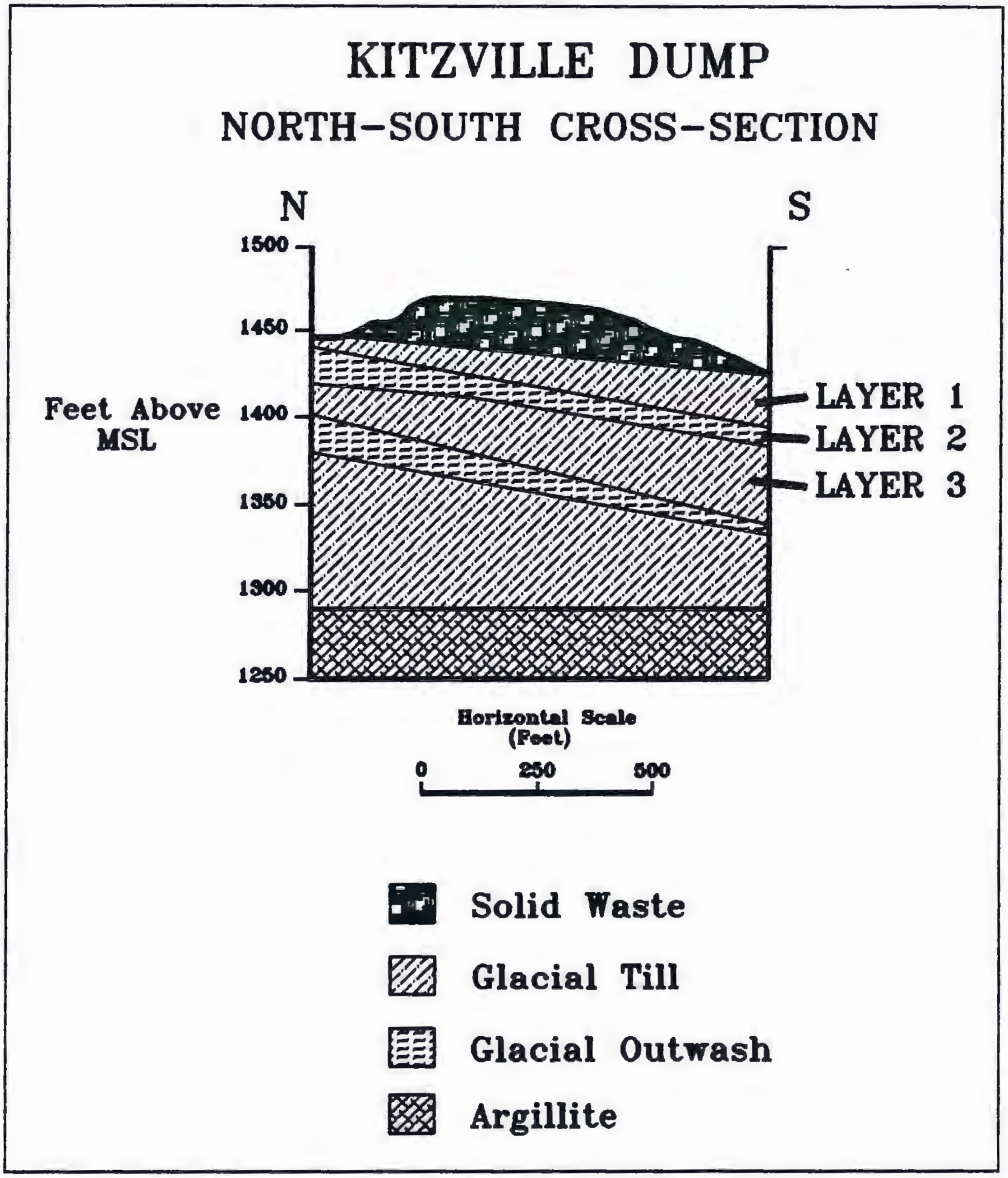

Figure 7 Cross-section of site illustrating the three modeled layers.

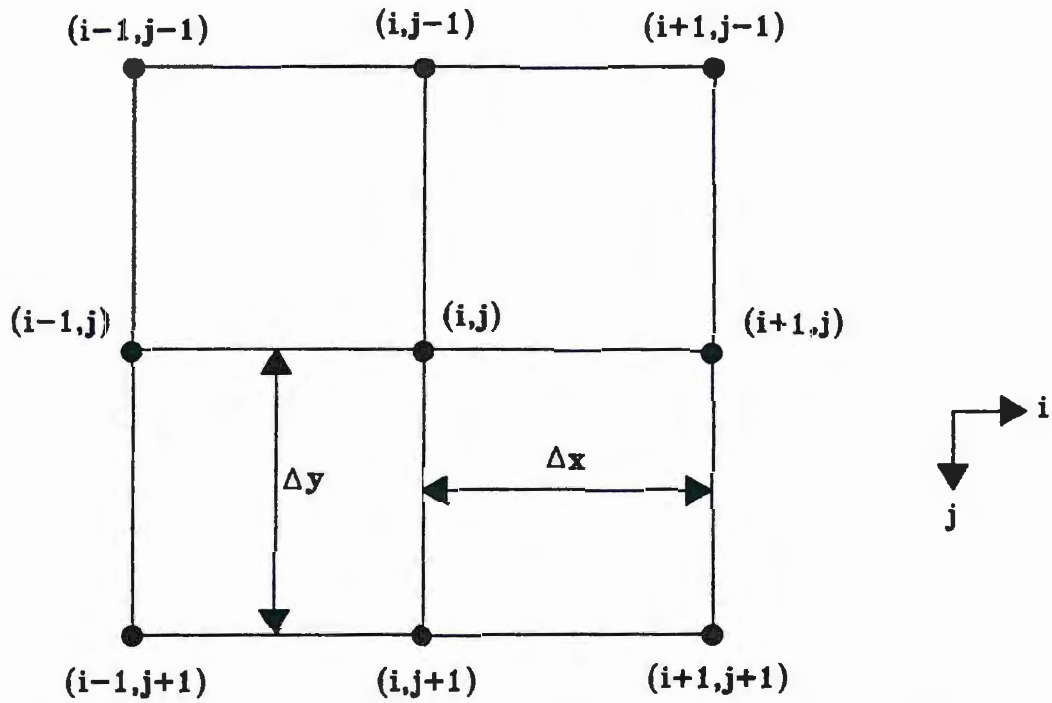

Figure 8 Finite-difference grid computer notation (Fetter, 1988).

the grid is 2750 feet and the total grid area covers 7.5 million square feet or approximately 174 acres. The grid is oriented with the general direction of regional groundwater flow.

Once the grid is designed, it is necessary to specify aquifer parameters and initial default data for the grid. Input to the model consists of entering the bottom and top elevation of each layer, hydraulic conductivity, porosity, head values, recharge, discharge and storage coefficients. Each of these default values may be changed by entering specific data at specific node points. Specific layer bottom and top elevations were obtained from site well log

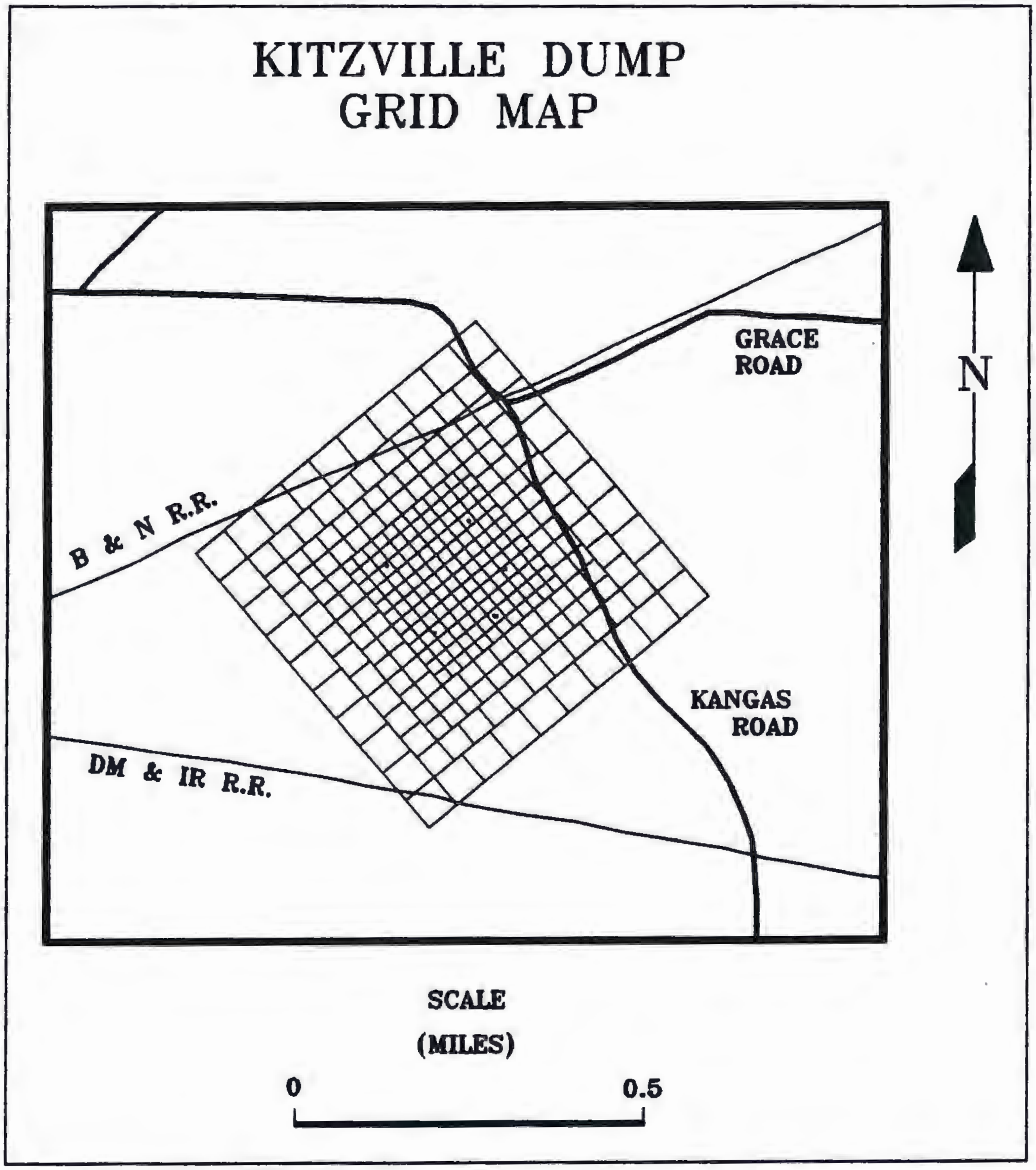

Figure 9 Grid map of study area.

data (Appendix A). Hydraulic conductivity values were obtained from aquifer tests performed at the site on monitoring wells MW-1 to 5 (Appendix B). A mean hydraulic conductivity value of 8.8×10^{-4} was utilized for the model.

Representative porosity values for glacial outwash and till units were obtained from Fetter (1988). The porosity values used were 0.25 and 0.15 respectively, for these units. The storativity value used for the confined aquifer (0.002) was a median representative value obtained from Freeze and Cherry (1979).

HYDROLOGIC EVALUATION OF LANDFILL PERFORMANCE (HELP) MODEL

In order to gain a better idea of the amount of water percolating through the solid waste at the dump site the Hydrologic Evaluation of Landfill Performance (HELP) Model was utilized. This software program, developed by Paul Schroeder of the U.S. Environmental Protection Agency, models the effects of precipitation, surface storage, run-off, infiltration, percolation, evapotranspiration, soil moisture storage and lateral drainage on a variety of land-fill designs. This model is run totally independent of the Intersat model and is used to provide greater accuracy for estimating percolation through and recharge to the various layers.

Two model configurations were used to represent the Kitzville Dump. The first model was set up to represent the dump site during its operating period (1946 to 1981) when there was no cover material. The second model was set up to represent the site after its closing when it was capped with a layer of silty sand.

In the first model, two layers were used, one to represent the solid waste and the other to represent the upper confining layer or till unit. The simulation was run for a five-year period of which the average annual totals are shown in Table 3.

TABLE 3

AVERAGE ANNUAL TOTALS FOR FIVE-YEAR HELP MODEL SIMULATION OF OPEN OR OPERATING KITZVILLE DUMP.

	INCHES	CU. FT.	PERCENT
Precipitation	31.30	913000	100
Runoff	1.57	45700	5.0
Evapo-transpiration	21.0	611000	67.0
Percolation from Layer 2	5.17	151000	16.50
Storage	3.60	105000	11.5

In the second model, an additional layer was added to represent the cover material. This three-layer model was also run for a five-year period (Table 4) with results that differed significantly from the first model.

The value of primary interest is the percolation from the bottom layer of each model (Layer 2 and 3, respectively) which shows that the addition of the cap reduced the estimated percolation into the shallow aquifer from 150,000 to

61,300 ft^3 annually.

TABLE 4

AVERAGE ANNUAL TOTALS FOR FIVE-YEAR HELP MODEL SIMULATION OF CLOSED OR CAPPED KITZVILLE DUMP.

	INCHES	CU. FT.	PERCENT
Precipitation	31.30	913000	100
Runoff	1.55	45200	4.95
Evapo-transpiration	24.2	705000	77.2
Percolation from Layer 3	2.10	61300	6.72
Storage	3.45	101000	11.1

MODEL SIMULATION

After the data are entered a solution technique is selected. The solution technique utilized with the Intersat model is the ADI or turbo method discussed previously which provides for quicker convergence than the Prickett method also available with this model (Hydrosoft, 1985).

A simulation time duration must also be selected. For this study steady-state conditions are being simulated. This requires a long simulation time, on the order of 1 x 10^{10} days (2.74 x 10^7 years). This allows sufficient time for head values to stabilize. Under conditions of steady-state flow there is no change of head with time and flow may be described by the two-dimensional partial differential equation known as the LaPlace Equation (Fetter, 1988):

$$\frac{\partial^2 h}{\partial x^2} + \frac{\partial^2 h}{\partial y^2} = 0$$

Once the simulation parameters are selected, an initial simulation run is performed to work out any "bugs" or data input errors which may keep the model from reaching a final solution. The initial simulation runs for the site model failed several times because of data input errors. These were easily corrected once found.

MODEL CALIBRATION

Initial estimates of aquifer parameters constitute the first step in a trial and error procedure known as model calibration or history matching (Mercer and Faust, 1981). The calibration procedure is used to refine initial estimates of aquifer parameters such that the computed head values agree reasonably well with the observed head values.

The number of runs required to produce a satisfactory match depends on the complexity of the flow system and length of observed history. Calibration of the groundwater flow model required approximately 30 to 40 simulation runs since varying any parameter any amount necessitated recomputation of the entire model. Hydraulic head values are dynamic or constantly changing due to varying parameters such as recharge. Recharge values varied considerably over the grid region due not only to the irregular topography, but also to the varying thickness of overlying materials. The recharge rate varied from approximately 0.003 to 0.02 GPD/ft^2 over the grid region. Figure 10 illustrates a contour map of computed hydraulic head values generated by the numerical model. These computed values agree reasonably well with the observed history of head values obtained from the site (Figure 5 and Appendix D).

During the simulation routine an output file is written containing basic grid dimensions, hydraulic head and flow velocities (Hydrosoft, 1985). This output file serves as primary input to the solute transport model Intertrans, described in Section VI.

PARAMETER UNCERTAINTY

One of the primary concerns of modeling any field problem is the quality of data available for model input. Without quality data, model output is meaningless. In other words, the reliability of modeled results increases proportionally with the reliability of input data (Prickett, 1979). In field situations there is a practical limit to the amount and quality of data that can be obtained. This practical limit can manifest itself in technical or economic

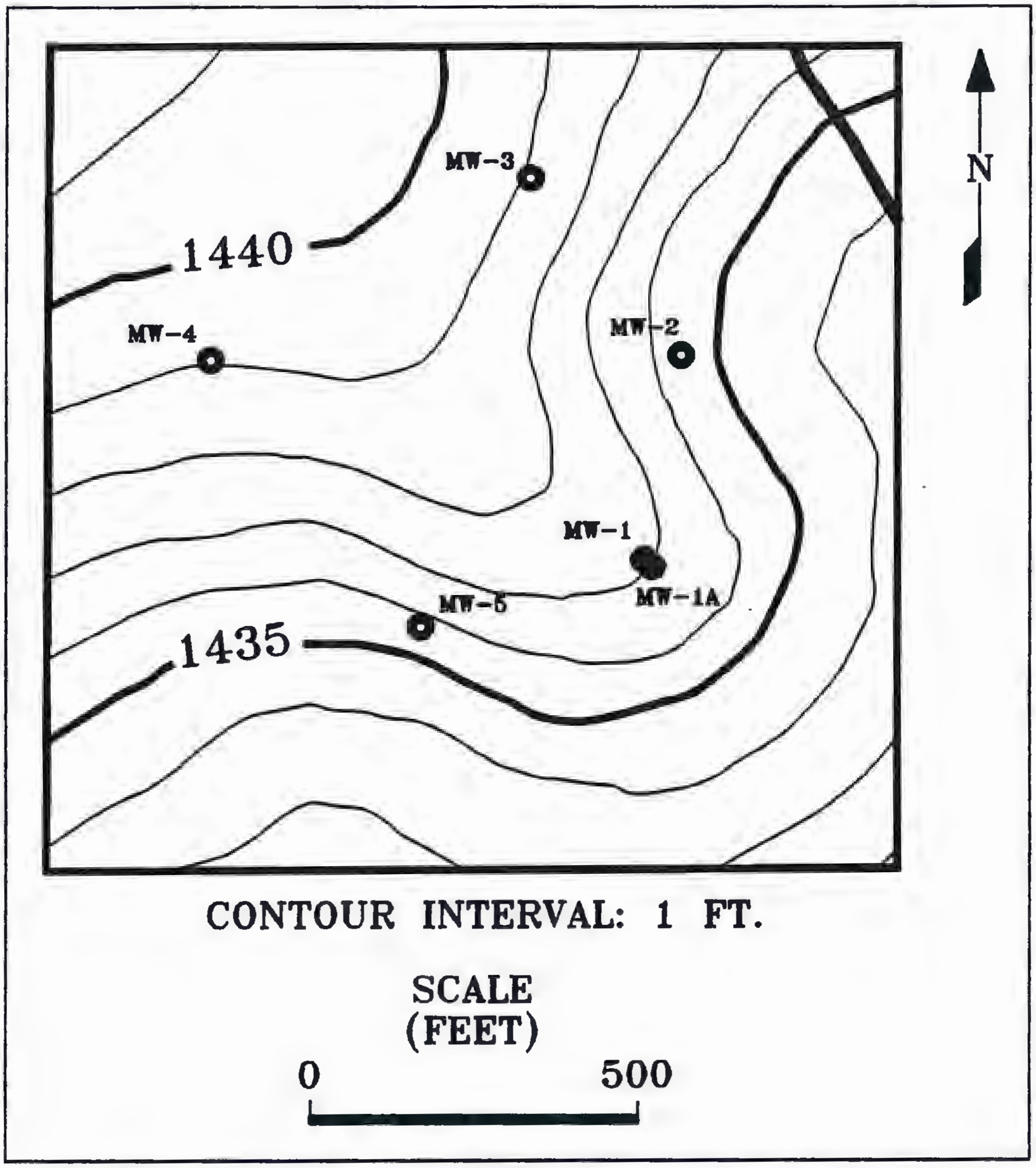

Figure 10 Contour map of hydraulic head values generated by the numerical flow model. Compare with Figure 5.

terms. For instance, it is certainly technically possible to drill test holes every few feet to determine aquifer parameters in a highly variable sequence, but it is hardly feasible in economic terms. So, in these instances one is left with making "educated estimates" of parameters based on the available data.

Parameter uncertainty can take the form of measurement and sampling errors from standardized procedures such as hydraulic conductivity tests, chemical analyses and geophysical surveys. The possibility exists for errors up to one order of magnitude or more in some parameters, depending upon the data collection method used (Mercer and Faust, 1981). In some cases, there are no reliable methods for obtaining data, such as dispersivities (Zuber, 1974).

Intrinsic parameter uncertainty results from the variability of natural properties and processes (Delhomme, 1979). One good example of this at the study site is the highly variable nature of the glacial sediments, which have changes in lithology and texture which are commonly too local or small-scale to be defined on the basis of available data.

It is apparent that increasing the quality of data is important to the reliability of models, but invariably certain parameters must be estimated. Usually these estimates can be bracketed within certain well-defined ranges.

SENSITIVITY ANALYSIS

The process whereby certain parameters are changed to learn what effect they have on computed results is termed sensitivity analysis (Mercer and Faust, 1981). Sensitivity analysis was performed on the groundwater flow model to gain a better understanding of the impact of estimated parameters on computed flow velocities and gradients. The parameters used in this analysis were hydraulic conductivity and recharge.

The calculated hydraulic conductivities at the site ranged from 3.0×10^{-4} to 1.4×10^{-3} (Table 1). The effect of increasing the hydraulic conductivity one order of magnitude had a significant effect on the flow velocities and gradients. The hydraulic flow gradient range was from 0.25 to 0.55 percent, a substantial difference. The average linear velocity increased significantly (up to 350+ ft\yr) even though there was some reduction in the median gradient level. Decreasing the hydraulic conductivity one order of magnitude had the reverse effect in that the flow gradients increased substantially (2 to 3 percent gradient), while the average linear velocity decreased to about 1.8 ft/yr.

The effect of varying recharge from the already diverse values across the model had results much as expected. Increased recharge had the effect of increasing head values, flow gradients, and flow rates. Decreased values of recharge resulted in decreased head values, flow gradients, and flow rates.

An analysis was also performed on the HELP model utilizing the runoff parameter which is based on the type of vegetative cover present at the site. This value was varied between 10 and 100 (the minimum and maximum values). The model default value of 89 for the closed-site model was based on the selection of fair grass as the vegetative cover. Increasing this value resulted in a higher runoff value and a decrease in percolation through Layer 3 (bottom layer). Decreasing this value reduced the amount of runoff, but increased the amount of evapotranspiration. The results for the pre-closing model are similar.

Since the actual vegetation on the site is fair grass, the estimated percolation values generated by the HELP model provide a very good indication of the site recharge.

Recharge and hydraulic conductivity are the two parameters which provide the primary variance within the model. From Table 1 it appears that the hydraulic conductivity at the site varies less than one order of magnitude. Therefore, recharge remains as the primary variable within the system for the calibration procedure.

SECTION V

REGIONAL AND SITE GROUNDWATER QUALITY

The quality of groundwater at the Kitzville Dump is presently being monitored, on a routine basis, under the direction of St. Louis County Health Department. Groundwater samples are collected and analyzed by Northeast Technical Services, Inc. of Virginia, Minnesota. Reports are forwarded by the county to the Minnesota Pollution Control Agency (MPCA) in St. Paul.

Chemical analyses are performed on all six monitoring wells three times per year. Groundwater samples are analyzed for ammonia, nitrate, chloride, sulfate, dissolved iron and chemical oxygen demand (Appendix E). Field water temperature and specific conductance are also measured along with field and lab pH (Appendix E).

Two initial rounds of sampling were performed in November 1982 and January 1983 for organics and metal analysis (Appendix E). The results of these analyses showed no significantly raised levels and no further tests have been done for these compounds.

WATER QUALITY STANDARDS

Water quality standards are regulations that place specific limitations on the quality of water that is utilized for a particular purpose (U.S. EPA, 1976). The U.S. Environmental Protection Agency (EPA) has established drinking water standards under provisions of Public Law 93-523, the Safe Drinking Water Act, and its amendments (U.S. EPA, 1984). One of the primary objectives of the Safe Drinking Water Act is to determine maximum contaminant level goals (MCLG), maximum contaminant levels (MCL), and secondary maximum contaminant levels for materials that may be found in drinking water. Maximum contaminant level goals are essentially health goals set at a level to prevent possible adverse effects with a reasonable safety margin. Maximum contaminant levels are enforceable standards that set the highest allowable concentration of a solute in a public water supply system. Secondary maximum contaminant levels (SMCL) are standards for substances such as iron, chloride, sulfate, and total dissolved solids (TDS), which can affect the aesthetic quality of water by imparting taste and odor and staining fixtures (Fetter, 1988).

The state of Minnesota has set Recommended Allowable Limits (RAL) for organics and metals in drinking water. The MCLG, MCL, SMCL, and RAL standards for the compounds examined in this study are listed in Table 5.

On a regional basis the quality of groundwater in this

TABLE 5

WATER QUALITY STANDARDS FOR CHEMICAL SPECIES EXAMINED IN THIS STUDY.

Contaminant	MCLG	MCL	SMCL	RAL
Benzene	0.00	0.003		0.007
Cadmium	0.005	0.01		0.005
Chloride			250	
Lead	0.02	0.05		0.02
Nitrate	10	10		
Sulfate			250	
Toluene		0.5		2.42

* All values in mg/l

area is ranked among the best in the state. Chemical analyses of chloride throughout the region in glacial drift deposits indicate a range from less than one to approximately 20 mg/l. This range of values is comparable to the values for chloride sampled at wells MW-3-5 and MW-1A. Regional field pH values range from about 6.5 to 8.3. Values of dissolved sulfate range from one to 80 mg/l (Lindholm and others, 1979).

LEACHATE CHARACTERISTICS

Leachate, which is water containing a high amount of dissolved solids derived from percolation through solid waste, is the primary agent of groundwater contamination at dump sites. When leachate from a landfill mixes with groundwater, it forms a plume that spreads in the direction of groundwater flow. Retardation and hydrodynamic dispersion will tend to decrease the concentration as the water proceeds away from the source. Retardation is an expression for many processes (complexation, ion exchange, etc.) which tend to remove solutes from groundwater, causing the solute front to advance more slowly than the rate of advecting groundwater. Hydrodynamic dispersion is essentially the process whereby contaminated groundwater is diluted with uncontaminated groundwater (Freeze and Cherry, 1979). These processes will be discussed further in the next section.

Landfill (dump) leachates can contain very high concentrations of both inorganic and organic compounds. Table 6 lists typical concentration ranges of leachates for the chemical species examined in this study.

The presence of a contaminant plume is indicated at the Kitzville Dump by the raised levels of chloride, sulfate, and specific conductance at monitoring wells MW-1 and MW-2 (Figure 11 and Appendix E).

These wells are down gradient from the dump site, indicating that the general movement of the contaminant plume is toward the southeast, in the direction of groundwater flow. Figure 11 represents a generalized concentration contour map of the high chloride levels at the study site (Appendix E). Chloride is a conservative solute, which means it does not react with other materials in the aquifer and essentially

TABLE 6

TYPICAL LEACHATE COMPOSITION VALUES, OF EXAMINED CONTAMINANTS, FROM ANALYSIS OF MUNICIPAL SOLID WASTE SITES IN WISCONSIN (MODIFIED FROM FETTER, 1988).

Parameter	Typical Range	Number of Analyses
Chloride	180-2651	303
Sulfate	8.4-500	9
Lead	ND-1.11	142
Cadmium	ND-0.07	158
Nitrate-nitrogen	ND-1.4	88
Ammonia-nitrogen	26-557	263
Specific Conductance	2840-15485	1167
COD	1120-50450	467
pH	5.4-7.2	1900

* All concentrations in mg/l except pH (std. units) and specific conductance (umhos/cm). ND indicates no data.

moves at the same rate as the groundwater (Drever, 1988). This makes chloride an excellent chemical species with which to examine the furthest extent of the contaminant plume at the site. Other chemical species will advance at some rate less than that of chloride owing to retardation processes mentioned previously.

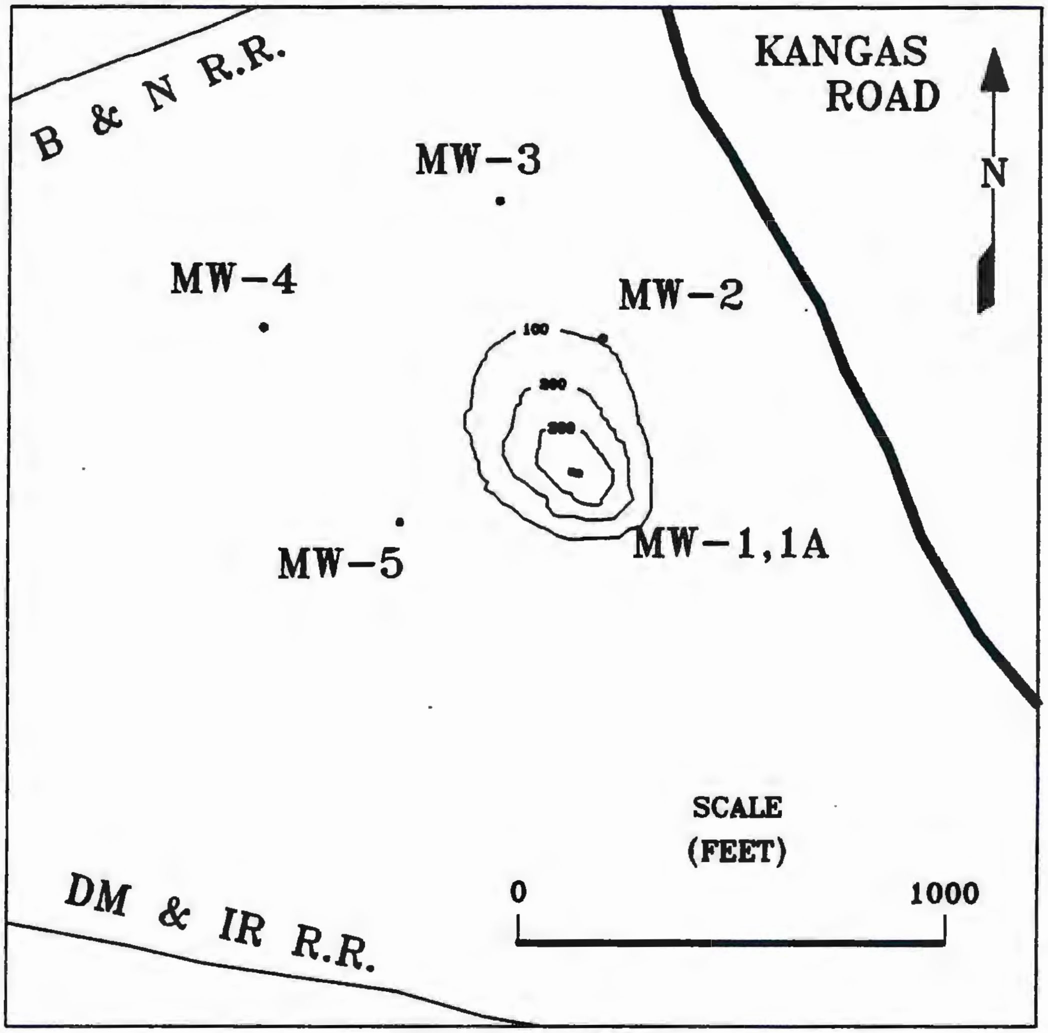

Figure 11 Generalized contour map of present chloride values in mg/l. Contour interval is 100 mg/l.

SECTION VI

NUMERICAL MODEL OF SITE CONTAMINANT TRANSPORT

INTRODUCTION

The determination of the present extent and direction of movement of the leachate plume at the Kitzville Dump is important not only from a research standpoint but also from an environmental one. The possibilities of future industrial and/or residential development in the vicinity of the Kitzville Dump reinforce the importance of gaining as much insight into this problem as feasible.

The Intertrans particle transport model utilized in this study is the companion program to the Intersat model described in Section IV. Groundwater flow is a process that can be modeled without consideration of solute transport. Solute transport modeling, however, requires either simultaneous solution with or results (e.g., velocities) from a groundwater flow model because the movement of solutes is controlled partially by groundwater movement (Mercer and Faust, 1981).

Advective transport is effectively defined by Intersat, while Intertrans simulates three-dimensional hydrodynamic dispersion (Hydrosoft, 1985). As mentioned in Section IV an output file is written by the groundwater flow model containing the advective parameters necessary for the solute transport model.

Included implicitly within the solute transport model are five discretizations. These are the X, Y, Z, and Time (T) coordinates, and contamination mass (M). The groundwater flow model determines Time, X, Y, and Z discretizations. The fifth discretization, contamination mass, is defined by the solute transport model and is represented as particles (Hydrosoft, 1985).

Solutes are transported by two primary methods: diffusion and advection. The process of diffusion occurs when chemical species dissolved in water move from areas of higher concentration to areas of lower concentration. Advection, on the other hand, is the process by which solutes are carried by moving groundwater (Freeze and Cherry, 1979).

When contaminated fluid flows through a porous material, it mixes with non-contaminated water, causing dilution of the contaminant. This process is known as dispersion. The mixing that occurs in the direction of fluid flow is called longitudinal dispersion. Mixing that occurs perpendicular to the pathway of fluid flow is lateral dispersion (Drever, 1988). Dispersion in the solute transport model is considered to be a random process bounded by a uniform distribution with standard deviation:

$$\sigma = \sqrt{2D_L Vt}$$

where,

D_L is the dispersivity

V is the velocity (cm/sec)

t is time (sec)

For the transport model, additional partial differential equations with appropriate boundary and initial conditions are required to express conservation of mass for the chemical species considered (Hydrosoft, 1985). The hydrodynamic dispersion coefficient (D), which includes both mechanical mixing and diffusion, is defined by:

$$D = D_L V_s + D^*$$

where,

D is the hydrodynamic dispersion coefficient

D_L is the dispersivity

V_s is the average linear groundwater velocity (cm/sec)

D^* is the molecular diffusion coefficient

The one-dimensional equation for hydrodynamic dispersion and advection is (Fetter, 1988):

$$D_L \frac{\partial^2 C}{\partial x^2} - V_s \frac{\partial C}{\partial x} = \frac{\partial C}{\partial t}$$

where,

D_L is the longitudinal dispersion coefficient

V_s is the average linear groundwater velocity (cm/sec)

C is the solute concentration (mg/l)

t is the time since solute invasion (sec)

Presently, chemical and physical processes that restrict the movement of a solutes in groundwater are represented in the dispersion-advection equation by a retardation factor. These processes (ion exchange, surface reaction, etc.) effectively define a different solute front for a wide range of chemical species. The expression for retardation is (Fetter, 1988):

$$R = \frac{V_s}{V_c} = 1 + \frac{\rho_b}{n} \times K_d$$

where,

R is the retardation factor

V_s is the average linear groundwater velocity (cm/sec)

V_c is the solute front velocity where the solute concentration is one-half of the original value (cm/sec)

P_b is the bulk density of the soil (g/cm^3)

n is the porosity of the soil

K_d is the distribution coefficient

For chloride the multiplication factor on the right side of the equation is very small, essentially giving a retardation factor of one. This means that chloride essentially moves at the same rate as the flowing groundwater, defining the furthest solute front from the source.

PRELIMINARY GEOPHYSICAL INVESTIGATION

ELECTRICAL RESISTIVITY SURVEY. The Bison Earth Resistivity meter was utilized at the Kitzville Dump in an attempt to delineate the extent of the contaminant plume. The basic principle underlying this survey was that resistivity of earth materials would be less in regions of greater ion concentration and greater in regions of less ion concentration. The presence of leachate would tend to increase the ionic concentration (Drever, 1988). Figure 12 shows a contour map of electrical resistivity values obtained from the site.

Apparent resistivity values collected to the south of the site decrease steadily toward the east (toward MW-1) and increase significantly in the north, west, and southwest

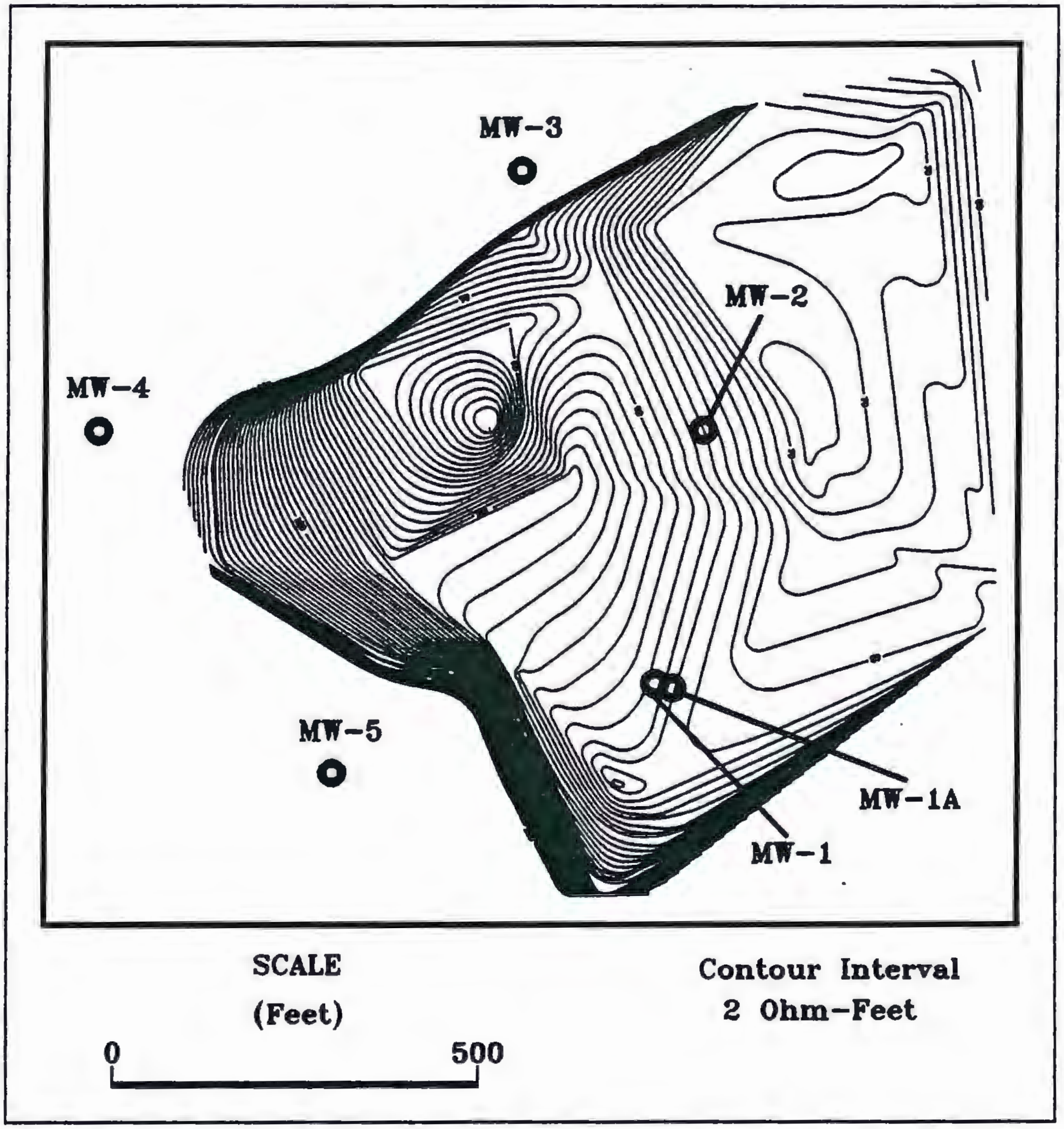

Figure 12 Contour map of apparent resistivity values from the Kitzville Dump.

regions of the site. Much of the area to the east and southeast of the site is comprised of swamp and is not suitable for testing.

SEISMIC SURVEY. The Bison Model 1500 Seismograph was utilized at the site to better delineate the depth and extent of the solid waste along the north and west margins. The seismic refraction process provided some general idea of the extent and depth of these margins but was of little or no benefit in some areas most likely due to the extreme heterogeneity of the solid waste in these regions.

DATA PREPARATION

The first requirement for the solute transport model is the reading of the groundwater flow model file which contains information such as hydraulic heads, layer thicknesses and model dimensions. Basic transport parameters are then entered for each modeled layer.

Longitudinal and transverse dispersivity values of 50 feet and 5 feet, respectively, were selected (Freeze and Cherry, 1979; Palmer and Johnson, 1989).

The gradient recomputation interval is the spatial increment which a particle is allowed to move before average linear velocity computations are repeated. This value is typically less than or equal to one-half the smallest node spacing. A value of 50 feet was utilized for this simulation.

Another important parameter, which is one of the primary variables in the calibration process, is the mass per particle. This parameter, expressed in pounds, together with the source load determines the number of discrete particles utilized to represent the distribution of the concentration of a chemical in the groundwater.

Retardation is also entered at this point in the program. For initial calibration a retardation factor of one was entered to represent chloride.

Following entry of the basic transport parameters, a rectangular cube particle source must be specified to represent the contaminant source. Two options are available at this point: a slug source or continuous source. A slug source is simply a one-time injection of a contaminant into a system. An example would be a petroleum truck or pipeline spill. A continuous source is a constant injection source, such as at the study site where there is an on-going process of percolation through the solid waste. A rectangular source area was defined approximately coincident with the margins of the solid waste. The next step is to define a source strength in pounds per day. This parameter was one of the primary values varied during calibration and will be more fully explained when that procedure is discussed.

CONTAMINANT SELECTION CRITERIA

The basis for selecting the chemical species to be examined in this study was the availability of analytical results and the characteristics of the particular species. The predominant chemical species used to calibrate the solute transport model was chloride. This solute was selected because of its conservative nature making it ideal for the calibration procedure. Two organic species were selected, benzene and toluene, because they exceeded the minimum detection limits in the 1983 analysis. Lead and Cadmium were also selected because these particular species have become of particular interest in recent years and have slightly elevated values at the study site (Appendix E).

MODEL SIMULATION

The process of running a solute transport model simulation begins with specifying a simulation time period. A two-step procedure was used at this point to represent the history of conditions found at the study site. Since the site was opened in approximately 1946 and closed and capped in 1981, there existed two primary sets of conditions. The first set of conditions was an open (uncapped) dump for a period of some 34 years. The second set of conditions was a closed (capped) dump from 1981 to present. This translates into an initial simulation period of 34 years with a higher leachate generation rate than the second simulation period of nine years when the site was capped.

MODEL CALIBRATION

As mentioned previously chloride was the chemical species used to calibrate the solute transport model. Since no chemical analyses were performed at the site prior to November of 1982 these values were used for calibration of the first simulation phase (Appendix E). The values of chloride at MW-1 and MW-2 from this first round of sampling are approximately double the values found in the following years. This could be explained by the significantly higher amount of leachate generated just prior to capping. The HELP model values of percolation or leachate generation show that there is approximately a 60 percent decrease after the site was capped (Tables 3 and 4).

An initial starting value of 10 lbs./day was specified as the contaminant source strength, but resulted in concentration levels much lower than observed values. This value and the mass per particle were varied until the concentration levels were comparable with observed values. The final mass per particle value was 200 lbs. and the final contaminant source strength was 40 lbs./day. A chloride concentration contour map is shown for this first simulation phase in Figure 13.

The second simulation phase involved reducing the contaminant source strength to correspond with the reduction in leachate generation. The initial simulation run for this phase utilized a contaminant source strength approximately 40 percent of that used in the first simulation phase. This is the amount of leachate reduction that resulted from the addition of the capping material. The results for this initial run resulted in concentration levels slightly lower than observed values. The source strength was then in-

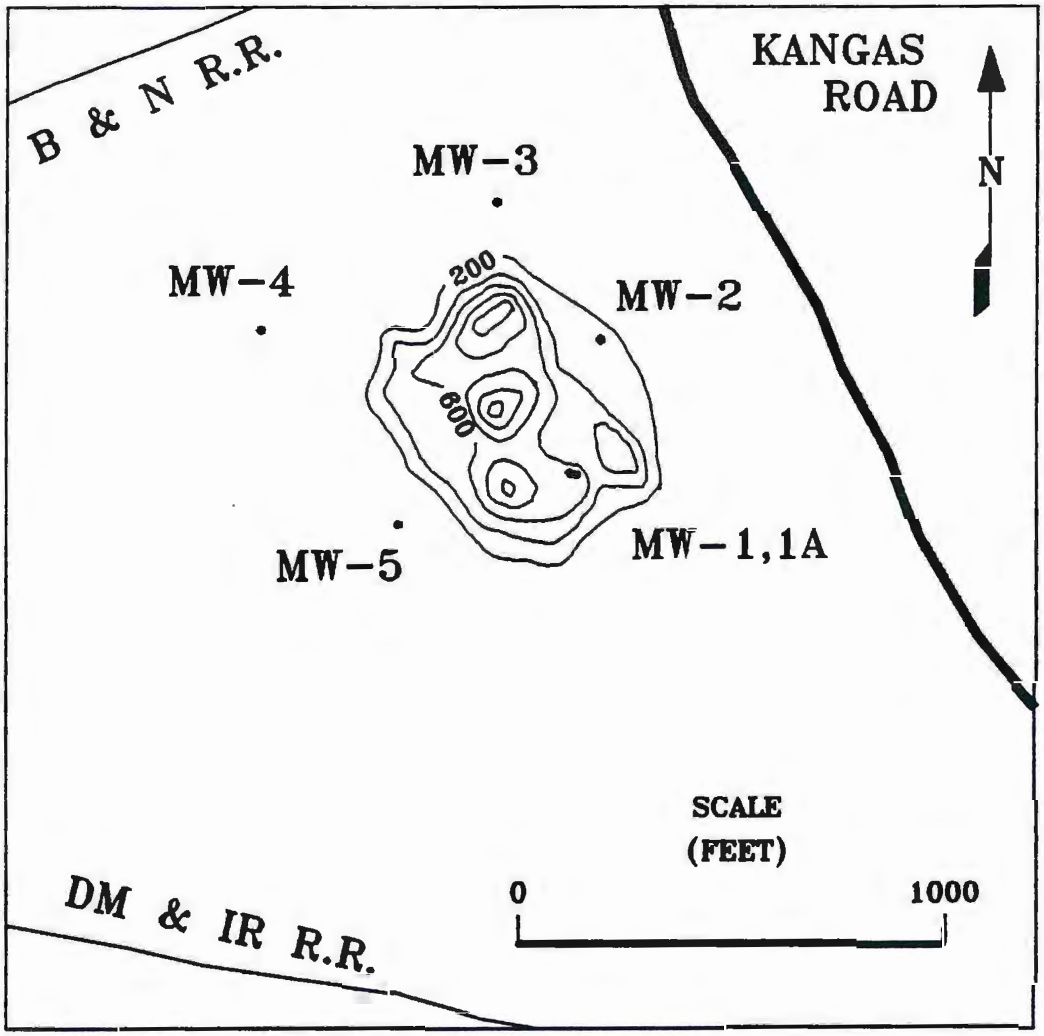

Figure 13 Solute transport model generated chloride concentration contour map of the Kitzville Dump prior to capping (1980). The contour interval is 200 mg/l.

creased slightly until the generated values compared with the post-closing observed values. The final contaminant source strength in this second simulation phase was 19 lbs./day. This represents a decrease of about 53 percent from the first simulation phase. A chloride concentration contour map showing the total 43 year simulation (Phase 1

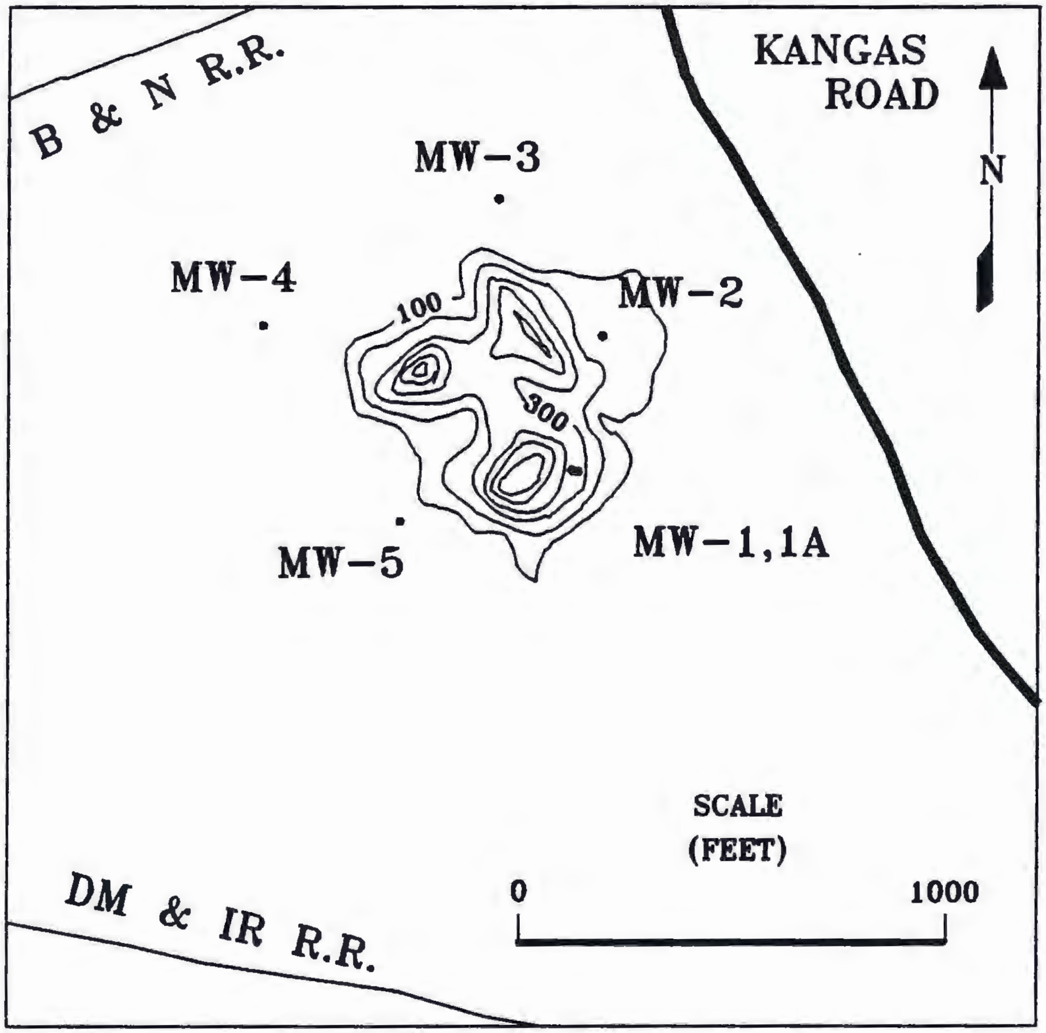

Figure 14 Solute transport model generated chloride concentration contour map of the Kitzville Dump at present (1990). The contour interval is 100 mg/l.

and 2) is illustrated in Figure 14.

SENSITIVITY ANALYSIS

A sensitivity analysis was performed utilizing the mass per particle, source strength, longitudinal and transverse dispersivity. The effect of varying the mass per particle for a given daily source load is to change the number of particles generated by the model. Varying of the source strength, as expected, would cause a corresponding increase or decrease in the concentration levels throughout the system.

Longitudinal and transverse dispersivity were varied in relation to one another to determine the effects on the plume. The values were varied within the typical field range of 33 to 328 feet (Anderson, 1979). The effect of increasing the longitudinal dispersivity with respect to transverse dispersivity was a lengthening of the plume with a corresponding thinning in the transverse direction. By increasing the transverse dispersivity with respect to the longitudinal dispersivity, the plume became less cigar-shaped and took on a greater roundness. Based on this analysis, it may be concluded that the dispersivity values used in the actual simulation are representative of the type of glacial sediments at the site.

ORGANICS

As mentioned previously, groundwater samples collected in September 1983 were analyzed for various organics (Appendix E). Two of the species analyzed; benzene and toluene, exceeded detection limits at MW-1. These chemical species were analyzed in the calibrated model to determine the extent of their respective solute fronts.

The retardation factors for both of these chemical species were determined by computing a distribution coefficient (K_d) from the organic carbon-water partition coefficient (K_{oc}) and the weight fraction of organic carbon (f_{oc}) in the aquifer. The K_{oc} value for both species were obtained from standard tables of organics (Roy and Griffin, 1985). An average f_{oc} value of 0.025 for glacial outwash aquifers was obtained from Schwartz and Smith (1987). The product of these two values is the distribution coefficient for the particular chemical species.

This K_d value, along with the bulk density and porosity of the aquifer, is then used to compute a retardation factor for each chemical species. This retardation factor is used by the solute transport model in calculating the migration of the chemical species. For both chemical species the two-phase simulation routine was used.

<u>BENZENE</u>. The K_{oc} value for benzene, which is 97, is used in the retardation equation on page 53 to arrive at a retardation value (R) of 17. An estimated initial benzene source concentration of 18 ug/l was used based on averages at other regional landfills (RREM, 1989; Evavold, 1990). Figure 15 shows a concentration contour map of benzene values calculated by the solute transport model. This figure shows that the amount of migration of the benzene plume is considerably less than that of the chloride plume for the same time period. As expected, the migration of

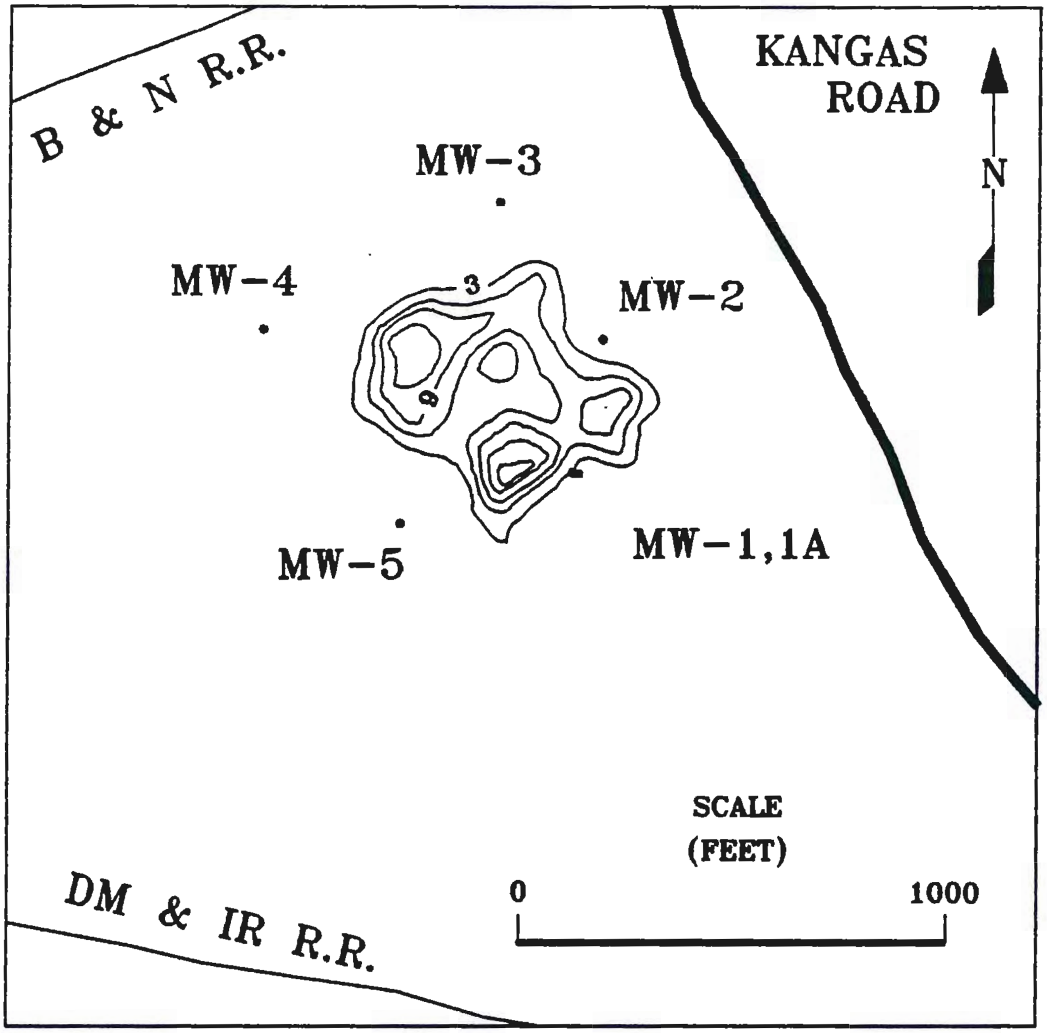

Figure 15 Concentration contour map of benzene at the Kitzville Dump (1990). The contour interval is 3 ug/l.

this plume is also in the general direction of groundwater flow.

<u>TOLUENE</u>. The K_{oc} value for toluene, which is 242, results in a retardation factor (R) of 44. An initial toluene source strength of 23 ug/l was used (RREM, 1989; Evavold, 1990). Figure 16 shows the concentration contour

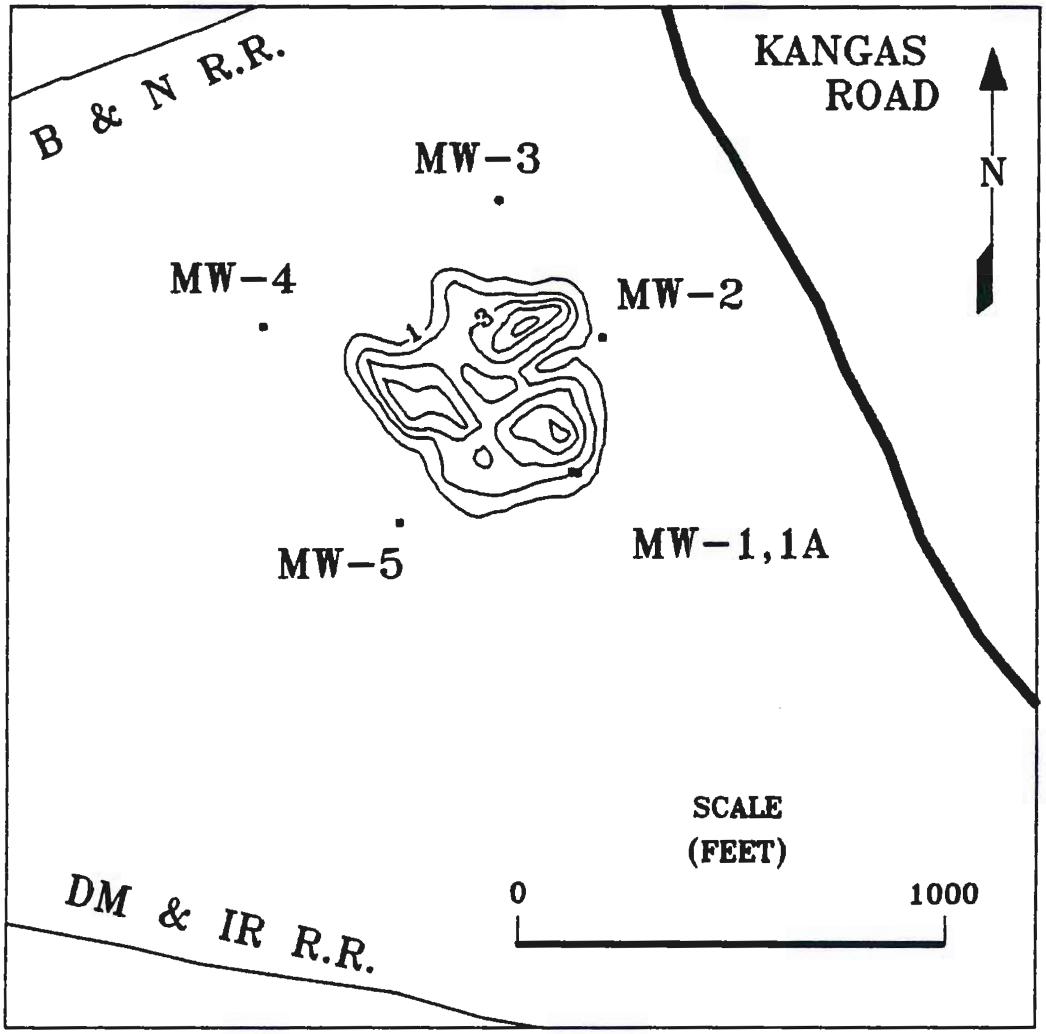

Figure 16 Concentration contour map of toluene at the Kitzville Dump (1990). The contour interval is 1 ug/l.

map of toluene for the simulation phase up to 1990.

METALS

Two metals, cadmium and lead, exceeded detection limits in the November 1982 analysis at MW-2 (Appendix E). This was the only well sampled during this round. The process for model calibration of these two species was slightly different from the organics. An estimated source concentration, based on other regional landfill data is used to provide a starting point for the calibration procedure. Retardation of metals is not associated with the organic carbon content of the aquifer media; therefore the distribution coefficient for metals cannot be determined in the same manner used for organics. For this study, the retardation factor for metals was optimized by calibrating to observed concentrations.

LEAD. The initial source concentration used for lead was 8 ug/l (RREM, 1989; Evavold, 1990; Fetter, 1988). A final retardation factor of 186 was determined for this species. Figure 17 shows the concentration contour map of lead for the simulation phase up to 1990.

CADMIUM. An initial source strength concentration of 4 ug/l was used for cadmium (RREM, 1989; Evavold, 1990; Fetter, 1988). A final retardation factor of 248 was determined for this particular chemical species. Figure 18 shows the concentration contour map for cadmium for the simulation phase up to 1990.

PREDICTIVE SIMULATION

One of the primary uses of the solute transport model is the prediction of solute transport over time. The model can be altered to reflect actual environmental changes or

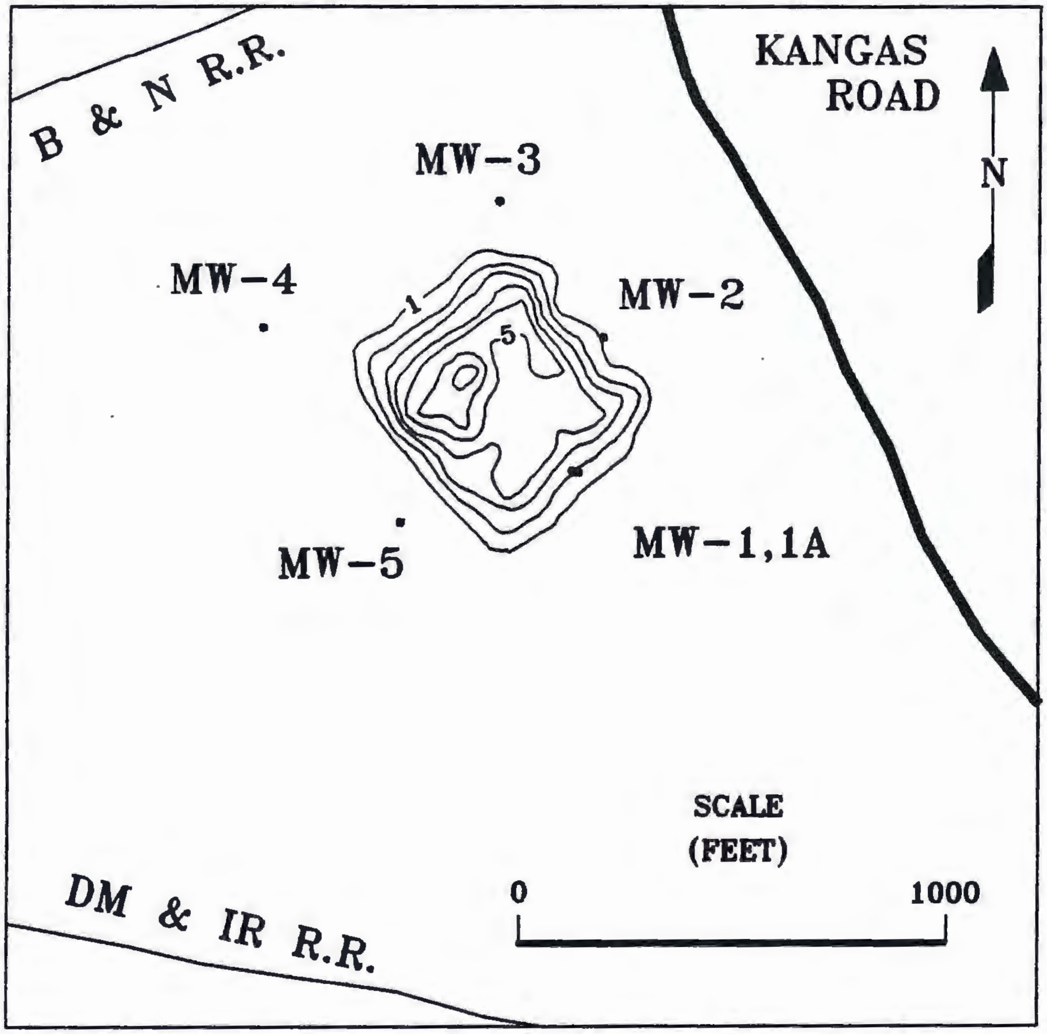

Figure 17 Concentration contour map of lead at the Kitzville Dump (1990). The contour interval is 1 ug/l.

even proposed environmental changes. This ability to gauge the effect of proposed changes provides a feedback mechanism for the planning stages of major projects.

Once the solute transport model was calibrated to present conditions at the site, an additional simulation of 10 years was run for chloride. Further simulation beyond

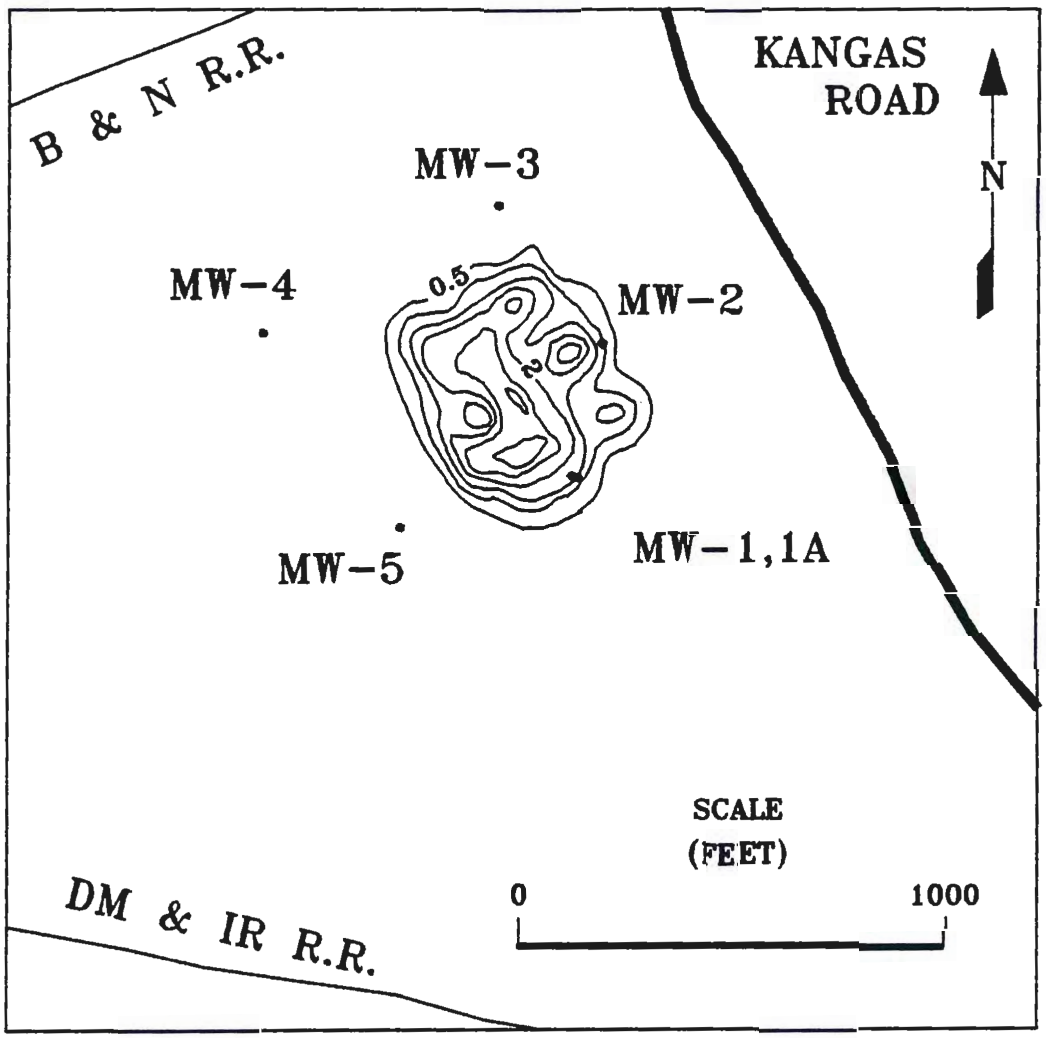

Figure 18 Concentration contour map of cadmium for the Kitzville Dump (1990). The contour interval is 0.5 ug/l.

this point was not possible for chloride due to the model limitation of 3000 particles. The mass per particle would have to be increased to simulate for longer time periods.

Figure 19 is a concentration contour map representing the predicted values of chloride at the site in the year 2000. The map clearly shows that the movement of the plume

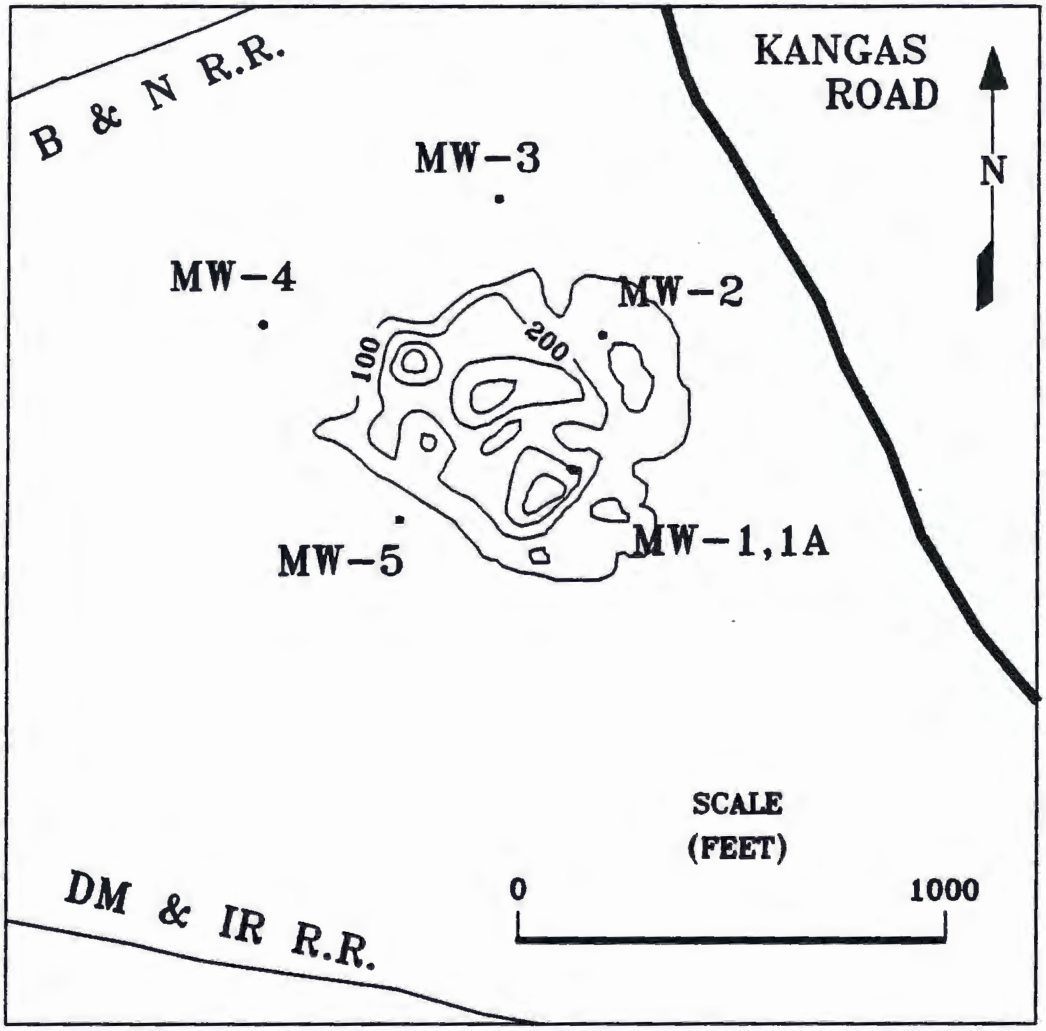

Figure 19 Concentration contour map of solute transport model estimated chloride values in the year 2000. The contour interval is 100 mg/l.

is to the east-southeast. This corresponds with the general direction of groundwater flow at the site. This predictive simulation assumes that present conditions at the site continue throughout the simulation period.

For each of the organics and metal species analyzed an additional 20 year simulation was run under the present

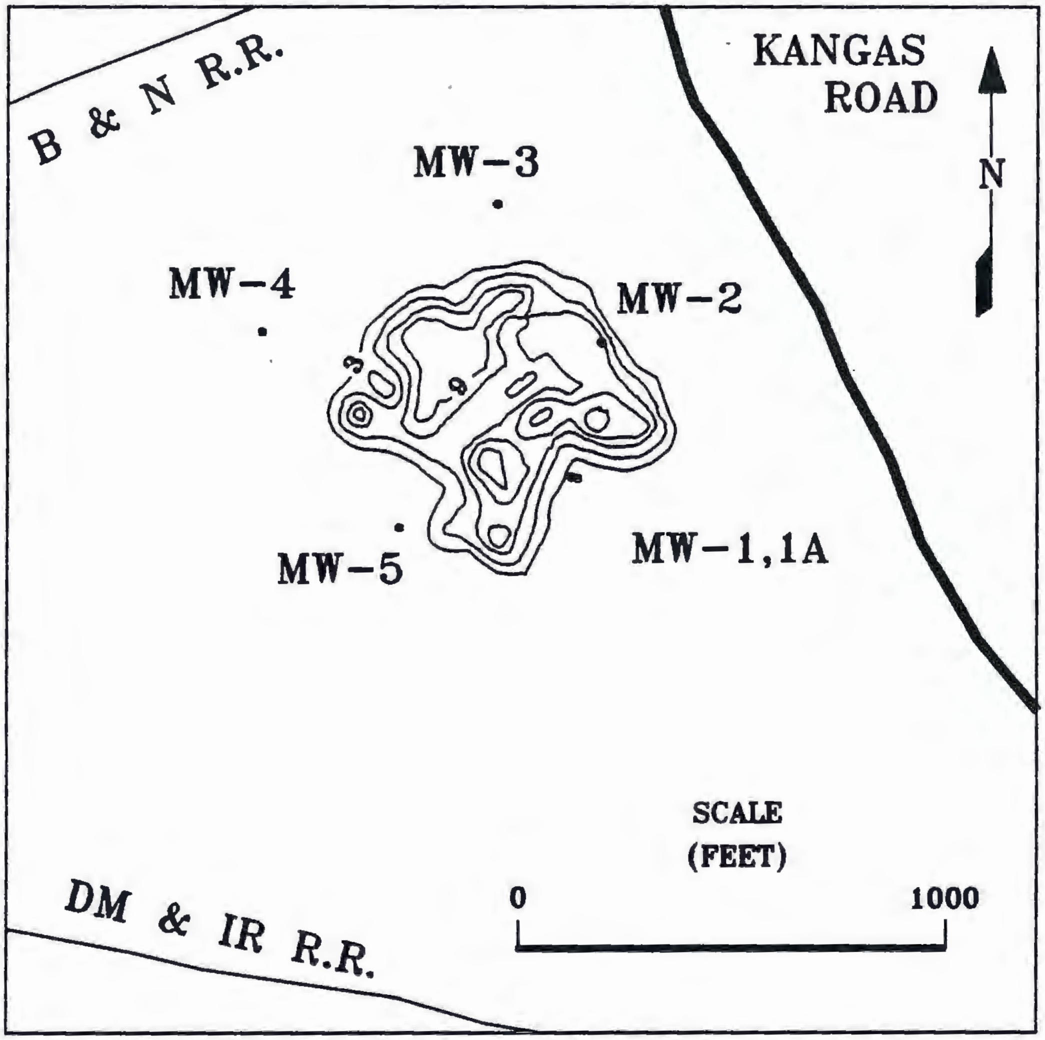

Figure 20 Concentration contour map of predicted benzene values at the Kitzville Dump (2010). The contour interval is 3 ug/l.

modeled conditions. The organics (Figures 20 and 21) both show plume migration in the general direction of groundwater flow with no substantial increase in concentration levels. Lead and Cadmium (Figures 22 and 23) both show very little migration over this simulation period due to their relatively high retardation factors.

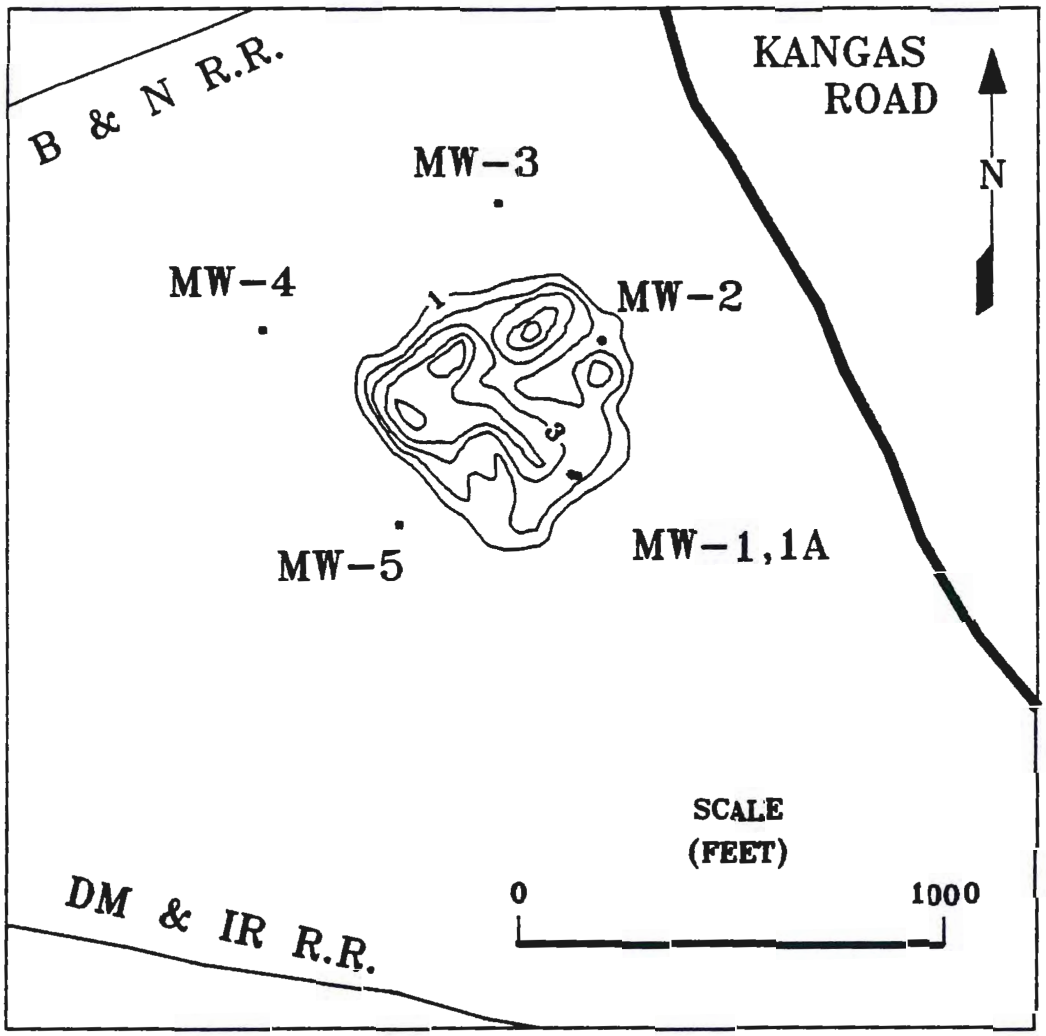

Figure 21 Concentration contour map of predicted toluene values at the Kitzville Dump (2010). The contour interval is 1 ug/l.

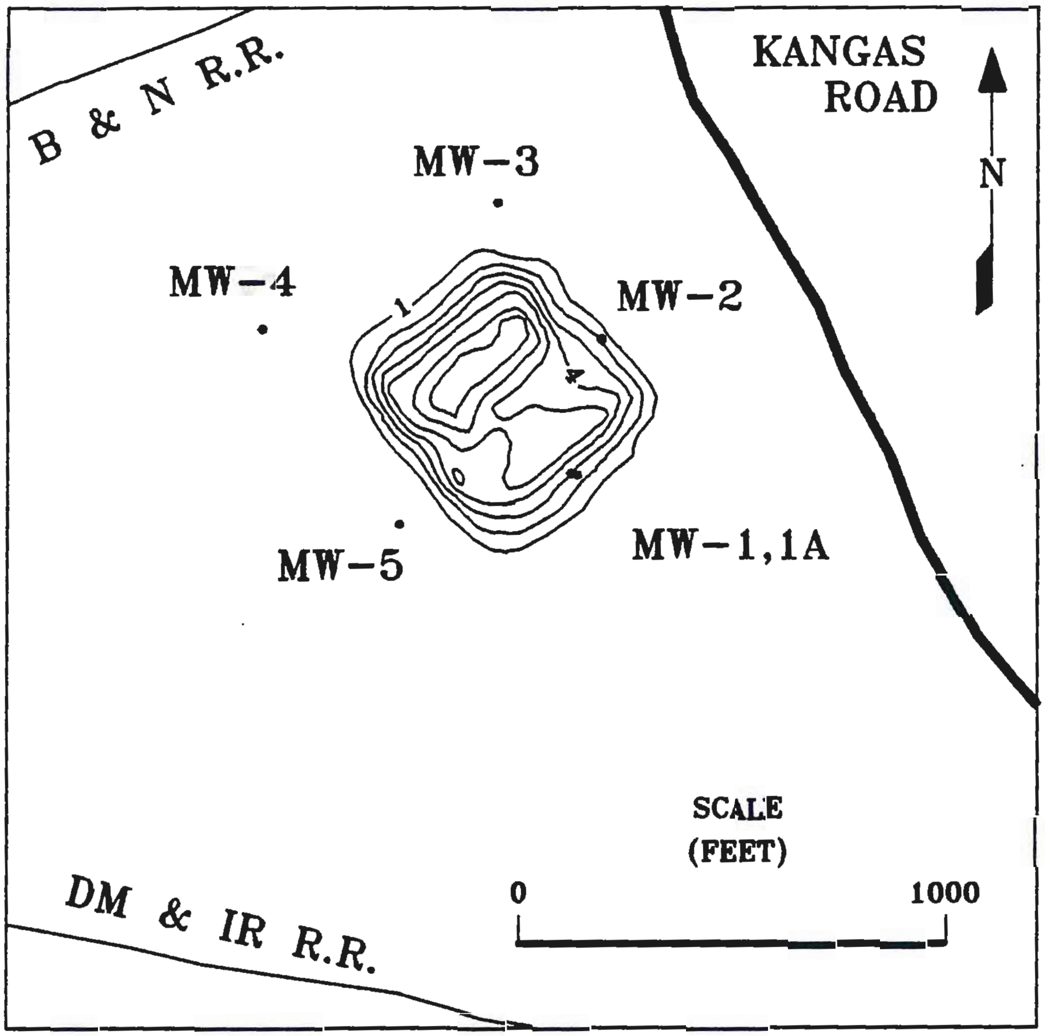

Figure 22 Concentration contour map of predicted lead values at the Kitzville Dump (2010). The contour interval is 1 ug/l.

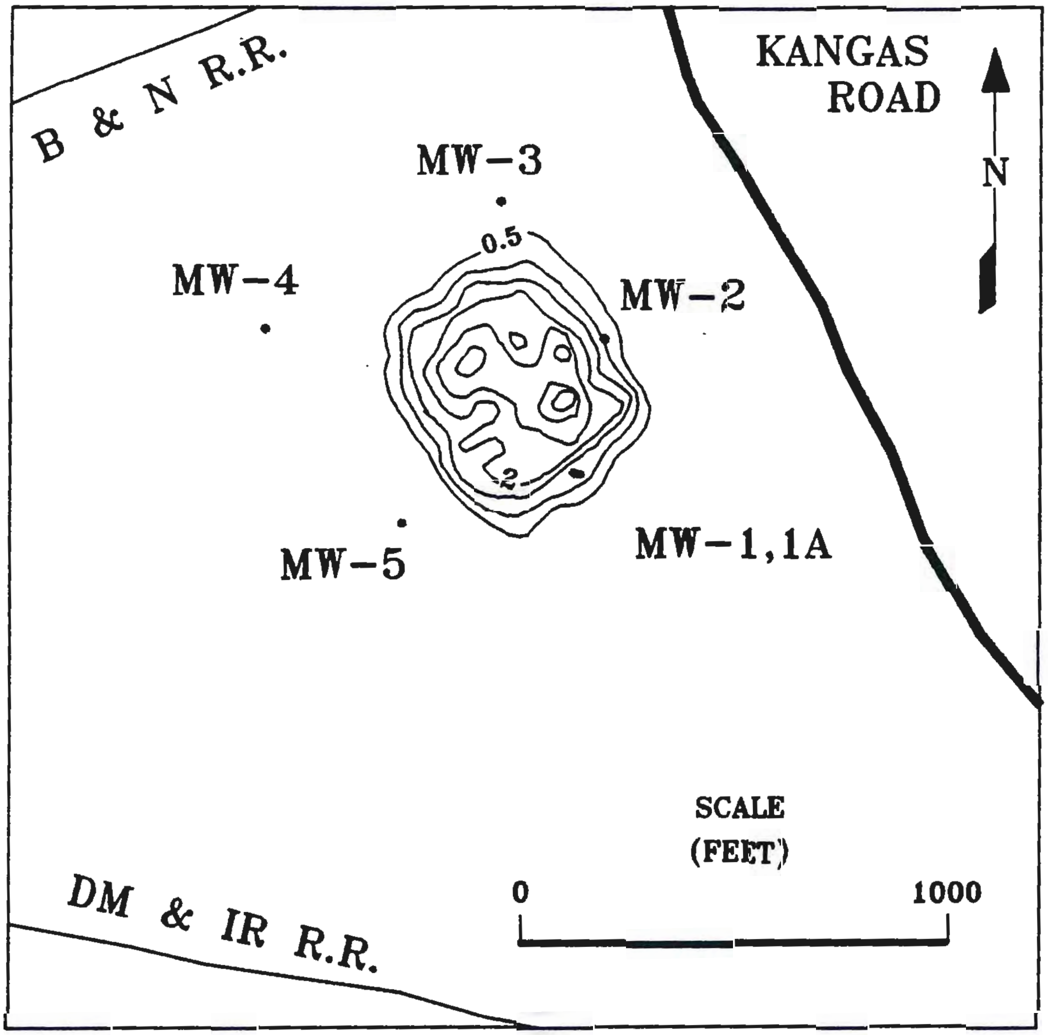

Figure 23 Concentration contour map of predicted cadmium values at the Kitzville Dump (2010). The contour interval is 0.5 ug/l.

SECTION VII

SUMMARY AND CONCLUSIONS

The Kitzville Dump was utilized as an industrial and municipal solid waste disposal site for approximately 35 years. In 1981 the site was capped and six monitoring wells were installed. Chemical analysis of groundwater samples taken from two of these wells indicated that leachate from the site had entered the groundwater.

The site has approximately 170 feet of glacial sediments overlying a bedrock surface of Precambrian argillite. Contained within these sediments are two confined glacial outwash aquifers; one shallow aquifer and a deep aquifer. The shallow aquifer, containing the contaminant plume, has a mean hydraulic conductivity of 8.81×10^{-4} cm\sec. The general groundwater flow direction is from the northwest to the southeast at an average rate of 20 feet per year.

Groundwater flow modeling of the site was accomplished using the Intersat numerical flow model. A three-layer system was modeled, where the top and bottom layers were confining units for the shallow aquifer between. A variable, rectangular grid system, covering approximately 174 acres, was used to discretize the study region. Constant head boundaries, oriented in the general direction of regional groundwater flow, set the initial hydraulic gradient.

The Hydrologic Evaluation of Landfill Performance (HELP) Model provided estimated leachate production values for both the uncapped and capped site configurations. These values indicate that the amount of leachate production was reduced by an estimated 59 percent when the site was capped in 1981.

Steady-state conditions were simulated, with the primary calibration variable being site recharge. The calibration procedure involved numerous simulation runs to reach a reasonable match with observed head data.

Parameter uncertainty exists in the form of either measurement errors or intrinsic uncertainty of physical properties or natural processes. Sensitivity analyses revealed that the estimated parameters used in the groundwater flow model represented realistic field values.

The quality of groundwater at the site is monitored on a routine basis and indicates that leachate from the site has entered the shallow aquifer. Values of chloride at monitoring wells MW-1 and MW-2 show a nine year history of elevated values, indicating that leachate is entering the aquifer on a continuing basis.

Water quality standards set by state and federal agencies represent contaminant levels that can cause adverse affects to varying degrees. Regionally, the quality of groundwater from the glacial outwash aquifers is ranked among the best in the state. Leachate from solid waste sites can contain very high concentrations of both organic and inorganic compounds. The chloride levels at monitoring well MW-1 exceed the Recommended Allowable Limit of 250 mg/l set by the state of Minnesota. Chemical analyses also show elevated values for other chemical species at the site.

Solute transport modeling of the contaminant plume at the site was accomplished with the Intertrans solute transport model. This companion program to the Intersat groundwater flow model, simulates three-dimensional hydrodynamic dispersion. Advective parameters and model dimensions are defined by the groundwater flow model and imported into the solute transport model. Various equations are used to represent the processes involved with solute transport and its corresponding representation in the numerical transport model. The processes involved in the solute transport simulation include advection, diffusion, dispersion, and retardation.

An electrical resistivity survey was performed at the site to try to identify the present extent of the contaminant plume. The results of this survey in comparison with the modeled extent of the contaminant plume indicate only marginal success. The results of the resistivity survey to the south of the site appear to define a flank of the contaminant plume. This limited success is due, in part, to the unsuitability of testing of much of the region of interest.

Seismic analyses were also performed to delineate the extent and thickness of the solid waste in poorly defined areas. The extreme heterogeneity of the solid waste in these regions severely limited the value of this data.

Various transport parameters were entered into the solute transport model to represent field conditions at the site. These values included dispersivity, retardation and a defined continuous contaminant source. Chloride was used as the primary calibrative parameter for the solute transport model.

Two simulation phases were used to represent solute transport at the site. The first phase represented the 34 years of operation when the site was uncapped and leachate generation was at its highest level. The second phase covered the nine years since the site was capped and leachate generation was significantly less. In both phases the model was calibrated to sampled chloride concentrations. Sensitivity analyses of various transport parameters, including longitudinal and transverse dispersivity, indicate that the values used in the model are representative of field values. Table 7 provides a summary of the various flow and transport parameters as optimized in this study.

A 10-year predictive simulation of chloride concentrations at the site indicate the general movement of

TABLE 7

SUMMARY TABLE OF FLOW AND TRANSPORT PARAMETERS AS OPTIMIZED IN THIS STUDY.

Geology	Glacial Till and Outwash
Classification of Aquifer Materials	Silty Sand (SM)
Hydraulic Conductivity (Mean)	8.81×10^{-4} cm/sec
Average Ground Water Velocity	20 ft/yr
Dispersivity	
Longitudinal	50 feet
Transverse	5 feet
Retardation Factors	
Benzene	17
Toluene	44
Lead	186
Cadmium	248

the plume is toward the southeast. This corresponds with the general groundwater flow direction. This predictive simulation also shows that the concentration levels of chloride at the site do not change substantially from present values. This agrees with data from the past nine years, which shows that the concentration levels at the site have remained relatively stable.

Analysis of organics and metals at the site show that the general direction of migrating solute fronts corresponds with the general direction of groundwater flow. The higher retardation values for these chemical species severely restrict their movement in relation to chloride.

Even though the chloride levels at the site are above recommended levels (RALs), the predicted extent of the contaminant plume poses no threat to any existing water supplies. The organics and metals at the site, while having significantly lower MCL values, also do not pose an immediate threat to any existing water supplies. However, should future development of property occur near the site, further analysis of the groundwater would be essential to ensuring adequate health standards are maintained.

Future research at this site might include analysis of the weight fraction of organic carbon in the soil. This would provide greater accuracy in the determination of retardation factors for organic contaminants. Also another round of sampling at wells MW-1 and MW-2 for organics and metals would be beneficial to any future research at this site.

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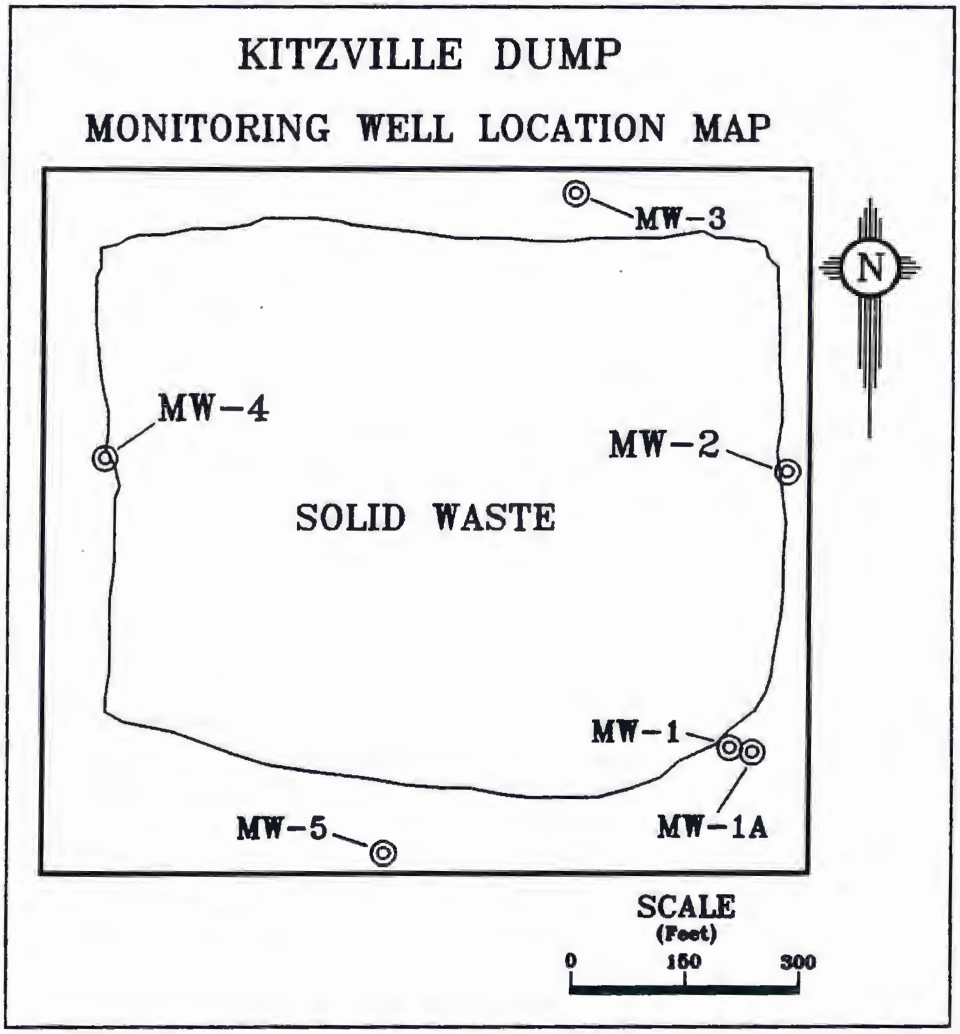
APPENDIX A

SOIL BORING LOGS

All soil boring data obtained from Braun Engineering Testing reports completed November 1982.



Note: The following log tables are not scaled.

ST-1 / MW-1

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1449.2	0.0		
1447.2	2.0	CLAYEY SILT, slightly organic, dark brown to black, some roots, wet. **(TOPSOIL)**	
1425.2	24.0	SILTY CLAY, a trace of fine to medium gravel, ranging in color from brown and brown mottled with gray to grayish brown, wet, rather stiff to very stiff. **(GLACIAL TILL)**	< 1440.2
1415.2	34.0	SLIGHTLY SILTY SAND, fine to medium grained, a trace of fine gravel, brown to grayish brown, water bearing, medium dense. **(GLACIAL OUTWASH)**	Screen 1410.0 1415.0
1401.2	48.0	SILTY SAND AND GRAVEL, numerous boulders, brown, wet, dense to very dense. **(GLACIAL TILL)**	

ST-1A / MW-1A

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1449.2	0.0		
1447.2	2.0	CLAYEY SILT, slightly organic, black, some roots, wet. (TOPSOIL)	
1425.2	24.0	SILTY CLAY, a trace of fine to medium gravel, ranging in color from brown to grayish brown and brownish gray, wet. (GLACIAL TILL)	< 1440.2
1414.2	35.0	SLIGHTLY SILTY SAND, fine to medium grained, a trace of fine gravel, a few boulders, brown, water bearing. (GLACIAL OUTWASH)	
1391.2	58.0	SILTY SAND AND GRAVEL, numerous boulders, brown, wet. (GLACIAL TILL)	
1385.2	64.0	GRAVEL AND BOULDERS. (GLACIAL TILL)	
1372.2	77.0	SILTY SAND, fine to medium grained, some fine to medium gravel, numerous boulders, brown to grayish brown, wet, very dense. (GLACIAL TILL)	
1362.2	87.0	SLIGHTLY SILTY SAND, fine to coarse grained, some fine gravel, grayish brown, a few seams and layers of brown sandy silt, water bearing, dense. (GLACIAL OUTWASH)	Screen 1370.2 1365.2

ST-2 / MW-2

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1446.7	0.0		
1444.7	2.0	CLAYEY SILT, slightly organic, black, some roots, wet. (TOPSOIL)	
1419.7	27.0	SILTY CLAY, a trace of fine to medium gravel, ranging in color from brown and reddish brown to grayish brown and brownish gray, moist to wet, rather stiff to stiff. (GLACIAL TILL)	< 1438.7
1396.2	50.5	SLIGHTLY SILTY SAND, mostly fine to medium grained, a trace of fine gravel, a few boulders between 42' and 50', ranging in color from brown and brown mottled with gray to grayish brown, water bearing, very loose to medium dense. (GLACIAL OUTWASH)	Screen 1416.7 – 1411.7

ST-3 / MW-3

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1446.3	0.0		
1445.3	1.0	SILTY CLAY, black, some roots, wet. (TOPSOIL)	
1441.8	4.5	SILTY CLAY, a trace of fine to medium gravel, brown, wet, rather stiff. (GLACIAL TILL)	< 1443.3
1427.3	19.0	SLIGHTLY SILTY SAND, mostly fine to medium grained, a trace of fine gravel, brown to grayish brown, water bearing, loose to medium dense. (GLACIAL OUTWASH)	Screen 1438.3 1433.3
1419.3	27.0	SAND, mostly medium grained, a trace of fine gravel, brown, water bearing, medium dense. (GLACIAL OUTWASH)	
1415.8	30.5	SILTY SAND, fine to medium grained, some fine to medium gravel, some boulders, brown, wet, dense. (GLACIAL TILL)	

ST-4 / MW-4

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1470.5	0.0		
1463.5	7.0	SILTY CLAY, brown to reddish brown, roots to 6", moist to wet, very stiff. (GLACIAL TILL)	
1432.5	38.0	SILTY SAND, fine to medium grained, some fine to medium gravel, numerous boulders, ranging in color from brown and brown mottled with rust to grayish brown and brownish gray, moist to wet, dense to very dense. (GLACIAL TILL)	< 1445.5 Screen 1439.5 1434.5

ST-5 / MW-5

Elev. (Feet)	Depth (Feet)	Description	Water Level (Feet)
1459.1	0.0		
1448.6	10.5	SILTY CLAY, a trace of fine to medium gravel, brown to reddish brown, moist to wet, stiff to very stiff. (GLACIAL TILL)	
1433.1	26.0	SLIGHTLY SILTY SAND AND GRAVEL, numerous boulders, brown, moist to wet, water bearing. (GLACIAL OUTWASH)	< 1441.6 Screen 1439.1 1434.1
1427.1	32.0	SLIGHTLY SILTY SAND, fine to medium grained, a little fine to medium gravel, brown, water bearing, very dense. (GLACIAL OUTWASH)	

APPENDIX B

MONITORING WELL HYDRAULIC CONDUCTIVITY DATA

MW-1

Static Water Level
Feet above MSL: 1436.24
Feet below TOC: 12.96

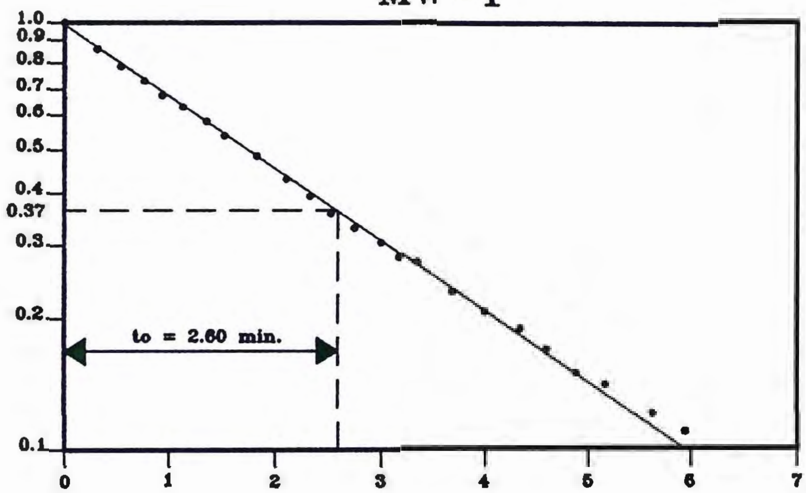
Depth of Well (ft): 37.83
Ho (ft): 3.81

Elapsed Time (*Minutes*)	Water Depth (Feet)	s' (Feet)	H/Ho
0.00	16.77	3.81	1.00
0.32	16.25	3.29	0.86
0.53	15.96	3.00	0.79
0.75	15.75	2.79	0.73
0.92	15.54	2.58	0.68
1.12	15.38	2.42	0.64
1.35	15.19	2.23	0.59
1.53	15.02	2.06	0.54
1.82	14.81	1.85	0.49
2.10	14.60	1.64	0.43
2.33	14.46	1.50	0.39
2.55	14.33	1.37	0.36
2.77	14.23	1.27	0.33
3.02	14.13	1.17	0.31
3.17	14.04	1.08	0.28
3.33	13.98	1.02	0.27
3.68	13.85	0.89	0.23
4.00	13.75	0.79	0.21
4.33	13.67	0.71	0.19
4.58	13.60	0.64	0.17
4.87	13.54	0.58	0.15
5.17	13.48	0.52	0.14
5.62	13.42	0.46	0.12
5.92	13.38	0.42	0.11

* All measurements made from top of well casing.

Head Ratio vs. Time

MW-1



Time t (minutes)

Calculation of Hydraulic Conductivity (K)

$$K = \frac{R^2 \ln(L/R)}{2Lt_o}$$

R = Radius of well casing = 2.54 cm
L = Length of well screen = 152.40 cm
t_0 = Time necessary for the water level to rise or fall to 37 percent of the initial change.

The two values listed above are the same for all monitoring wells on the site.

t_0 (from graph) = 2.60 min. = 156 sec.

$$K = \frac{(2.54)^2 \ln(152.40/2.54)}{2(152.40)(156)}$$

$$= 5.56 \times 10^{-4}\ cm/sec$$

MW-1A

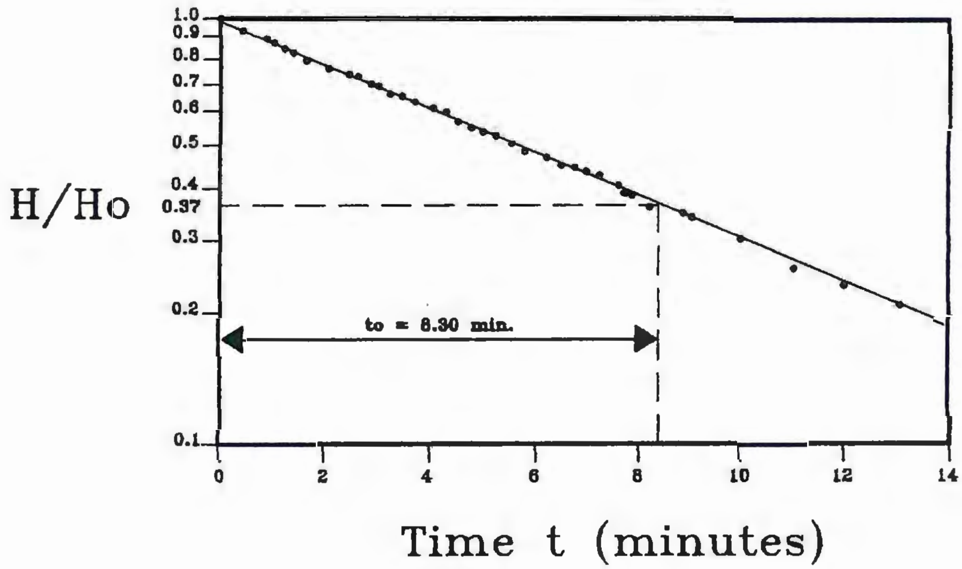
Static Water Level
Feet above MSL: 1436.24
Feet below TOC: 12.92

Depth of Well (ft): 89.50
Ho (ft): 5.29

Elapsed Time (Minutes)	Water Depth (Feet)	s' (Feet)	H/Ho
0.00	18.21	5.29	1.00
0.37	17.83	4.91	0.93
0.78	17.58	4.66	0.88
1.03	17.46	4.54	0.86
1.27	17.33	4.41	0.83
1.48	17.21	4.29	0.81
1.80	17.08	4.16	0.79
2.07	16.92	4.00	0.76
2.33	16.77	3.85	0.73
2.62	16.71	3.79	0.72
2.85	16.58	3.66	0.69
3.10	16.50	3.58	0.68
3.33	16.42	3.50	0.66
3.52	16.29	3.37	0.64
3.75	16.21	3.29	0.62
4.02	16.08	3.16	0.60
4.25	16.02	3.10	0.59
4.48	15.92	3.00	0.57
4.73	15.83	2.91	0.55
5.02	15.75	2.83	0.53
5.27	15.69	2.77	0.52
5.53	15.58	2.66	0.50
5.82	15.48	2.56	0.48
6.17	15.38	2.46	0.47
6.43	15.31	2.39	0.45
6.67	15.25	2.33	0.44
6.93	15.17	2.25	0.43
7.20	15.13	2.21	0.42
7.45	15.06	2.14	0.40
7.67	15.00	2.08	0.39
7.90	14.94	2.02	0.38
8.30	14.83	1.91	0.36
8.55	14.81	1.89	0.36
8.83	14.75	1.83	0.35
9.07	14.73	1.81	0.34
10.00	14.50	1.58	0.30
11.00	14.29	1.37	0.26
12.00	14.15	1.23	0.23
13.00	14.04	1.12	0.21

Head Ratio vs. Time

MW-1A



t_0 = 8.30 min. = 498 sec.

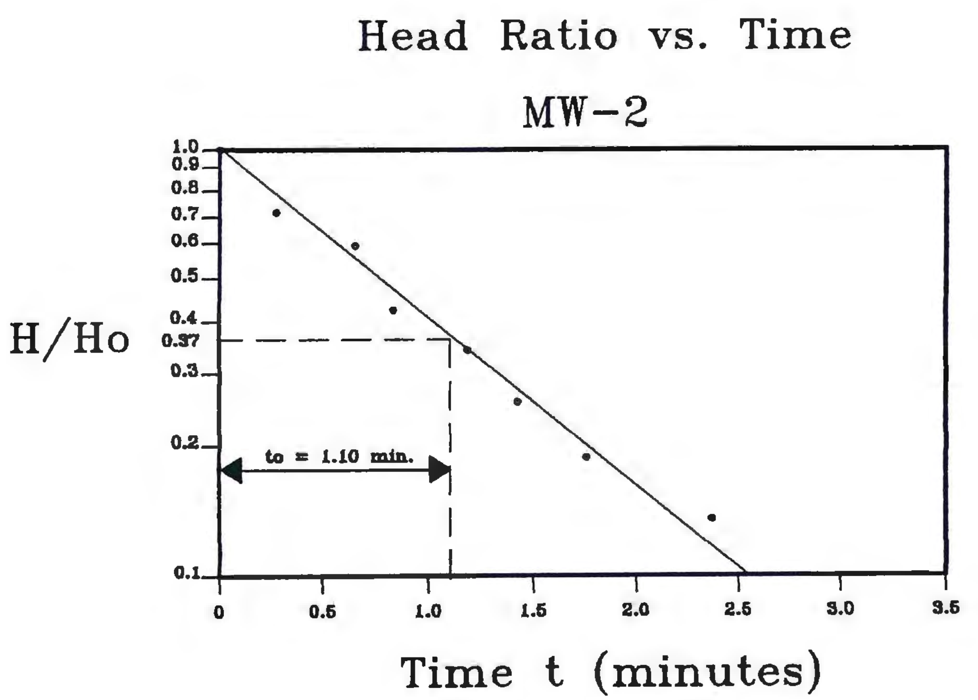
K = 1.74 x 10^{-4} cm/sec

MW-2

Static Water Level
  Feet above MSL: 1433.82
  Feet below TOC: 12.88

Depth of Well (ft): 38.25
Ho (ft): 0.87

Elapsed Time (Minutes)	Water Depth (Feet)	s (Feet)	H/Ho
0.00	12.01	0.87	1.00
0.33	12.26	0.62	0.71
0.59	12.31	0.57	0.65
0.82	12.51	0.37	0.43
1.22	12.58	0.30	0.34
1.55	12.64	0.24	0.28
1.75	12.71	0.17	0.19
2.35	12.76	0.12	0.14



To = 1.10 min. = 66 sec.

K = 1.31 x 10^{-3} cm/sec

MW-3

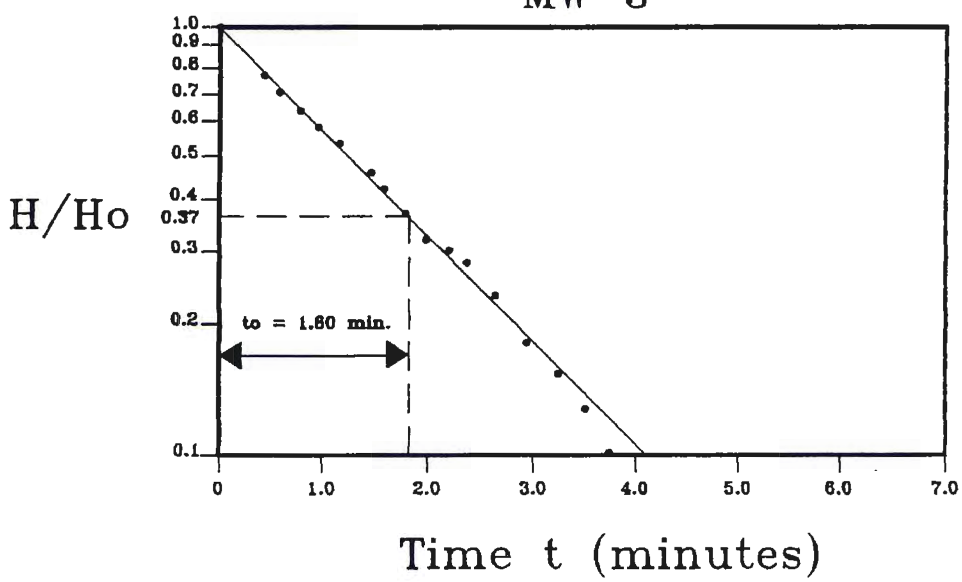
Static Water Level
Feet above MSL: 1438.55
Feet below TOC: 7.75

Depth of Well (ft): 16.04
Ho (ft): 4.29

Elapsed Time (Minutes)	Water Depth (Feet)	s' (Feet)	H/Ho
0.00	12.04	4.29	1.00
0.37	11.08	3.33	0.78
0.58	10.77	3.02	0.70
0.82	10.42	2.67	0.62
0.97	10.21	2.46	0.57
1.15	10.00	2.25	0.52
1.40	9.71	1.96	0.46
1.57	9.54	1.79	0.42
1.80	9.35	1.60	0.37
2.00	9.12	1.37	0.32
2.17	9.04	1.29	0.30
2.33	8.93	1.18	0.28
2.68	8.73	0.98	0.23
2.97	8.54	0.79	0.18
3.25	8.42	0.67	0.16
3.50	8.29	0.54	0.13
3.75	8.19	0.44	0.10

Head Ratio vs. Time

MW-3



To = 1.80 min. = 108 sec.

K = 8.02×10^{-4} cm/sec

MW-4

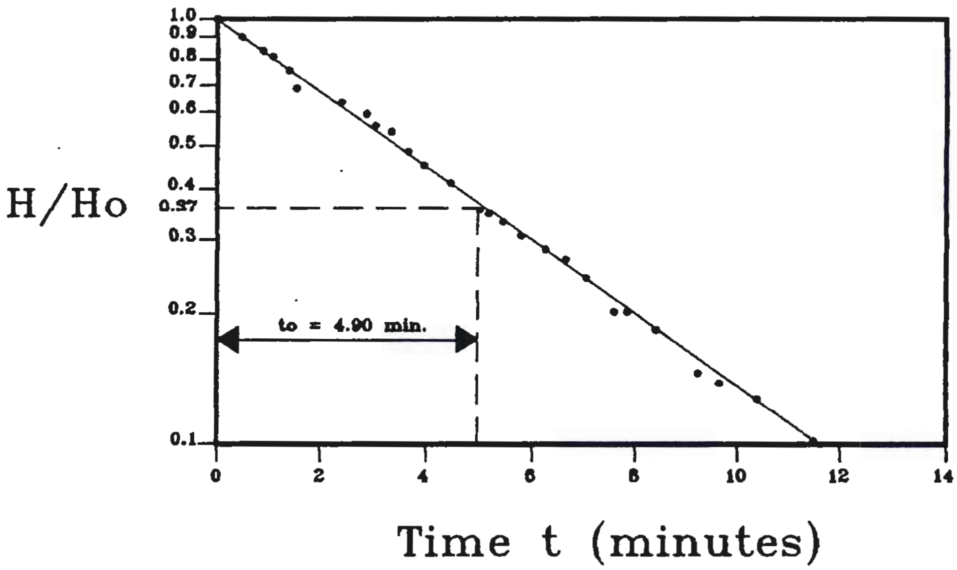
Static Water Level
    Feet above MSL: 1440.00
    Feet below TOC: 30.50

Depth of Well (ft): 39.50
Ho (ft): 4.12

Elapsed Time (Minutes)	Water Depth (Feet)	s (Feet)	H/Ho
0.00	26.38	4.12	1.00
0.43	26.75	3.75	0.91
0.78	27.08	3.42	0.83
1.12	27.17	3.33	0.81
1.40	27.42	3.08	0.75
1.58	27.67	2.83	0.69
2.33	27.92	2.58	0.63
2.73	28.08	2.42	0.59
3.00	28.25	2.25	0.55
3.22	28.33	2.17	0.53
3.65	28.54	1.96	0.48
3.97	28.67	1.83	0.44
4.37	28.83	1.67	0.41
4.92	29.00	1.50	0.36
5.17	29.06	1.44	0.35
5.40	29.13	1.37	0.33
5.80	29.21	1.29	0.31
6.28	29.33	1.17	0.28
6.62	29.40	1.10	0.27
7.07	29.52	0.98	0.24
7.62	29.67	0.83	0.20
7.95	29.69	0.81	0.20
8.43	29.75	0.75	0.18
9.22	29.88	0.62	0.15
9.77	29.92	0.58	0.14
10.42	29.96	0.54	0.13
11.45	30.08	0.42	0.10

Head Ratio vs. Time

MW-4



T_o = 4.90 min. = 294 sec.

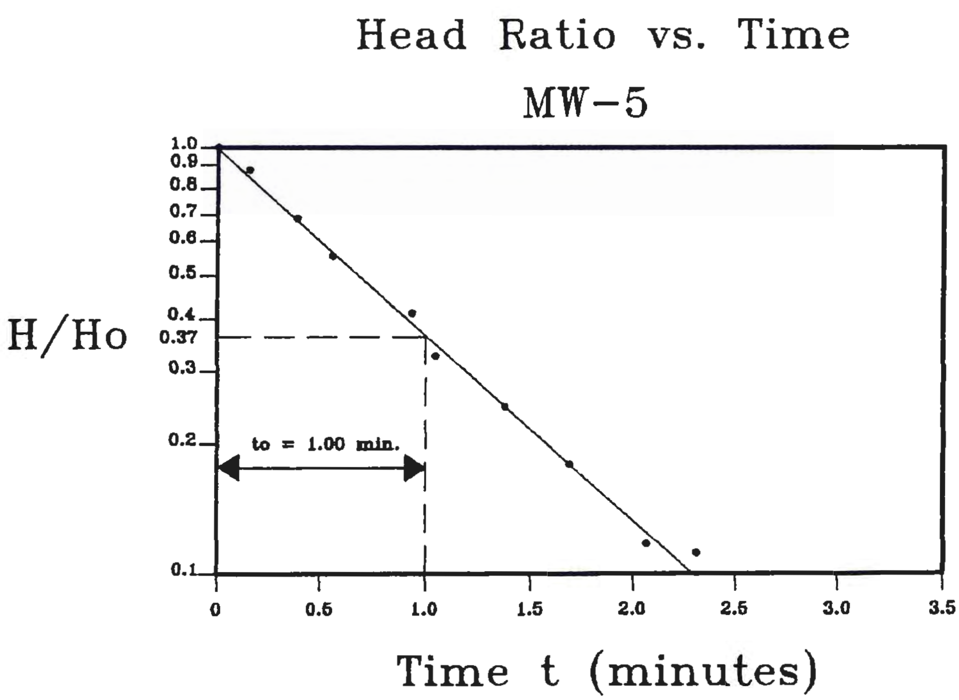
K = 2.95×10^{-4} cm/sec

MW-5

Static Water Level
Feet above MSL: 1435.58
Feet below TOC: 23.52

Depth of Well (ft): 39.50
Ho (ft): 4.12

Elapsed Time (Minutes)	Water Depth (Feet)	s (Feet)	H/Ho
0.00	22.29	1.23	1.00
0.21	22.44	1.08	0.88
0.43	22.67	0.85	0.69
0.58	22.84	0.68	0.55
0.94	23.02	0.50	0.41
1.07	23.11	0.41	0.33
1.38	23.30	0.22	0.25
1.71	23.36	0.16	0.18
2.08	23.35	0.15	0.12
2.35	23.34	0.14	0.11



To = 1.00 min. = 60 sec.

K = 1.44 x 10^{-3} cm/sec

APPENDIX C

Geulph Permeameter Field Data and Calculations

Location: North of site, near MW-3
Depth: 26 cm

Height of water in well: 5 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	2.2	-	-
10	10	2.2	0.0	0.00
20	10	2.2	0.0	0.00
25	5	2.5	0.7	0.14
30	5	2.5	0.0	0.00
40	10	3.0	0.5	0.05
50	10	3.2	0.2	0.02
60	10	3.6	0.4	0.04
70	10	4.3	0.7	0.07
75	5	4.6	0.3	0.06
80	5	4.9	0.3	0.06
85	5	5.2	0.3	0.06

Height of water in well: 10 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	50.5	-	-
2	2	50.9	0.4	0.2
4	2	51.3	0.4	0.2
6	2	51.7	0.4	0.2
8	2	52.1	0.4	0.2

R_1 = 0.06 cm/min = 0.001 cm/sec
R_2 = 0.2 cm/min = 0.0033 cm/sec

$$K = [(\text{Shape Factor}_2)(\text{Reservoir Constant})(R_2)] - [(\text{Shape Factor}_1)(\text{Reservoir Constant})(R_1)]$$

Inner Reservoir Constant = 2.19 cm^2
Shape $Factor_1$ = 0.0054
Shape $Factor_2$ = 0.0041

$$K = [(0.0041)(2.19)(0.0033)] - [(0.0054)(2.19)(0.001)]$$

$$= 1.81 \times 10^{-5} \text{ cm/sec}$$

Location: North of site, near MW-3
Depth: 51 cm

Height of water in well: 5 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	0.8	-	-
15	15	0.8	0.0	0.00
30	15	0.8	0.0	0.00
60	30	1.2	0.4	0.01
90	30	1.7	0.5	0.02
100	10	1.9	0.2	0.02
110	10	2.2	0.3	0.03
120	10	2.5	0.3	0.03
130	10	2.8	0.3	0.03

Height of water in well: 10 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	48.0	-	-
2	2	48.3	0.3	0.15
4	2	48.4	0.1	0.05
6	2	48.6	0.2	0.10
16	10	48.9	0.3	0.03
26	10	49.3	0.4	0.04
36	10	49.7	0.4	0.04
46	10	50.1	0.4	0.04

R_1 = 0.03 cm/min = 0.0005 cm/sec
R_2 = 0.04 cm/min = 0.00067 cm/sec

K = [(0.0041)(2.19)(0.00067)] - [(0.0054)(2.19)(0.0005)]

= 1.03×10^{-7} cm/sec

Location: North end of cover material.
Depth: 17 cm

Height of water in well: 5 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	1.5	-	-
5	5	2.1	0.6	0.12
15	10	2.7	0.6	0.06
25	10	3.3	0.6	0.06
35	10	3.4	0.1	0.01
45	10	3.8	0.4	0.04
55	10	4.2	0.4	0.04
65	10	4.6	0.4	0.04
75	10	5.0	0.4	0.04

Height of water in well: 10 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	55.7	-	-
2	2	56.6	0.9	0.45
4	2	57.1	0.5	0.25
6	2	57.5	0.4	0.20
8	2	57.9	0.4	0.20
10	2	58.2	0.3	0.15
12	2	58.5	0.3	0.15
14	2	58.8	0.3	0.15

R_1 = 0.04 cm/min = 0.00067 cm/sec
R_2 = 0.15 cm/min = 0.0025 cm/sec

$K = [(0.0041)(2.19)(0.0025)] - [(0.0054)(2.19)(0.00067)]$

$= 1.45 \times 10^{-5}$ cm/sec

Location: North end of cover material.
Depth: 41 cm

Height of water in well: 5 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	3.3	-	-
2	2	4.8	1.5	0.75
4	2	6.3	1.5	0.75
6	2	7.8	1.5	0.75
8	2	9.3	1.5	0.75
10	2	10.8	1.5	0.75

Height of water in well: 10 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	16.0	-	-
1	1	18.2	2.2	2.2
2	1	19.8	1.6	1.6
3	1	21.6	1.8	1.8
4	1	23.4	1.8	1.8
5	1	25.2	1.8	1.8
6	1	27.0	1.8	1.8
7	1	28.8	1.8	1.8

Outer Reservoir Constant = 35.11 cm^2

R_1 = 0.75 cm/min = 0.0125 cm/sec
R_2 = 1.80 cm/min = 0.03 cm/sec

$K = [(0.0041)(35.11)(0.03)] - [(0.0054)(35.11)(0.0125)]$
$= 1.59 \times 10^{-3}$ cm/sec

Location: South end of cover material.
Depth: 24 cm

Height of water in well: 5 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	1.2	-	-
5	5	1.7	0.5	0.10
15	10	2.2	0.5	0.10
25	10	2.6	0.4	0.04
35	10	2.8	0.2	0.02
45	10	3.2	0.4	0.04
55	10	3.5	0.3	0.03
65	10	3.8	0.3	0.03
75	10	4.1	0.3	0.03

Height of water in well: 10 cm

Time (min)	Time Interval (min)	Reservoir Water Level (cm)	Water Level Change (cm)	Rate of Water Level Change (cm/min)
0	-	51.4	-	-
2	2	52.2	0.8	0.40
4	2	52.9	0.7	0.35
6	2	53.6	0.7	0.35
8	2	54.2	0.6	0.30
10	2	54.7	0.5	0.25
12	2	55.2	0.5	0.25
14	2	55.7	0.5	0.25

R_1 = 0.03 cm/min = 0.0005 cm/sec
R_2 = 0.25 cm/min = 0.0042 cm/sec

$K = [(0.0041)(2.19)(0.0042)] - [(0.0054)(2.19)(0.0005)]$

$= 3.18 \times 10^{-5}$ cm/sec

APPENDIX D

STATIC WATER LEVEL DATA

STATIC WATER LEVEL = Water level elevation in feet above MSL
DTW = Depth to water from top of casing in feet

MW-1

Top of Casing Elevation (Feet) = 1449.20

Date	DTW	Static Water Level
18 Nov 82	9.20	1440.00
25 Jan 83	10.80	1438.40
21 Mar 83	10.40	1438.80
14 Sep 83	10.10	1439.10
5 Mar 84	11.03	1438.17
14 May 84	11.17	1438.03
16 Jul 84	9.40	1439.80
7 Sep 84	11.26	1437.94
18 Oct 84	12.00	1437.20
2 Apr 85	14.03	1435.17
15 Jul 85	10.83	1438.37
14 Oct 85	12.00	1437.20
7 Apr 86	11.75	1437.45
30 Jun 86	10.83	1438.37
1 Oct 86	10.58	1438.62
20 Apr 87	10.50	1438.70
30 Jun 87	11.30	1437.90
1 Oct 87	11.71	1437.49
20 Apr 88	11.98	1437.22
29 Jun 88	13.34	1435.86
29 Sep 88	11.10	1438.10
21 Apr 89	10.19	1439.01
8 Oct 89	13.00	1436.20

MW-1A

Top of Casing Elevation (Feet) = 1449.20

Date	DTW	Static Water Level
18 Nov 82	9.20	1440.00
25 Jan 83	10.80	1438.40
21 Mar 83	10.40	1438.80
14 Sep 83	10.10	1439.10
14 May 84	11.50	1437.70
16 Jul 84	9.60	1439.60
7 Sep 84	11.01	1438.19
18 Oct 84	10.26	1438.94
2 Apr 85	14.07	1435.13
15 Jul 85	11.83	1437.37
14 Oct 85	11.00	1438.20
7 Apr 86	12.17	1437.03
30 Jun 86	11.83	1437.37
1 Oct 86	11.75	1437.45
20 Apr 87	11.59	1437.61
30 Jun 87	12.02	1437.18
1 Oct 87	12.68	1436.52
20 Apr 88	12.20	1437.00
29 Jun 88	14.67	1434.53
29 Sep 88	12.06	1437.14
21 Apr 89	11.14	1438.06
8 Oct 89	12.92	1436.28

MW-2

Top of Casing Elevation (Feet) = 1446.70

Date	DTW	Static Water Level
18 Nov 82	8.10	1438.60
25 Jan 83	9.60	1437.10
21 Mar 83	9.40	1437.30
14 Sep 83	9.10	1437.60
5 Mar 84	9.91	1436.79
14 May 84	10.58	1436.12
16 Jul 84	10.20	1436.50
7 Sep 84	9.99	1436.71
18 Oct 84	10.07	1436.63
2 Apr 85	14.66	1432.04
15 Jul 85	10.50	1436.20
14 Oct 85	11.83	1434.87
7 Apr 86	12.33	1434.37
30 Jun 86	11.17	1435.53
1 Oct 86	11.33	1435.37
20 Apr 87	11.36	1435.34
30 Jun 87	11.57	1435.13
1 Oct 87	12.36	1434.34
20 Apr 88	12.67	1434.03
29 Jun 88	13.93	1432.77
29 Sep 88	12.24	1434.46
21 Apr 89	11.04	1435.66
8 Oct 89	12.88	1433.82

MW-3

Top of Casing Elevation (Feet) = 1446.30

Date	DTW	Static Water Level
18 Nov 82	3.30	1443.00
25 Jan 83	5.10	1441.20
21 Mar 83	4.50	1441.80
14 Sep 83	3.20	1443.10
5 Mar 84	4.87	1441.43
14 May 84	4.17	1442.13
16 Jul 84	3.30	1443.00
7 Sep 84	5.33	1440.97
18 Oct 84	5.66	1440.64
2 Apr 85	9.60	1436.70
15 Jul 85	6.13	1440.17
14 Oct 85	6.29	1440.01
7 Apr 86	5.67	1440.63
30 Jun 86	5.42	1440.88
1 Oct 86	4.17	1442.13
20 Apr 87	5.05	1441.25
30 Jun 87	6.11	1440.19
1 Oct 87	7.59	1438.71
20 Apr 88	7.31	1438.99
29 Jun 88	8.94	1437.36
29 Sep 88	6.54	1439.76
21 Apr 89	4.60	1441.70
8 Oct 89	7.17	1439.13

MW-4

Top of Casing Elevation (Feet) = 1470.50

Date	DTW	Static Water Level
18 Nov 82	26.50	1444.00
25 Jan 83	28.00	1442.50
21 Mar 83	27.90	1442.60
14 Sep 83	27.90	1442.60
5 Mar 84	28.43	1442.07
14 May 84	30.00	1440.50
16 Jul 84	19.60	1450.90
7 Sep 84	28.08	1442.42
18 Oct 84	28.89	1441.61
2 Apr 85	33.34	1437.16
15 Jul 85	29.42	1441.08
14 Oct 85	30.42	1440.08
7 Apr 86	29.75	1440.75
30 Jun 86	29.17	1441.33
1 Oct 86	29.58	1440.92
20 Apr 87	29.44	1441.06
30 Jun 87	29.29	1441.21
1 Oct 87	30.15	1440.35
20 Apr 88	31.36	1439.14
29 Jun 88	32.09	1438.41
29 Sep 88	30.42	1440.08
21 Apr 89	28.65	1441.85
8 Oct 89	30.50	1440.00

MW-5

Top of Casing Elevation (Feet) = 1459.10

Date	DTW	Static Water Level
18 Nov 82	17.50	1441.60
25 Jan 83	19.80	1439.30
21 Mar 83	19.60	1439.50
14 Sep 83	19.60	1439.50
5 Mar 84	20.34	1438.76
14 May 84	21.83	1437.27
16 Jul 84	18.90	1440.20
7 Sep 84	20.40	1438.70
18 Oct 84	20.78	1438.32
2 Apr 85	20.17	1438.93
15 Jul 85	22.33	1436.77
14 Oct 85	22.50	1436.60
7 Apr 86	20.58	1438.52
30 Jun 86	22.42	1436.68
1 Oct 86	22.50	1436.60
20 Apr 87	22.31	1436.79
30 Jun 87	22.69	1436.41
1 Oct 87	23.11	1435.99
20 Apr 88	24.10	1435.00
29 Jun 88	25.26	1433.84
29 Sep 88	23.06	1436.04
21 Apr 89	21.69	1437.41
8 Oct 89	23.42	1435.68

APPENDIX E

CHEMICAL ANALYTICAL DATA

CHLORIDE

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
10 Nov 82	700.00	17.00	360.00	25.00	9.50	14.00
13 Jan 83		11.00			8.00	14.00
14 Sep 83	326.00		194.00	12.00	2.00	15.00
14 May 84	250.00	66.00	173.00	7.00	2.00	4.00
16 Jul 84	67.00	393.00	155.00	10.00	2.00	6.00
18 Oct 84	350.00	30.00	148.00	12.00	20.00	10.00
2 Apr 85	250.00	18.60	89.00	9.30	2.48	13.70
15 Jul 85	375.00	20.00	105.00	4.50	<0.10	14.50
14 Oct 85	200.00	30.00	84.00	2.30	5.00	1.70
7 Apr 86	325.00	19.50	90.00	7.70	12.00	12.80
30 Jun 86	260.00	42.00	80.00	10.50	3.50	5.00
1 Oct 86	319.00	20.00	87.50	11.50	3.50	0.50
20 Apr 87	362.00	17.50	100.00	10.00	6.50	6.50
30 Jun 87	390.00	20.50	75.00	5.50	7.50	12.00
1 Oct 87	450.00	19.00	81.00	16.00	13.00	14.50
20 Apr 88	375.00	15.00	72.00	18.00	14.00	9.00
29 Jun 88	375.00	17.00	95.00	29.50	25.00	11.50
29 Sep 88	216.00	12.50	90.00	12.50	18.00	4.00
21 Apr 89	390.00	19.20	100.00	7.50	19.50	2.40

SULFATE

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
14 May 84	230.70	20.80	283.40	26.70	5.40	18.30
16 Jul 84	15.70	179.30	248.30	151.70	8.60	15.00
18 Oct 84	204.00	11.20	206.00	74.00	17.00	11.80
2 Apr 85	204.00	11.20	206.00	74.00	17.00	11.80
15 Jul 85	19.00	17.50	205.00	120.00	14.50	19.00
14 Oct 85	26.50	21.00	180.00	110.00	2.00	13.00
7 Apr 86	135.00	18.00	180.00	158.00	13.50	20.50
30 Jun 86	175.00	23.00	150.00	320.00	10.50	14.00
1 Oct 86	190.00	26.00	135.00	320.00	11.00	14.00
20 Apr 87	172.50	17.00	100.00	275.00	12.00	13.00
30 Jun 87	210.00	18.00	160.00	130.00	13.00	29.00
1 Oct 87	233.00	15.00	225.00	235.00	25.00	15.00
20 Apr 88	250.00	18.50	210.00	220.00	19.00	29.00
29 Jun 88	255.00	16.50	200.00	245.00	16.50	20.00
29 Sep 88	268.00	16.50	188.00	393.00	16.00	13.50
21 Apr 89	282.00	19.50	182.00	210.00	14.50	11.00

AMMONIA

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
14 May 84	0.44	0.02	0.60	0.01	0.16	<.01
16 Jul 84	0.08	0.12	0.23	0.01	0.09	0.03
18 Oct 84	0.15	0.08	0.19	0.03	0.09	0.05
2 Apr 85	0.20	0.22	0.22	0.14	0.20	0.29
15 Jul 85	0.15	0.11	0.65	0.07	0.31	0.15
14 Oct 85	0.72	0.90	0.07	0.37	0.30	0.12
7 Apr 86	0.40	0.42	0.41	0.08	0.34	0.24
30 Jun 86	0.80	0.53	0.31	0.33	0.65	0.52
1 Oct 86	0.29	0.23	0.31	0.10	0.53	0.12
20 Apr 87	0.27	0.25	0.44	0.12	0.36	0.08
30 Jun 87	0.19	0.13	0.26	<.01	0.32	0.10
1 Oct 87	0.33	0.18	0.16	0.11	1.65	0.26
20 Apr 88	0.24	0.15	0.13	0.15	0.15	0.17
29 Jun 88	0.22	0.11	0.15	<.10	0.31	0.18
29 Sep 88	0.27	0.12	0.17	0.11	0.49	0.14
21 Apr 89	0.24	0.13	0.19	<.10	0.26	0.10

IRON (DISSOLVED)

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
14 May 84	0.05	0.11	16.50	0.26	13.00	0.37
16 Jul 84	0.05	0.12	13.10	0.24	10.40	0.02
18 Oct 84	1.54	0.12	13.45	0.13	15.98	0.23
2 Apr 85	0.43	0.24	8.30	0.84	17.00	0.93
15 Jul 85	5.80	0.55	9.00	<.02	9.50	0.05
14 Oct 85	0.24	5.10	7.60	0.01	11.00	0.42
7 Apr 86	2.90	0.31	9.90	<.15	0.17	0.15
30 Jun 86	5.90	0.38	10.00	<.10	11.00	<.10
1 Oct 86	4.50	0.39	7.40	<.07	3.12	0.17
20 Apr 87	8.70	0.72	9.60	0.40	2.30	0.33
30 Jun 87	10.00	0.65	8.00	<.04	8.00	0.14
1 Oct 87	16.00	1.10	8.70	<.04	38.00	<.04
20 Apr 88	12.00	0.53	15.00	0.20	19.00	1.20
29 Jun 88	12.00	0.88	5.90	0.25	11.00	1.30
29 Sep 88	14.50	1.13	8.04	0.13	12.20	0.17
21 Apr 89	17.20	2.00	18.30	0.17	21.80	0.69

NITRATE/NITRATE AS N

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
10 Nov 82	0.02	<0.01	0.01	0.10	0.01	0.02
13 Jan 83		<0.01			<0.01	0.16
14 May 84	0.18	2.17	2.30	2.22	3.26	17.91
16 Jul 84	0.20	0.97	7.46	3.77	4.06	3.08
18 Oct 84	2.39	0.84	0.48	1.48	2.90	2.90
2 Apr 85	0.18	0.25	0.32	0.25	0.19	0.25
15 Jul 85	0.37	0.43	0.44	0.50	0.37	0.56
14 Oct 85	0.22	0.20	0.18	0.52	0.82	1.25
7 Apr 86	0.18	0.26	0.08	0.20	0.55	0.90
30 Jun 86	1.12	0.16	0.31	0.14	0.08	0.31
1 Oct 86	1.80	0.55	1.02	0.46	0.30	0.55
20 Apr 87	1.10	0.25	0.44	0.26	0.15	0.26
30 Jun 87	1.40	0.30	0.64	0.16	0.25	0.64
1 Oct 87	1.20	0.23	0.51	0.22	0.19	0.43
20 Apr 88	1.00	0.20	0.35	0.15	0.15	0.20
29 Jun 88	1.20	0.26	0.47	0.27	0.30	0.26
29 Sep 88	1.20	0.22	0.45	0.34	0.31	0.39
21 Apr 89	1.05	0.15	0.61	0.20	0.18	0.25

CHEMICAL OXYGEN DEMAND (COD)

(mg/l)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
10 Nov 82	<1.00	5.00	15.00	5.00	317.00	54.00
13 Jan 83		5.00			44.00	<3.00
14 Sep 83	31.00		28.00	12.00	28.00	20.00
14 May 84	5.00	29.00	30.00	5.00	14.00	31.00
16 Jul 84	32.00	20.00	20.00	34.00	30.00	32.00
18 Oct 84	14.00	44.00	22.00	4.00	21.00	61.00
2 Apr 85	444.00	92.00	72.00	8.00	73.00	66.00
15 Jul 85	34.00	25.00	30.00	25.00	16.00	39.00
14 Oct 85	56.00	38.00	49.00	26.00	18.00	44.00
7 Apr 86	60.00	42.00	50.00	31.00	26.00	61.00
30 Jun 86	22.50	19.00	29.00	32.00	28.00	69.00
1 Oct 86	18.50	9.50	15.00	25.00	26.00	82.00
20 Apr 87	19.00	41.50	25.50	27.00	67.00	61.00
30 Jun 87	37.50	19.00	31.50	21.00	84.00	58.50
1 Oct 87	9.00	16.00	23.00	3.00	83.00	33.00
20 Apr 88	25.00	12.00	20.00	25.00	76.00	52.00
29 Jun 88	11.00	2.00	19.00	9.00	36.00	51.00
29 Sep 88	15.00	27.00	15.00	7.50	67.00	88.50
21 Apr 89	25.50	11.50	18.50	11.00	39.00	290.00

FIELD SPECIFIC CONDUCTANCE

(umhos/cm)
(Corrected to 25°C)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
10 Nov 82	990	490	940	520	470	940
13 Jan 83		371			524	218
14 Sep 83	1270		1110	580	269	180
14 May 84	1600	1400	1800	640	620	260
16 Jul 84	1200	1800	1500	740	470	220
18 Oct 84	1750	370	1250	540	390	200
2 Apr 85	1200	440	1100	370	400	230
15 Jul 85	810	235	610	285	135	125
14 Oct 85	170	230	210	395	252	199
7 Apr 86	600	200	381	248	111	111
30 Jun 86	1010	315	705	535	165	143
1 Oct 86	1000	300	650	500	1600	120
20 Apr 87	1100	261	770	302	149	139
30 Jun 87	1050	252	680	128	160	145
1 Oct 87	1100	252	690	500	295	142
20 Apr 88	1070	288	680	253	165	120
29 Jun 88	1150	269	700	382	203	107
29 Sep 88	1005	232	620	380	151	160
21 Apr 89	1260	281	770	310	192	73
8 Oct 89	1258	310	833	634	285	99

pH DATA

(Field/Lab)

Date	MW-1	MW-1A	MW-2
10 Nov 82	6.9 / 6.9	8.8 / 8.6	6.8 / 6.8
13 Jan 83		7.4 / 8.4	
14 Sep 83	10.0 / 9.7		6.6 / 6.7
14 May 84	10.6 / 10.4	10.7 / 10.5	7.1 / 7.1
16 Jul 84	10.7 / 10.8	8.4 / 8.5	7.1 / 7.0
18 Oct 84	6.9 / 7.3	7.4 / 8.1	7.7 / 6.9
2 Apr 85	9.4 / 8.0	8.7 / 7.9	6.5 / 7.1
15 Jul 85	7.1 / 7.1	7.6 / 7.6	7.1 / 7.2
14 Oct 85	7.0 / 7.4	6.9 / 6.9	7.3 / 7.0
7 Apr 86	7.8 / 7.2	8.2 / 7.6	7.7 / 7.0
30 Jun 86	7.5 / 6.7	7.7 / 7.2	7.9 / 6.8
1 Oct 86	7.9 / 6.8	8.2 / 7.4	7.9 / 6.8
20 Apr 87	7.0 / 7.0	7.6 / 7.7	7.1 / 6.9
30 Jun 87	7.6 / 6.9	8.0 / 7.6	7.7 / 6.9
1 Oct 87	7.7 / 6.3	8.1 / 6.5	7.7 / 6.4
20 Apr 88	8.2 / 6.3	8.0 / 6.8	8.2 / 6.3
29 Jun 88	7.7 / 6.5	8.2 / 6.8	8.0 / 6.9
29 Sep 88	7.5 / 6.7	8.0 / 6.3	7.5 / 6.8
21 Apr 89	6.7 / 6.6	7.0 / 7.3	6.9 / 6.8

pH DATA

(Field/Lab)

Date	MW-3	MW-4	MW-5
10 Nov 82	7.2 / 7.5	9.2 / 9.0	9.0 / 8.9
13 Jan 83		6.2 / 7.2	6.4 / 6.3
14 Sep 83	6.6 / 7.0	6.8 / 7.2	5.7 / 5.9
14 May 84	7.0 / 7.0	7.0 / 7.0	6.6 / 6.6
16 Jul 84	7.0 / 7.0	6.9 / 6.8	6.2 / 6.0
18 Oct 84	6.5 / 7.1	6.2 / 6.7	6.5 / 6.8
2 Apr 85	6.2 / 7.2	6.5 / 6.7	5.6 / 6.4
15 Jul 85	6.9 / 7.2	7.4 / 7.1	6.8 / 6.8
14 Oct 85	7.1 / 6.8	7.2 / 6.7	7.4 / 6.7
7 Apr 86	7.9 / 7.1	7.8 / 6.7	7.9 / 6.4
30 Jun 86	7.9 / 6.7	7.9 / 6.5	7.8 / 6.3
1 Oct 86	8.0 / 6.7	8.1 / 7.3	8.0 / 7.1
20 Apr 87	7.2 / 6.9	7.0 / 6.9	6.6 / 6.3
30 Jun 87	7.4 / 6.7	6.9 / 6.5	6.4 / 5.9
1 Oct 87	7.8 / 6.6	7.6 / 6.1	7.8 / 6.2
20 Apr 88	7.4 / 6.4	8.1 / 5.6	8.1 / 6.3
29 Jun 88	8.1 / 6.5	7.5 / 6.4	7.7 / 6.3
29 Sep 88	7.6 / 6.8	8.3 / 6.3	7.7 / 6.2
21 Apr 89	6.9 / 7.1	6.5 / 6.2	6.9 / 6.8

FIELD WATER TEMPERATURE

(°C)

Date	MW-1	MW-1A	MW-2	MW-3	MW-4	MW-5
14 May 84	6.0	6.0	7.0	5.0	8.0	6.0
2 Apr 85	2.0	2.0	3.0	0.0	2.0	3.0
15 Jul 85	7.9	8.3	9.3	8.0	8.6	7.1
14 Oct 85	8.3	8.9	8.8	9.0	8.9	8.9
7 Apr 86	7.1	8.0	9.0	5.0	5.8	6.7
30 Jun 86	7.6	8.2	8.8	9.0	8.2	6.8
1 Oct 86	9.0	7.9	9.0	11.0	7.8	7.9
20 Apr 87	8.2	7.9	8.2	4.9	8.9	7.0
30 Jun 87	8.0	8.0	8.5	7.6	7.9	6.9
1 Oct 87	7.1	7.5	8.9	8.8	7.1	7.1
20 Apr 88	7.1	7.8	7.3	2.8	6.3	5.8
29 Jun 88	9.0	9.3	9.8	7.8	10.7	7.9
29 Sep 88	7.3	8.0	8.9	10.8	7.3	8.9
21 Apr 89	8.5	8.9	9.1	5.9	8.0	7.7
8 Oct 89	8.5	8.7	9.8	10.4	7.0	7.5

METALS AND ORGANICS ANALYSIS

(ug/l)

	MW-2 (NOV. 1982)	MW-1	MW-2 (SEPT. 1983)	MW-4
Beryllium	0.06	<0.05	<0.05	
Cadmium	1.1			
Chromium	2.1			
Copper	20			
Lead	1			
Mercury	<0.4			
Nickel	18	12	12	
Selenium	25	9	3	
Silver	<0.04			
Zinc	130			
Benzene	<1.0	2.8	<0.5	<0.5
Toluene	<1.0	1.2	<0.5	<0.5