KANSAS GEOLOGICAL SURVEY
OPEN-FILE REPORT 87-8

EFFECTS OF THE BARTON COUNTY LANDFILL ON LOCAL WATER QUALITY

By

M. S. Kukuk

Kansas Geological Survey
1930 Constant Avenue
University of Kansas
Lawrence, KS 66047-3726

EFFECTS OF THE BARTON COUNTY LANDFILL ON LOCAL WATER QUALITY

by

Michael S. Kukuk
Kansas Geological Survey

Open-File Report 87-8
April, 1987

EFFECTS OF THE BARTON COUNTY LANDFILL ON LOCAL WATER QUALITY

Michael S. Kukuk
M.S. in Environmental Health Science

ABSTRACT

The objective of this study was to assess the present and future impacts of leachate from the Barton County sanitary landfill on water quality in the local drinking-water aquifer and the Cheyenne Bottoms Waterfowl Refuge, a downgradient wetland. This was accomplished by analyzing local ground and surface waters for organic and inorganic contaminants, estimating the quantity of leachate generated and its likely direction of flow, and evaluating the influence of landfill-operating practices on leachate generation.

A monitor-well network of seven wells was installed to sample ground-water quality inside the landfill's property limits. Eight local domestic wells and three downgradient surface-water ponds also were sampled. One shallow well tapping an unused upper aquifer was found to be highly contaminated with volatile organic chemicals (VOCs). This well was directly adjacent to some of the earliest waste cells constructed at the landfill. A second shallow well showed possible leachate contamination by inorganic chemicals and dissolved organic carbon (DOC). The well that supplies water for the landfill's scalehouse also contained trace amounts of VOCs. The remaining water-quality sampling points and the Cheyenne Bottoms seemed to be unaffected by landfill leachate. The installation of additional monitor wells in the upper aquifer, to more precisely delineate the degree of contamination that has occurred, is recommended.

A computer-based soil-moisture model was used to estimate leachate generation at the landfill. Model results show that significant quantities of leachate can theoretically be generated in average and above-average precipitation years. Vertical hydraulic conductivities (K) were determined for the landfill's cover and underlying soils. This information was used to predict rates of infiltration into the landfill as well as the direction of leachate movement out of the landfill. K-values for the landfill's cover soil were found to vary significantly and are thought to be controlled by the clay content and the degree of compaction. K-values for the underlying soils were reported to be much lower, thus a lateral direction of leachate flow is expected. Eventually, this could lead to the contamination of nearby, local domestic wells.

Current landfill-operating practices do not fully conform to accepted sanitary landfill practices in that the solid waste in the active trenches and the "wet pit" is not covered daily with soil. Thus, precipitation is allowed to fall directly on the wastes during the period the wastes are exposed. This practice may contribute significantly to leachate generation and should therefore be abandoned.

Department of Civil Engineering
April, 1987

Thesis Supervisor
Dr. Stephen J. Randtke
Associate Professor

ACKNOWLEDGMENTS

I would like to express my sincere thanks and appreciation to the following people whose assistance made this report possible:

Professor Stephen Randtke, University of Kansas, and Thomas McClain, Kansas Geological Survey (KGS) hydrologist, for their guidance and assistance throughout the course of this study;

Larry Hathaway, KGS chemist, and Richard Pierce, analyst for the Kansas Department of Health and Environment (KDHE), who provided assistance with the water-quality analyses;

Charles Linn, chief of KDHE's solid-waste section, who provided logistical support for the study;

Dullen Edman, Barton County engineer, who accommodated all our requests during the field work associated with the study;

Marla Adkins-Heljeson, KGS editor, who provided a consistent style and format for this manuscript;

Pamela Wright, KGS secretary, who did the final typing;

Robert Shapiro, KGS research assistant, who helped with the graphics; and

Melvin Kleinschmidt, KGS head driller, who drilled and installed the monitor wells used in this study.

Finally, special thanks go to my wife Laura Kukuk and my son Kyle Kukuk for their understanding and support of my educational pursuits. Their continual encouragement and many sacrifices are greatly appreciated.

TABLE OF CONTENTS

Page

Chapter

TABLE OF CONTENTS (continued)

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1: INTRODUCTION

BACKGROUND

Municipal sanitary landfills are often cited as sources of ground-water contamination. The regulations promulgated on May 19, 1980, by the United States Environmental Protection Agency (USEPA) under the Resource Conservation and Recovery Act (RCRA) are intended to minimize ground-water contamination due to landfill leachate. This is accomplished by regulation of landfill-construction practices and establishment of onsite monitor-well networks. These networks are specified to have one upgradient and three downgradient monitor wells installed in the uppermost aquifer adjacent to the land-disposal facility. The ultimate goal is to minimize the environmental impacts of landfills. A major environmental impact resulting from the disposal of solid waste into a sanitary landfill occurs when water passes through the solid waste, accumulates various contaminants, and percolates or leaches into underlying ground waters or downgradient surface waters, seriously degrading water quality. Overall, the effectiveness of the regulations depends on siting considerations as well as on design and management procedures.

Potential pollution from a sanitary landfill must be considered in its design and location. Understanding the factors and conditions involved in the production of leachate is necessary for its elimination and control. Because of the high cost of cleaning up contaminated ground water, it is important to be able to predict the extent of a leachate problem and to attempt to prevent its occurrence. A major goal in planning, designing, and operating a sanitary landfill should be to minimize all sources of water reaching the wastes so that leachate production is eliminated. Leachate production can be minimized or nearly eliminated by preventing water contact with the wastes by the use of surface and subsurface drainage and properly selected cover material that is graded and seeded with a high-evapotranspiration crop.

SCOPE AND OBJECTIVES

Presented here are the results of a site investigation of the Barton County sanitary landfill near Great Bend, Kansas. The investigation was completed by the Kansas Geological Survey (KGS), with technical assistance provided by the Kansas Department of Health and Environment (KDHE), and in cooperation with the Barton County Engineers Office. The primary purpose of this study was to give an initial indication of possible leachate formation and the effects any such leachate might have on the local water quality. This includes the local drinking-water aquifer as well as long-term water-quality effects on the Cheyenne Bottoms Waterfowl Refuge, a downgradient wetland.

RESEARCH APPROACH

To begin the study, an initial site visit was made to determine soil types, site drainage, vegetation, waste-cell construction, and other physical factors that could affect the generation and migration of leachate at the landfill. Also, water levels were recorded from the available domestic and stock wells near the landfill, which provided data for the construction of a general potentiometric-surface map of the drinking-water aquifer underlying the landfill. All available well logs were examined to better define the geology of the site and the depths of possible water-bearing zones.

After reviewing the well logs it was decided to drill three monitor wells into the drinking-water aquifer and three monitor wells into a shallow zone that had water-bearing potential. One shallow monitor well was placed topographically downgradient in a clay zone.

Since this study was undertaken with the intention of giving preliminary indications of possible water-resource contamination by the landfill, a better understanding of the characteristics of municipal solid waste leachate was needed. To achieve this, a literature review was performed and is presented in chapter 2. Based on a knowledge of the types of chemicals typically present in leachate and the analytical capabilities that were available, the water samples were analyzed for volatile organic chemicals (VOCs), dissolved organic carbon (DOC), and the major inorganic anions and cations. Results from these analyses are shown and discussed in chapter 6.

Soil samples were collected from the landfill's cover soil as well as the underlying Dakota clay. Vertical hydraulic conductivity (K) values were determined for these samples. K-values for the cover soil govern the rate at which water can infiltrate the landfill surface. K-values for the underlying clays govern the rate and direction of leachate movement out of the landfill.

In an effort to quantify leachate generation at the landfill, a soil-moisture computer model was used to estimate leachate quantities that the landfill could theoretically be capable of yielding during above-average, below-average and average precipitation water years. The results of the model are discussed in chapter 7. Finally, possible leachate-migration pathways are analyzed and discussed in chapter 8.

CHAPTER 2: MUNICIPAL SOLID WASTES AND LANDFILL LEACHATES

COMPOSITION OF SOLID WASTE

The composition of solid waste in a landfill plays a vital role in determining the chemical characteristics of leachate. The typical composition of municipal solid wastes and their normal range of variation is given in table 1. The major components of solid waste include: food wastes, paper, cardboard, plastics, textiles, rubber, leather, garden trimmings, wood, glass, tin cans, nonferrous metals, ferrous metals, dirt, ashes and brick. The degradation of each of the various components of solid waste imparts different chemical constituents to the leachate. Microbial growth and the accompanying biochemical breakdown of solid wastes are responsible for many of the components of leachate. As water enters the landfill and the solid wastes become saturated, the soluble material initially present in the solid wastes will dissolve and microorganisms will begin to grow on the moist wastes. Soluble organic matter, nutrients, dissolved minerals, and other constituents are then released into the water as by-products of the microbial metabolism of waste components, or as a result of the alteration of the chemical environment (e.g., pH, oxygen concentration, etc.) by the microbes.

In addition to the major constituents of municipal solid waste described above, there are some components that are generally small in volume but which may be highly significant sources of pollution. These are such things as batteries (mercury, lead, and cadmium), waste solvents, and common household products including drain cleaners, paints and paint thinners, automotive oil and antifreeze, pesticides, and oven cleaners. These so called "empty" product containers are regularly

Table 1. Composition of solid wastes.

Component	Percentage by weight	
	Range	Typical
Food wastes	6–26	15
Paper	25–45	40
Cardboard	3–15	4
Plastics	2–8	3
Textiles	0–4	2
Rubber	0–2	0.5
Leather	0–2	0.5
Garden trimmings	0–20	12
Wood	1–4	2
Glass	4–16	8
Tin cans	2–8	6
Nonferrous metals	0–1	1
Ferrous metals	1–4	2
Dirt, ashes, brick	0–10	4

Adapted from Tchobonaglous et al., 1977.

disposed of at sanitary landfills where the residues can find their way into the soil and water systems. The product containers can release small amounts of toxic chemicals into the leachate. If the contaminants are not adsorbed by the soil, they can enter the ground water, where they are of concern, even at trace levels.

Principal mechanisms by which the solid-waste mass is transferred to percolating water can be divided into three major categories. One major mechanism is the entrainment of solid-waste particulate and soluble material by the percolating water. The other two major mechanisms are the dissolution of soluble salts in the wastes and the biodegradation of organic material to gaseous and soluble forms.

BIOLOGICAL DEGRADATION OF SOLID WASTES

Municipal solid-waste stabilization in a sanitary landfill can be separated into two major biological stages: 1) an aerobic degradation phase; and 2) an anaerobic degradation phase, which develops once the oxygen originally present in the landfill is consumed. The large amount of organic matter in solid wastes allows biodegradation to proceed. Garbage, which consists of food wastes and garden trimmings, makes up approximately 27% of municipal solid waste. It is primarily organic in nature and is generally the first component of solid waste to undergo biodegradation.

Aerobic Degradation

Aerobic degradation of the solid wastes will occur first in a sanitary landfill. Bacteria will begin to grow on the surface of the biodegradable fractions of the wastes and start metabolism by hydrolyzing complex organic structures to simple, soluble molecules.

Leachate produced during the aerobic phase also is characterized by the dissolution of highly soluble salts initially present in the landfill. The leachate formed during this initial phase is most likely a result of moisture that was squeezed out of the wastes during compaction and cell construction.

Anaerobic Degradation

The aerobic degradation phase is generally short because of the high biochemical oxygen demand (BOD) of the solid wastes and limited amount of oxygen present in a sanitary landfill. Once the oxygen is exhausted, the microorganisms cannot completely metabolize the organic matter and begin to break it down to organic acids which are readily soluble in water. Thus, soluble organic acids begin to accumulate in the landfill. The microorganisms involved in these processes obtain energy for growth from the chemical reactions that occur during metabolism and a portion of the organic waste is converted into cellular or exocellular material (Crawford and Smith, 1985).

As the initial anaerobic biodegradation processes occur, the organic-acid accumulation yields a low pH leachate and considerable concentrations of inorganic ions (e.g., Cl, SO_4, Ca, Mg, Na). The increase in cation and anion concentrations probably results from the leaching of readily solubilized materials including those originally available in the solid waste and those made available by biodegradation of organic matter.

The second stage of anaerobic biodegradation is characterized by methane fermentation by methanogenic bacteria. The anaerobic conditions and the soluble organic acids create an environment where the methane bacteria can grow. The methanogenic bacteria utilize the end products from the first stage of anaerobic degradation to yield methane and carbon dioxide. Methane fermentation generally begins within one year following solid-waste placement (Walsh and Kinman, 1979). The methanogenic bacteria prefer a relatively neutral pH (6.6 to 7.4) and do not like acid conditions. The acid formation in the first stage tends to lower the pH and if acid formation is excessive, the activity of the methanogenic bacteria can be inhibited.

The methane and carbon dioxide produced by the bacteria tend to come out of solution and rise to the surface of the landfill. This gas production can be considered as a dangerous problem, a desirable asset, or an inconsequential matter, depending on the philosophy on which the landfill is managed. In some cases where low-permeability clay covers have been placed over the surface of landfills, the gases have been forced to move laterally through the soil instead of migrating vertically to escape into the atmosphere at the land surface. On numerous occasions dangerous levels of flammable and asphyxiating methane gas have accumulated in the basements of buildings that are located close to landfills. On the other hand, if a sanitary landfill is designed with the intent to collect the methane gas, the gas can be captured, concentrated, and used as a source of energy.

LEACHATE GENERATION

In general, it has been found that the quantity of leachate is a direct function of the amount of external water entering the landfill. In fact, if a landfill is constructed properly, the production of measurable quantities of leachate can be eliminated (Tchobanoglous et al., 1977).

Initially the solid waste acts as a sponge, absorbing water until it becomes saturated. When the solid waste is 40-60% saturated, bacterial growth and solid-waste degradation will begin (Metzler, 1981). As additional water enters the landfill, the zone of saturation will move downward through the solid wastes. The migrating zone of saturation will transport dissolved substances and the byproducts of microbial metabolism as well as to flush out relatively insoluble contaminants that can be carried along with the saturated zone. When sufficient water enters the landfill to saturate the entire thickness of the waste cells, water movement will continue downward, out of the landfill towards the ground water. However, if precipitation is low, the zone of saturation will oscillate with the occurence of precipitation events. During extended dry periods, the zone of saturation will move upward due to direct evaporation and evapotranspiration at the landfill surface. Solubilization and biochemical stabilization will continue in the saturated zones, but there will be no flow of leachate out of the landfill. During wet periods, the zone of saturation will move downward as precipitation enters the landfill. The duration and intensity of the wet period will determine how far the zone of saturation will migrate towards the bottom of the landfill before the next dry period begins.

LEACHATE MIGRATION

Once leachate leaves the landfill, it will migrate towards the ground water. The transport rate and direction through the soil zone will depend on the porosity and permeability of the landfill soils. In general, leachate will follow the path of least resistance. Leachate will tend to flow laterally around the lower permeability soils such as clay and vertically through the higher permeability soils. Clays have the greatest adsorptive capacity of the soil types. Clays generally carry a net negative charge, thus they will attract positively charged ions and remove them from the migrating leachate solution. Negatively charged ions such as chloride will pass through the clays relatively unaffected. Thus, chloride can be used as a possible tracer to delineate the extent to which the leachate plume has spread in the ground water.

When leachate reaches an aquifer, it will travel in the direction of ground-water flow. Leachate generally remains in a relatively discrete zone within the ground water and forms an elongated plume in the direction of ground-water flow. Considering the large areal extent of most landfills, the time span elapsed for possible contamination, and the retardation effects due to attenuation and geochemical reactions with aquifer materials, most subsurface leachate contamination cannot be attributed to a single plume. Rather, the contaminated zone must be regarded as a multiplicity of plumes, superimposed in three dimensions.

LEACHATE CHARACTERISTICS AND POTENTIAL ENVIRONMENTAL EFFECTS

The chemical characteristics of sanitary landfill leachate vary widely due to different waste characteristics, climatic conditions, and the physical composition of the wastes (i.e. bailed or shredded wastes versus unprocessed wastes), etc. This large variance can be seen in the values of table 2. Also, co-disposal of liquid wastes and sewage sludges with solid wastes can significantly alter leachate quantities and characteristics.

When leachate is released by a landfill, underlying ground waters and downgradient surface waters can be adversely affected. If leachate percolates to a drinking-water aquifer in high enough concentrations, it

Table 2. Typical leachate characteristics.

Constituent	Concentration Range (mg/L, except pH)
BOD_5 (5-day biochemical oxygen demand)	2,000-30,000
TOC (total organic carbon)	1,500-20,000
TDOC (total dissolved organic carbon)	200-30,000
COD (chemical oxygen demand)	3,000-45,000
Total suspended solids	200-1,000
Total dissolved solids	5,000-40,000
Organic nitrogen	10-600
Ammonia nitrogen	10-800
Nitrate	5-40
Total phosphorus	1-70
Ortho phosphorus	1-50
Alkalinity as $CaCO_3$	1,000-10,000
pH	5.3-8.5
Total hardness as $CaCO_3$	300-10,000
Calcium	200-3,000
Magnesium	50-1,500
Potassium	200-2,000
Sodium	200-2,000
Chloride	100-3,000
Sulfate	100-1,500
Total iron	50-600

Sources: Freeze and Cherry, 1979; Tchobanoglous et al., 1977.

can render the aquifer water useless for further human consumption. This can have very serious implications for the local population, especially if ground water is their only available source of water.

In cases where leachate-contaminated ground water flows into a surface-water body, water quality can be degraded to a point where normal use of the water body will have to be discontinued. Fish and other biological life forms might be killed or have their growth retarded due to high concentrations of leachate. If readily biodegradable organics are present in the incoming leachate, oxygen depletion of the surface-water body will occur. Some bottom-feeding fish, such as carp and catfish, have a tendency to concentrate contaminants (e.g., heavy metals and synthetic organic chemicals) within their body tissue and may become inedible. Also, if a leachate-contaminated water body is used as water supply for stock animals or waterfowl, these uses also might have to be discontinued.

As previously discussed, when leachate reaches a drinking-water supply, many undesirable effects can occur. For example, inorganic constituents such as salts can impart a displeasing taste to drinking water. As can be seen in table 2, high concentrations of salts are common in municipal landfill leachates. Another inorganic constituent of special concern is nitrate. The presence of an excessive concentration of nitrate in drinking water may result in cyanosis (blue babies) of infants to whom the water is being fed. Water containing more than 45 mg/L of nitrate (as NO_3) should not be used in formula preparation or as drinking water for babies under 3 months of age (Hem, 1985).

Other contaminants such as trace metals, waste solvents, etc., are not shown in table 2, but can pose potentially adverse environmental problems if they find their way into a drinking-water supply. Trace metals have been found to be toxic to fish and human beings above threshold concentrations. Waste solvents have become a human health concern as there is a potential health risk associated with their long-term consumption. Although such contaminants represent only a small fraction of the leachate composition, they can have potentially significant impacts on local water quality. Consequently, they are being looked at carefully by regulatory agencies.

CHAPTER 3: SITE DESCRIPTION

STUDY LOCATION

The predominant industries in Barton County are agriculture and petroleum. The county has approximately 32,000 residents and is located in south-central Kansas. For the past 15 years, residential, commercial, and industrial refuse has been disposed of at the county landfill. The landfill site is composed of about 240 acres and is situated on the Cheyenne Bottoms- Arkansas River valley divide in the eastern half of sec. 12, T. 19 S., R. 13 W (figure 1).

The landfill site is positioned in a ravine on the north slope of the divide that drains directly towards the Cheyenne Bottoms. The ravine represents a local watershed that exists inside the landfill's property line. The Cheyenne Bottoms-Arkansas River valley divide is a ridge of Dakota Formation covered with wind-deposited loess that separates the Arkansas River valley from the Cheyenne Bottoms. The divide is from 1 to 3 miles wide and extends roughly from U.S. Highway 281, north of Great Bend, to a point northwest of Ellinwood, Kansas.

HISTORY OF THE BARTON COUNTY LANDFILL

The Barton County sanitary landfill began operation on January 1, 1972. It was one of the first sanitary landfills to begin operation in the state of Kansas. According to the county engineer, the landfill receives approximately 90 tons of solid waste per day for approximately 302 days each year. Residential, commercial, industrial, and some permitted hazardous wastes such as pesticides, acids, caustics, asbestos, and solvents have been disposed of at the landfill over its 15-year history.

The waste-disposal operation began in the southernmost portion of

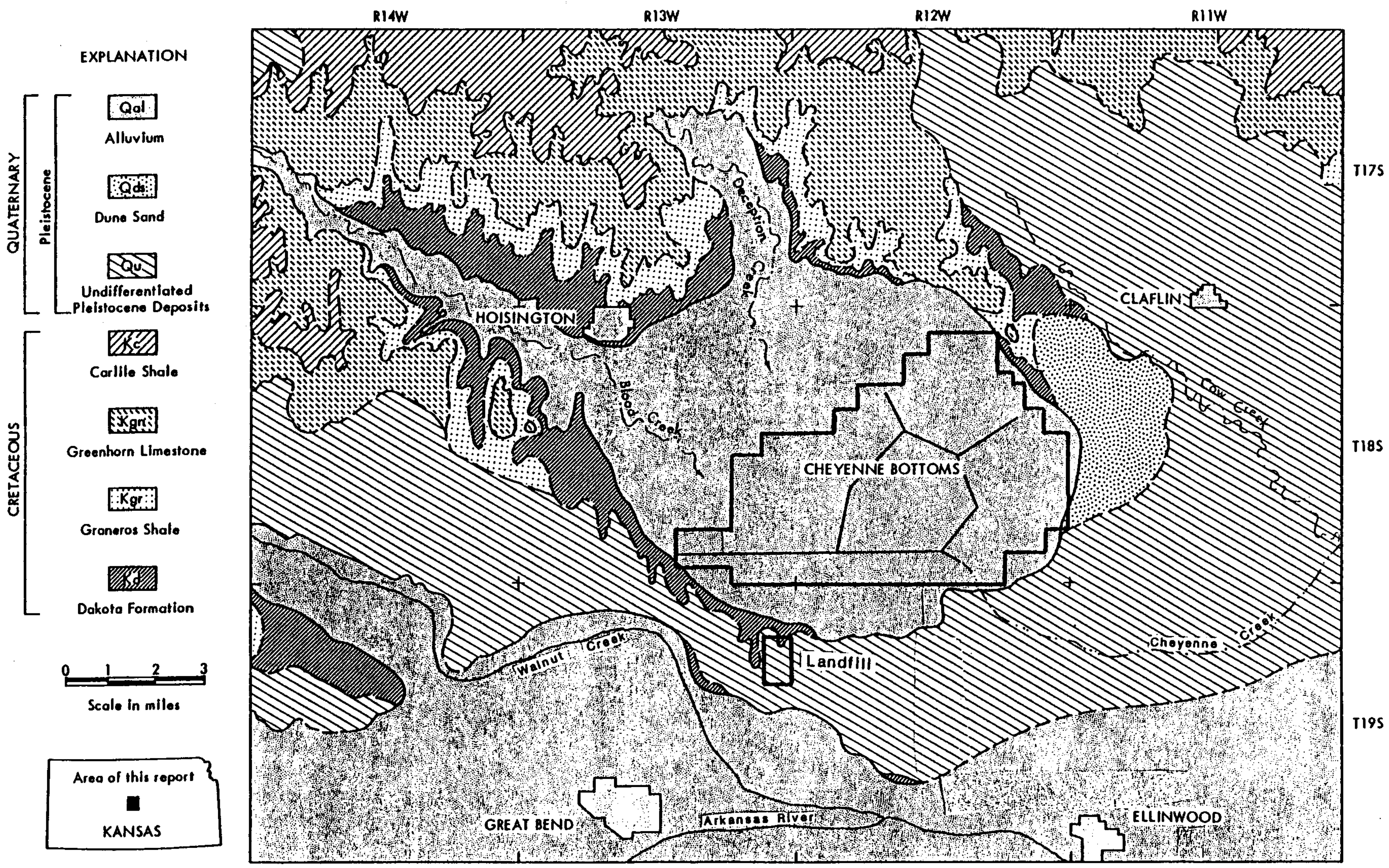

(Areal Geology from Latta, 1950)

Figure 1. Areal geology of Cheyenne Bottoms and Barton County, Kansas.

the ravine and proceeded northward as the ravine was filled (figure 2). The design of the landfill has employed the use of trenches excavated approximately 20-40 feet into the loess. The wastes bear directly on the loess. A typical trench would have dimensions of 30 feet in width and 200-300 feet in length. Once a trench is excavated, approximately 6 feet of compacted solid waste is accumulated followed by a 6-inch layer of loess to cover the waste cell. A period of three to four weeks is usually required before the 6-foot thickness of solid waste can be attained. The creation of waste cells is continued until the top of the trench is 3 feet below the desired final land-surface elevation.

Following the completion of a solid-waste trench, a 3-foot mantle of soil consisting of 2 feet of loess and 1 foot of topsoil is compacted on top of the uppermost cell. The 3-foot mantle is then sloped to provide surface drainage from the site northward. It was observed that proper surface drainage was not consistently achieved over the completed areas of the landfill. Many small depressions in the soil mantle were noted, which allowed water to accumulate after a precipitation event. These depressions are suspected to be caused by differential settlement of the decomposing solid waste, perhaps coupled with inadequate sloping of the soil mantle.

When precipitation occurs, solid waste is deposited in what has been termed as the landfill's "wet pit." The "wet pit" is a north-south-trending open trench that is approximately 100 feet wide, 700 feet long, and 40 feet deep. The "wet pit" is located due north of the scale house near the entrance of the landfill in order to reduce the travel distance of vehicles manuevering over muddy roads.

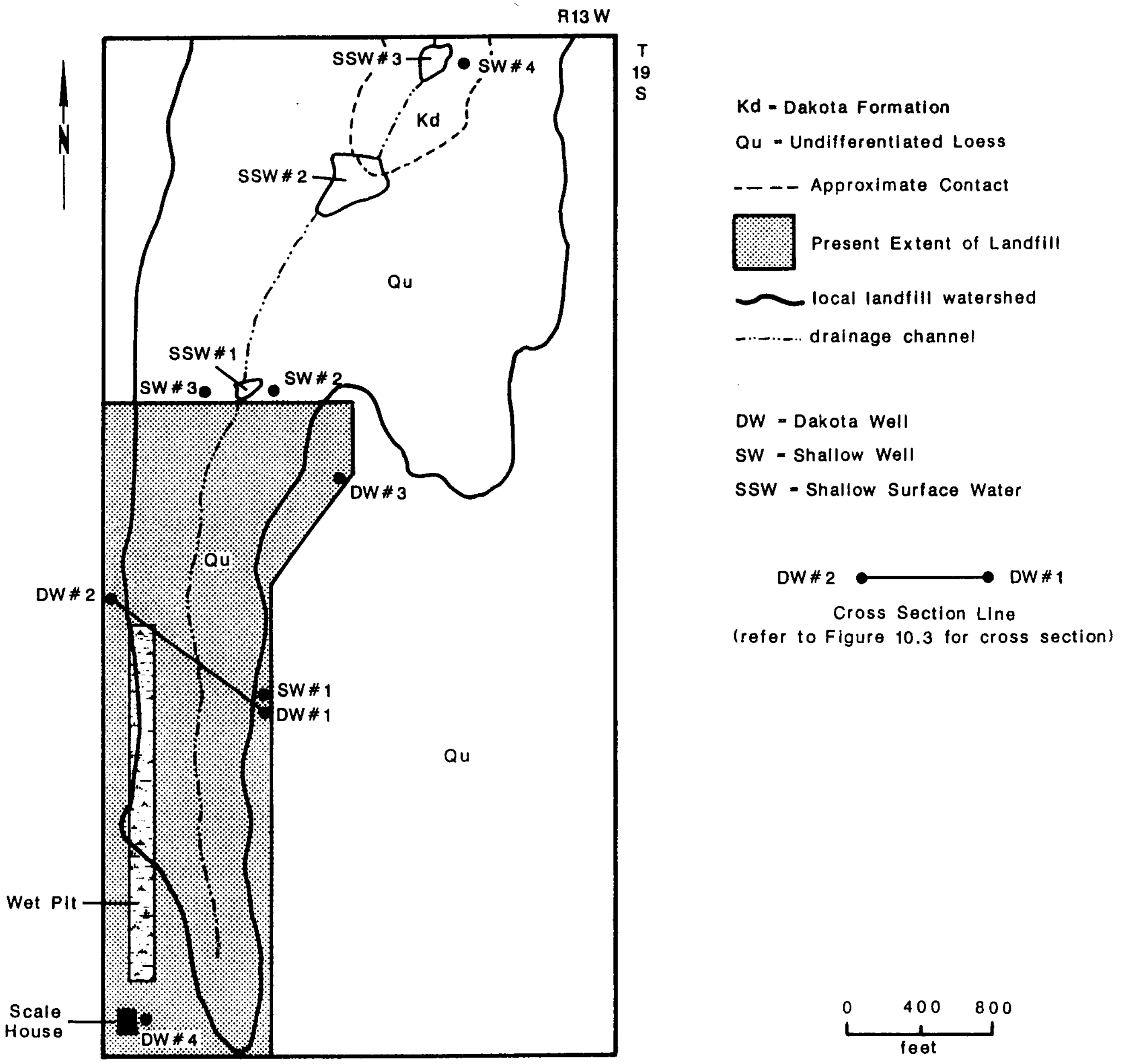


Figure 2. Areal view of the Barton County landfill.

Solid waste accumulated in the "wet pit" is covered with soil once or twice during the year and is open to the atmosphere the remainder of the time. The "wet pit" is a cause of concern for officials of the Kansas Department of Health and Environment because the pit can easily accumulate several feet of standing water in its lowest point. This water is in direct contact with the wastes in the pit which can result in direct leaching from the wastes.

The Barton County sanitary landfill opened eight years prior to the RCRA legislation. Therefore, as was the case with the majority of landfills in operation before RCRA, no monitor wells were installed prior to the opening of the site. Additionally, no soil borings were completed at the site to determine geologic-unit thicknesses and the depth to water. However, geologic information from the numerous oil-well logs in the area were used to determine the suitability of the site for the location of the landfill.

SITE GEOLOGY

Geologic formations at the site consist of a lower bedrock unit, the Cretaceous Dakota formation, which is overlain by Pleistocene loess. The loess is late Pleistocene in age and is composed chiefly of loess or loess-like silt, with some sand and gravel deposits near the bottom of the formation. The silt is tan, yellow tan, brown, and gray in color and contains some fine sand, clay, and caliche nodules. Fragments of "algal limestone," Cretaceous sandstone, and "ironstone" are common at the base of the formation (Latta, 1950).

Approximately 90% of the landfill-site land surface is underlain by the loess which ranges in thickness from several inches near SSW #2 and thickens to more than 60 feet at the southern boundary of the landfill. The loess unconformably overlies the Dakota Formation (figure 3), which is 200-300 feet thick in this area.

The Dakota Formation is Cretaceous in age and is composed of alternating beds of vari-colored clay, shale, siltstone, and sandstone. Clay is the dominant type of rock in the Dakota Formation and is light to dark gray, white, tan, brown, yellow, or red in color. White, gray, yellow, tan, and brown fine- to coarse-grained sandstone occurs as thin beds in the clay and as lenses ranging from a few feet to more than 30 feet in thickness. The sandstone lenses occur as discontinuous bodies that may be encountered in any part of the formation (Latta, 1950). The Dakota Formation crops out in the north end of the ravine between SSW #2 and SSW #3, where the loess has been removed by erosion (figure 2).

SITE GEOLOGIC HISTORY

Cretaceous deposits were laid down on a long-eroded, deeply weathered Permian surface. During the Early Cretaceous, the sea was transgressing northward over the area and deposited the Cheyenne Sandstone on the shoreline of the transgressing sea. Marine sediments of the Kiowa Shale followed (figure 4 and figure 5). A general regression of the sea marked the end of the Early Cretaceous and the beginning of the deposition of the Dakota Formation. This regression of the sea was not continuous but was marked by minor fluctuations of the shoreline resulting in marine and continental beds being interbedded (Latta, 1950).

The Dakota Formation generally is considered to have been deposited under marine conditions from this regressing sea and under nonmarine conditions in a bordering low-lying coastal and deltaic plain (Keene and

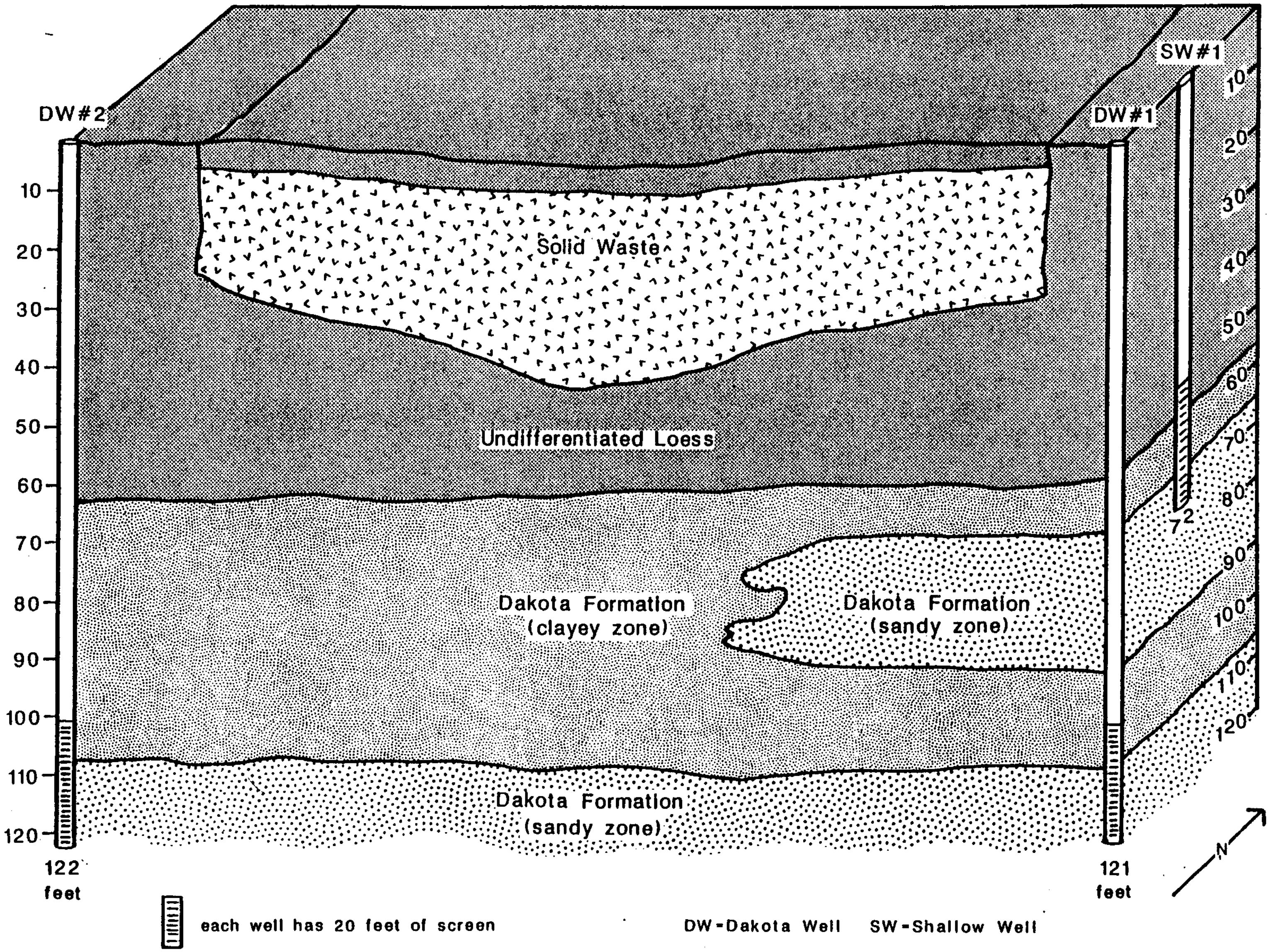


Figure 3. Geologic cross section from DW#2 to DW#1.

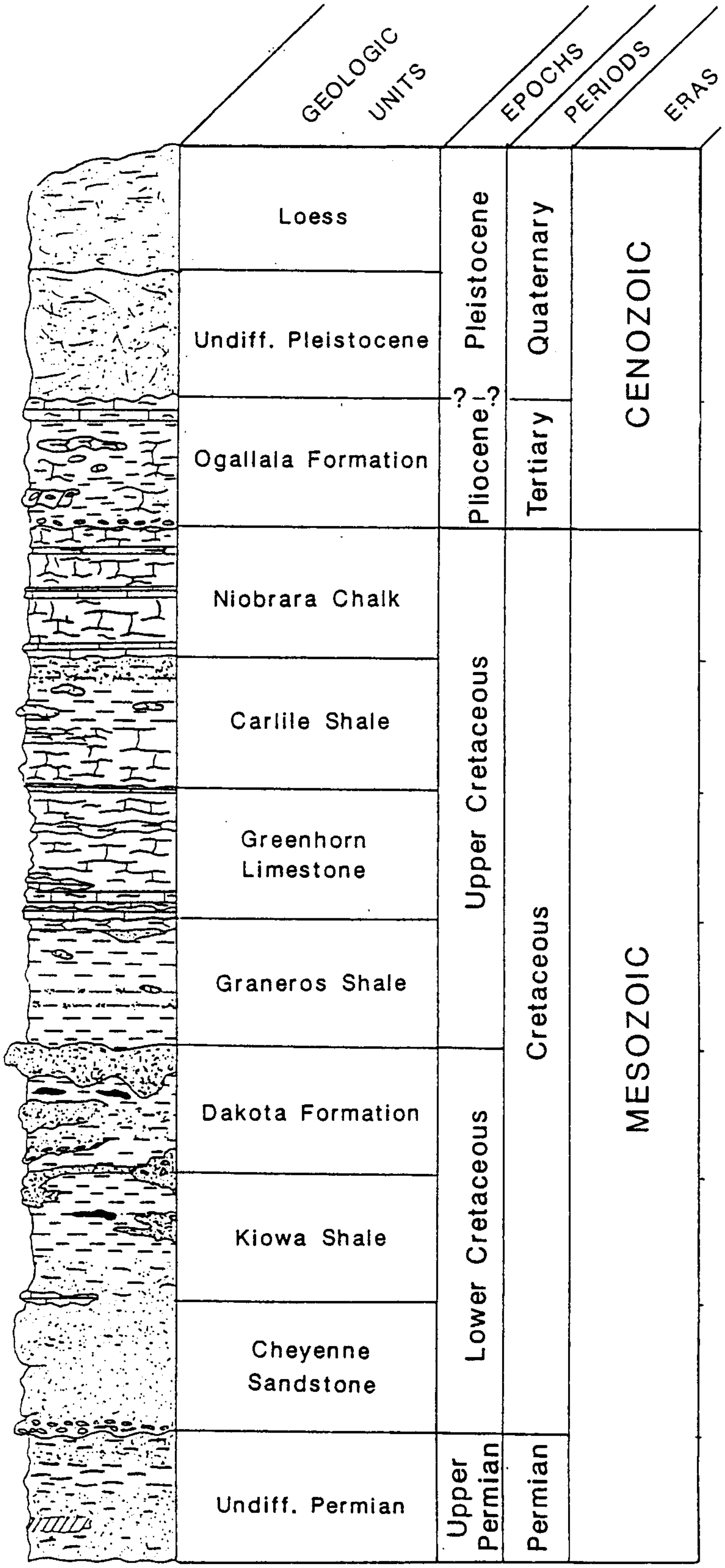


Figure 4. Geologic column.

Qu	Undifferentiated deposits of eolian, fluvial, and colluvial deposits of mostly silt, occasionally overlying sand and gravel.
To	Ogallala Formation: Crossbedded, generally arkosic gravel, sand, and silt, locally cemented with calcium carbonate. Containing limestone, volcanic ash, opaline sandstone, and bentonitic clay.
Kn	Niobrara Chalk: Interbedded, soft, calcareous shale, chalk, and chalky limestone.
Kc	Carlile Shale: Chalky shale and thin beds of chalky limestone.
Kgh	Greenhorn Limestone: Chalky shale alternating with beds of hard chalk; contains thin beds of hard crystalline limestone.
Kgs	Graneros Shale: Noncalcareous shale containing selenite, pyrite, and thin beds of fine-grained sandstone.
Kd	Dakota Formation: Varicolored clay, shale, siltstone, occurring in alternating beds of ironstone, and sandstone.
Kk	Kiowa Formation: Light-gray to black shale and sandy shale, with beds and lenses of medium- to fine-grained gypsum.
Kkc	Cheyenne Sandstone: Sandstone, very fine to fine to medium-grained siltstone, with some clay and shale.
P	Undifferentiated red beds consisting of red siltstone, shale, sandstone, with lesser amounts of gypsum, salt, anhydrite, limestone, and dolomite.

Figure 5. Description of geologic units.

Bayne, 1977). At the beginning of Late Cretaceous time, the area was again submerged beneath a shallow sea and many hundreds of feet of shale and limestone (Graneros Shale, Greenhorn Limestone, Carlile Shale, and Niobrara chalk) were deposited.

During most of the Tertiary Period, the surface was exposed and several hundred feet of Late Cretaceous rocks were eroded, hence their absence in the drilling logs. Deposition of sediments of the Ogallala Formation occurred during the Pliocene epoch of the Tertiary. These deposits were thin in this area and were removed by erosion during Pleistocene time. Much of the present topography of this area is the result of erosion which began during the Pleistocene and which has continued to the present time (Bayne, 1977).

The Pleistocene history of this area is very complex and accurate interpretation is difficult. Late in the Pleistocene erosion occurred in the Cheyenne Bottoms Valley that removed the silt, sand, and gravel of the undifferentiated Pleistocene formation which was deposited during the early Pleistocene. These eroded sediments had been originally deposited by streams carrying material from the Rocky Mountains and from areas of Tertiary rocks to the west. This late Pleistocene erosion was followed by stream deposition of silt, sand, and gravel in the valleys and windblown silt on the divides (undifferentiated loess).

SITE WATER RESOURCES

The importance of maintaining suitable ground-water resources in this portion of the state is illustrated by the fact that almost the entire population of Barton County obtains its water supply from wells. In addition to human uses, wells supply water for livestock and several waterfowl and fishing ponds throughout the county.

The water-bearing formations that underlie the Barton County landfill are in the Dakota Formation as well as in a thin, water-bearing zone at the unconformable contact between the loess and the Dakota Formation. This thin zone will be referred to in this paper as the upper aquifer. Currently, the upper aquifer is not known to be used as a water supply in the area.

Dakota Formation

The discontinuous sandstone lenses of the Dakota Formation are the chief sources of water in the upland areas of Barton County, which includes the Barton County landfill area. Approximately one-third of the recorded wells in Barton County during the 1940's obtained their water from the Dakota Formation, thus making it a very important water resource (Latta, 1950). The shallowest sandstone lens that is tapped for domestic use near the landfill occurs at a depth of approximately 105 feet below the landfill surface. Three monitor wells were placed in this lens to provide water-quality information concerning the local drinking-water aquifer.

Water from the Dakota Formation in the vicinity of the landfill is under confined pressure. The Dakota clay layers above the sandstones, and the Kiowa Shale below, act as confining beds. The quantity of water available from the wells is quite variable, with yields ranging from a few gallons per minute to a few hundred gallons per minute (Latta, 1950).

Literature reports indicate that the general flow of ground water in the Dakota Formation is in an east-northeasterly direction (Keene and Bayne, 1977). This is due to recharge of the aquifer in the uplands to the west and the discharge area to the northeast. Figure 6 shows the

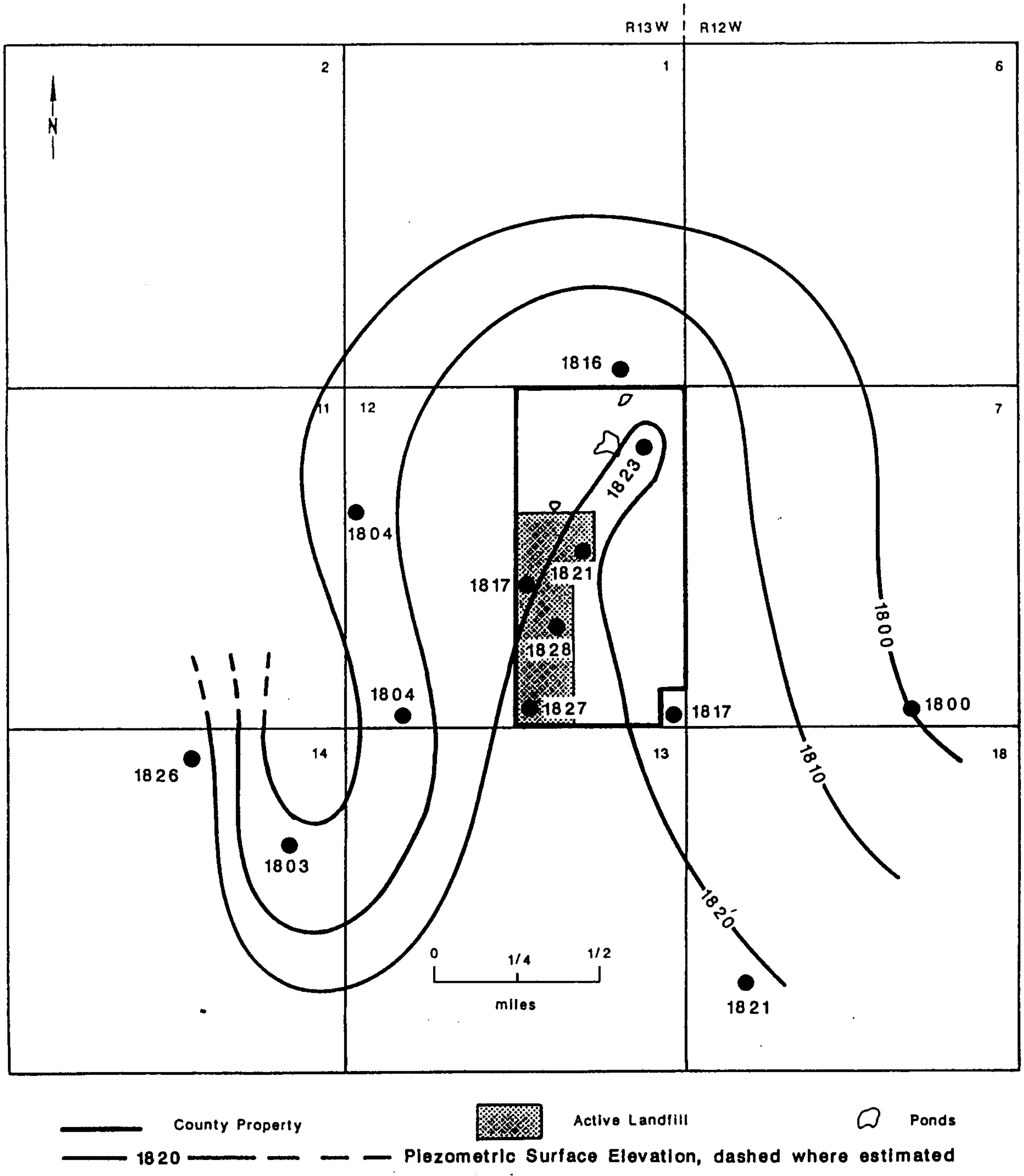


Figure 6. Piezometric surface of the Dakota Formation.

piezometric-head contours of the Dakota Formation in the vicinity of the landfill. The contours are generalized due to the limited number of wells in the study area as well as to the wide range of well-completion depths, screened intervals, and gravel-pack intervals. These factors can all have an effect on water levels inside the well casing. Water-level measurements that were recorded specifically for this study to determine ground-water-flow direction in the Dakota Formation were in general agreement with the ground-water-flow directions indicated by Keene and Bayne (1977).

Upper Aquifer

The upper aquifer is located on an unconformity between the loess and the Dakota Formation. An unconformity is a surface that represents a time interval during which deposition was nonexistent, or the surface of the existing rocks was weathered, eroded, or fractured. Aquifers are commonly associated with unconformities, either in the weathered or fractured zone directly underlying the surface of the buried landscape or in permeable zones in coarse-grained sediments laid down on top of this surface when the system entered a new era of accretion (Freeze and Cherry, 1979). Coarse sand and fine gravels were found while drilling through the upper aquifer (see drilling logs in appendix 1).

The upper aquifer underlies the landfill locally and is much closer in vertical distance to the waste cells than the Dakota aquifer. However, there is not sufficient water-level data from observation wells to construct a water-level contour map of the upper aquifer. Therefore, a contour map of the top of the Dakota Formation was prepared with the assumption that the ground water would flow along the upper surface of the Dakota clay (figure 7). This contour map will be presented later

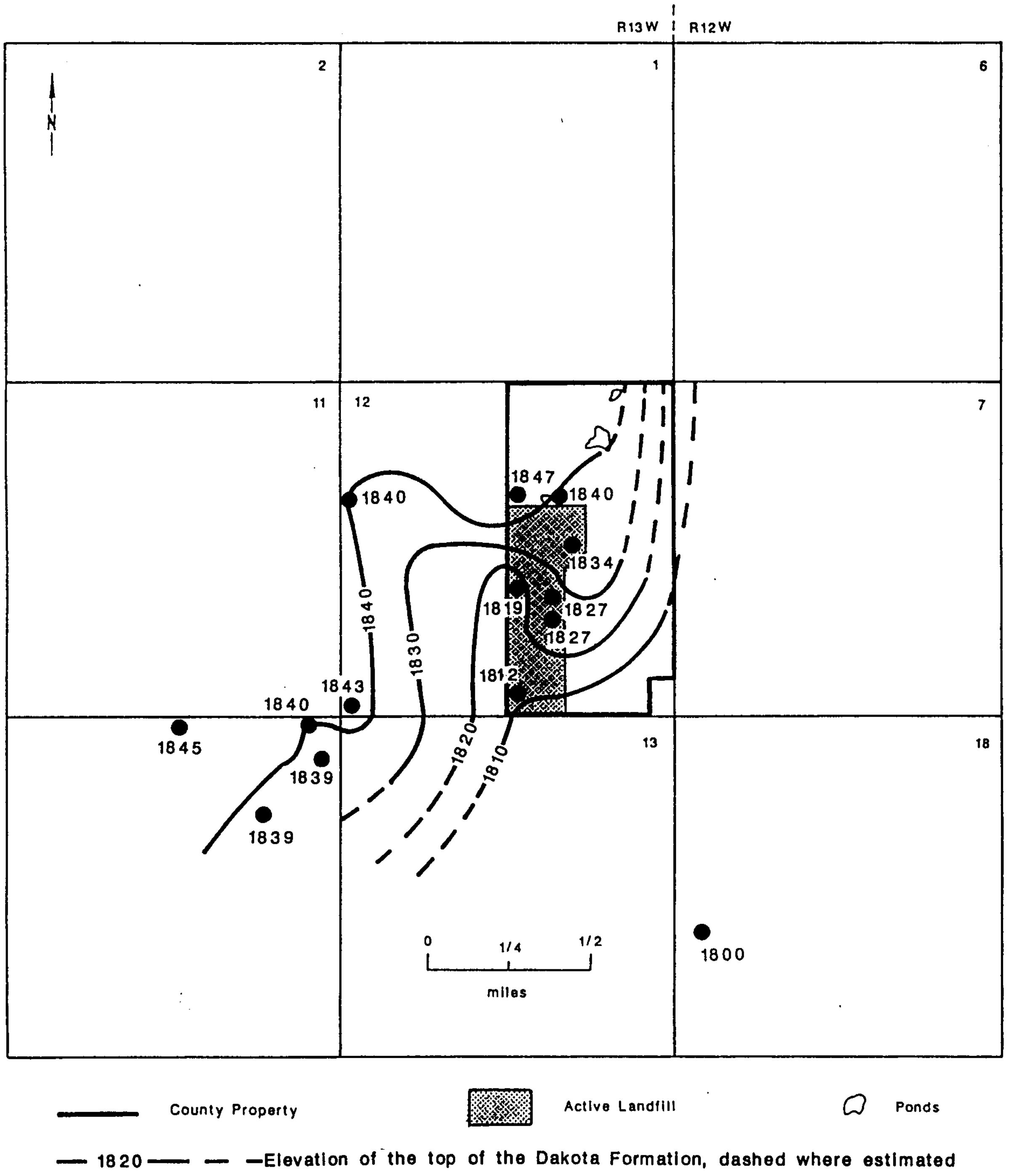


Figure 7. Elevation of the top of the Dakota Formation.

with respect to possible contaminant-migration pathways.

Drilling samples showed that the lower 5-15 feet of the loess contains more coarse sand and fine gravel than the upper part of the formation. The upper 2-6 feet of clay in the underlying Dakota Formation has been weathered and reworked and has a much higher sand and gravel content than the lower Dakota clays. The combination of the porous loess with the porous upper Dakota allows for the accumulation and migration of water in the upper aquifer. This upper aquifer has had very little documentation in the literature because it generally does not yield water in sufficient quantities to be used as a water supply.

The loess-Dakota contact, which the upper aquifer surrounds, crops out near the currently inactive north boundary of the landfill in the lower portions on the ravine. During the course of the field work, no springs or seeps were observed near the contact outcrop. However, no observations were made in this study during or after a prolonged precipitation event.

Surface Waters

Located in the north section are three surface-water ponds connected in series by a drainage channel. The ponds are referred to in this report as SSW #1, SSW #2, and SSW #3, going from south to north. SSW represents shallow surface water as shown in figure 2. These ponds act as holding basins for the surface runoff generated from the active portion of the landfill, as well as for a drinking-water supply for cattle. When the ponds reach their respective water-holding capacities, water flows over a spillway northward to the next pond until, eventually, the water will flow out of the SSW #3 and enter a small drainageway that discharges into the Cheyenne Bottoms.

CHAPTER 4: DESIGN AND CONSTRUCTION OF THE MONITOR-WELL NETWORK

WELL PLACEMENT

The emphasis of this study was to identify if and where ground water was being contaminated by leachate from the landfill. To obtain this data, seven monitor wells, 2 inches in diameter, were installed by the Kansas Geological Survey near the fenced boundaries which surround the active portion of the landfill (figure 2). Of the seven wells installed, three wells (DW #1,2,3) were considered deep wells or Dakota wells and were completed to a depth of approximately 122 feet by a mud rotary-drilling rig. The locations of the wells in reference to the active portion of the landfill are as follows: DW #1 to the east, DW #2 to the west, and DW #3 to the northeast. The wells penetrate approximately 15 feet into a sandstone lens of the Dakota Formation. Machine-slotted well screen was placed over the lower 20 feet of borehole. This sandstone lens represents the shallowest source of drinking water presently being used in the area. The purpose of the Dakota wells was to provide ground-water-quality data on the shallowest drinking-water aquifer that would be affected first by leachate which originated from the landfill.

The remaining four wells are considered to be shallow wells (SW #1,2,3,4), and their purpose was to provide ground-water-quality data for the upper aquifer. SW #1 is 15 feet north of DW #1 and was drilled and constructed in the same manner as the Dakota wells (rotary drilling and 20 feet of well screen). This well provides a control to compare water quality between the upper aquifer and the drinking-water aquifer at the same location. SW #1 is 72 feet in depth and penetrates the upper 6 feet of a 23- foot sandstone lens encountered 6 feet below the loess-Dakota contact. This sandstone lens is typical of sandstones in the Dakota in that it was discontinuous over the site and was not encountered in the other Dakota wells. The 20-foot screened interval allowed ground water from the upper aquifer as well as any water in the upper sandstone lens to be sampled.

SW #2 and #3 are located directly north of the current landfill fence line and are positioned on the east (#2) and west (#3) sides of SSW #1. SW #2 is 26 feet in depth and SW #3 is 27 feet in depth. Both wells penetrate several feet into the top of the Dakota Formation. The screened interval is 10 feet. This allowed the sampling of water obtained exclusively from the upper aquifer. These wells are also topographically downgradient from the waste cells.

SW #4 is located near the north boundary of the landfill just to the east of SSW #3. The well is 20 feet in depth, the entire length being in Dakota clay. The screened interval is 10 feet, but no water has been found in the well. The purpose of this well was to provide downgradient, shallow ground-water data at a reasonable distance from the active landfill, but obviously no data can be attained from a dry well. SW #2, 3, and 4 were drilled with a hollow-stem auger.

The areal placement of the wells was chosen with respect to the amount of drilling time available (two weeks) balanced with an attempt to ensure sufficient coverage on all sides of the landfill with an emphasis on downgradient wells. Safety precautions were taken during the actual drilling of the monitor wells to ensure the safety of all personnel working in direct contact with the water from the wells. All personnel who handled the water or handled the equipment that came in contact with the water from the wells wore rubber gloves in the event the water was contaminated. Also, a GasTechtor Model 3220S was used to monitor air quality above the well head. As described earlier, methane-gas production is a common byproduct of the decomposition of solid wastes. No gases were detected by the GasTechtor.

WELL CONSTRUCTION

Materials used in construction of the monitor wells were provided by Barton County and included 2-inch-diameter, Schedule-80, Polyvinyl Chloride (PVC) threaded well casing and screen, gravel pack, fine sand, cement, slip caps, bailers, and poly rope. Miscellaneous items were provided as needed by the Kansas Geological Survey. The gravel pack and fine sand were supplied by the Stone Sand Company of Great Bend. Two loads of the gravel pack and one load of fine sand were rejected upon arrival at the landfill due to contamination by diesel fuel and asphalt being dispersed throughout the sand matrix. The truck that had hauled the sand had previously contained asphalt; diesel fuel had been used to clean out the truck bed prior to hauling the gravel pack and fine sand to the landfill. Following the rejection of the contaminated sand, a clean truck was dispatched that delivered clean sand for use in the monitor wells.

The seven wells were constructed in accordance with KDHE requirements (see figure 8 and appendices 3 and 4). Deviations from standard monitor-well construction which were approved by KDHE included: the screened interval did not extend over the entire aquifer thickness; 2-inch-diameter casing was used instead of 4 inch; drilling rig and equipment were not decontaminated between each well; boreholes were grouted beginning approximately 10 feet above the well screen and continuing to land surface; very fine sand was used between the gravel

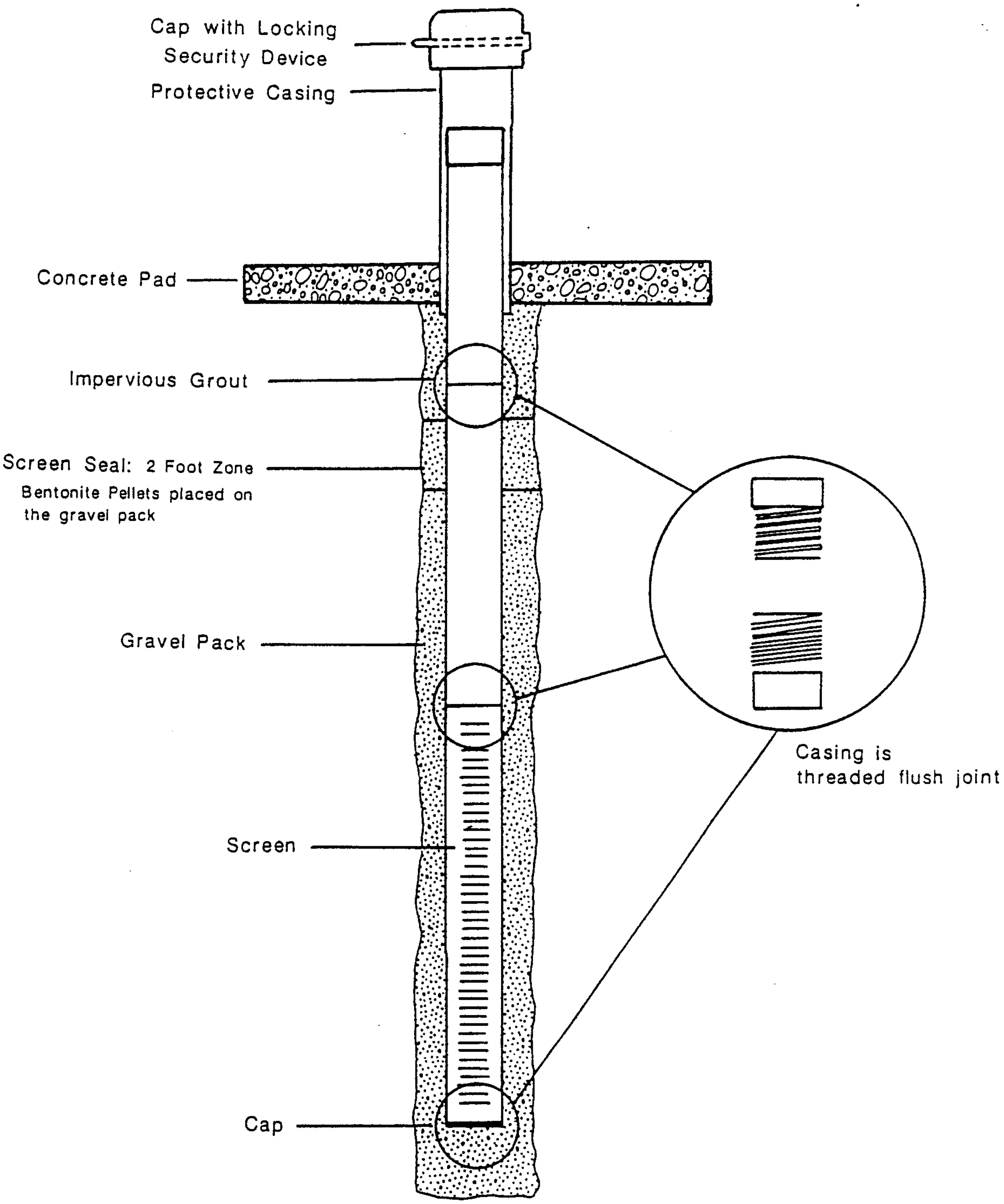


Figure 8. Standard monitoring well design.

(From KDHE 1986)

pack and bentonite seal to prevent encroachment of the grout into the gravel pack. When grout invades the upper portion of the gravel pack the result often is an erroneously high pH when water samples are analyzed. The gravel pack, bentonite pellets, fine sand, and grout were placed beginning at the bottom of the borehole and continuing upwards using 1/2-inch PVC tremie pipe. This procedure was employed in the three Dakota wells as well as SW #1. SW #2, #3, and #4 were shallow enough to allow materials to be poured in from the land surface.

The importance of a proper grout seal is to prevent migration of water or contaminants down the borehole to ensure the isolation of the particular aquifer of interest. Grout was allowed to cure for approximately 24 hours before development was started. This lag time was included to further reduce the chance of uncured grout being pulled into the gravel pack by the sudden evacuation of water from the well.

Temperature logs were run on DW #1 seven hours after grout placement and DW #3, 14 hours after grout placement. This procedure was undertaken to verify that the grout placed down the hole actually was resting on top of the gravel pack and was not lodged higher up in the annulus of the borehole. The temperature probe was able to detect the depth of the grout seal because the heat of hydration of curing grout was sufficient to produce measurable warming inside the well casing.

To prevent cross contamination between wells during the development and sampling procedures, an individual bailer was dedicated to each well. Conventional bailers have been shown to produce little sample alteration (Houghton and Berger, 1984). The six monitor wells that produced water were extensively developed with the bailers immediately after the grout cured. Developing involves the removal of well water in order to clear the well of drill cuttings and mud. Extraneous matter could affect the chemical analysis of the water. To quantify how well the well water was cleaned during development, water samples were collected from DW #2 and DW #3 prior to and after development. Clays in these samples were analyzed by the Kansas Geological Survey using X-ray diffraction to determine a general ratio between the quantity of drilling mud (bentonite clay) and the quantity of Dakota clay present before and after development. The results showed a significant reduction in the amount of bentonite left in the post-development water sample.

CHAPTER 5: SAMPLING PROCEDURES AND ANALYTICAL METHODS

A total of 17 water-sample locations were used during this initial investigation of the Barton County landfill (figure 9). Of the 17 sample locations, six samples were from the monitor wells installed by the KGS, eight from local domestic wells, and three from the downgradient surface-water ponds. The samples were collected on July 8 and 9 of 1986, and were kept on ice until they were transferred to their respective analytical locations on July 10. The samples were analyzed for volatile organic chemicals (VOCs) using the gas chromatograph/mass spectrophotometer (GC/MS) in the environmental-organic laboratory of the KDHE. Dissolved organic carbon (DOC) analyses were performed by the author using a Xertex/Dohrmann Total Organic Carbon Analyzer (Model DC-80) in the Environmental Health Science and Engineering Laboratory of the University of Kansas. The inorganic analyses were performed in the geochemistry laboratory of the KGS.

The procedure for collecting ground-water samples was as follows:

- Water levels were recorded in each well prior to sampling.
- At least three casing volumes were extracted from the wells using either a bailer or a Bennett Sample Pump to remove the stagnant water inside the well. Domestic wells were evacuated by removing 50 gallons of water through outdoor spigots.
- Immediately following evacuation of the well casing, samples were collected for laboratory analysis and for field tests (temperature, pH, and specific conductivity). Samples were obtained using the same method that was employed to evacuate the wells.

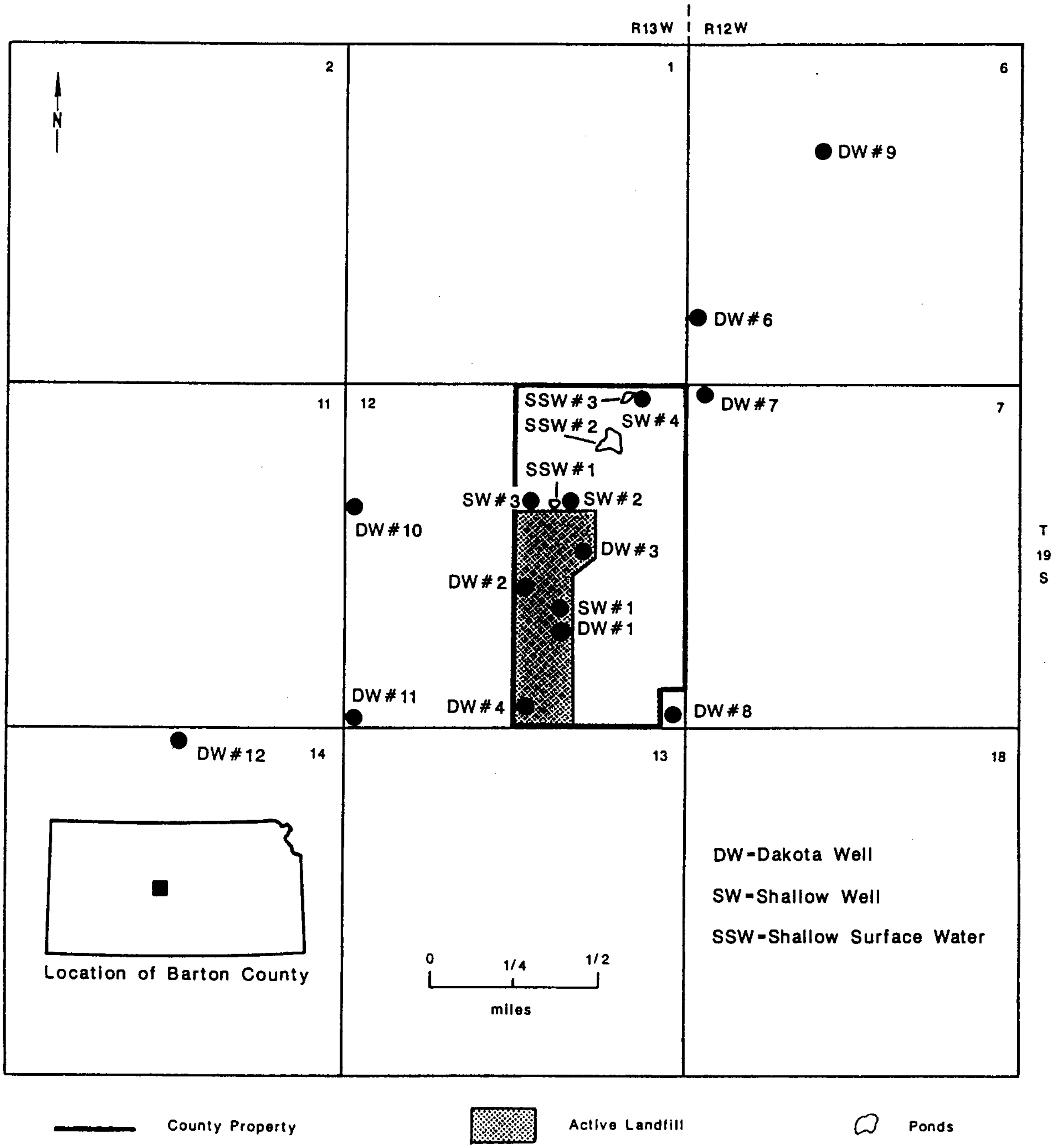


Figure 9. Location of water-quality sampling points.

- Samples were refrigerated after collection and transported to the respective laboratories for analysis.

Samples for the determination of VOC's were collected in accordance with KDHE regulations. The VOC samples were collected in 40-mL glass bottles with teflon^{TM} septa and plastic hole caps, provided by the environmental-organic laboratory of KDHE. The bottles were completely filled with sample to prevent any air from coming in contact with the sample and volatilizing any of the VOC's before analysis.

DOC samples were collected in 400-mL glass bottles with teflon^{TM}-lined screw caps. Approximately 150 mL of water was collected in each bottle. The DOC samples were acidified in the field to a pH of two units with concentrated phosphoric acid. The low pH coupled with the refrigeration of the samples should have retarded any biological activity that could have degraded some of the carbon before analysis. The acid also converts the inorganic carbon (bicarbonate alkalinity) to CO_2 gas which was later driven out of solution by aeration. In the laboratory, the samples were first filtered through a 0.45-micrometer glass-fiber filter, followed by a five-minute purging with nitrogen gas to remove any inorganic or volatile carbon in the samples.

Samples for determination of all inorganic constituents except phosphate were collected in 250-mL polyetheylene bottles and acidified. Samples for phosphate determination were collected in unacidified 500-mL polyetheylene bottles.

Dakota wells #1 and #3 were sampled using the closed-system Bennett Sample Pump. The other four monitor wells were sampled with the individual bailer dedicated to the well. Domestic wells (DW #4,6,7,8,9,10,11,12) were sampled directly from the outdoor spigot nearest the well. The shallow surface-water ponds (SSW) were sampled by wading out into the pond and filling sample bottles directly with pond water. Rubber gloves were worn by all personnel when handling the sample water.

CHAPTER 6: ANALYTICAL RESULTS AND DISCUSSION

Laboratory results of the chemical analyses are shown in tables 3 and 4. Table 3 lists VOC and DOC concentrations in the samples. Table 4 is a summary of the concentrations of dissolved inorganic constituents in the samples. The data indicate that SW #1 is the well affected most by the landfill. This was to be expected since this well is the closest to any of the buried waste cells. The approximate location of the top of the well screen in reference to the buried waste is 10 feet to the east of and 15 to 30 feet below the bottom of the waste cells. SW #1 is screened completely through the upper aquifer and extends 6 feet into an upper, discontinuous, Dakota sandstone lens (figure 3).

ORGANIC CONSTITUENTS

Municipal landfill leachates often show the presence of organic compounds, which are either flushed out of the solid wastes by the percolating water or produced during the biodegradation of the wastes. Synthetically produced volatile organic chemicals would be an example of organics that were flushed out of the solid wastes and not biologically produced. Total organic carbon and dissolved organic carbon could be produced by diverse microbial populations stabilizing the wastes.

Volatile Organic Chemicals

Very high concentrations of VOC's were found in samples from SW #1, while much smaller concentrations were found in DW #2, #4, and #12, and SW #2. VOC's are chemicals used or produced in the manufacturing of many products such as detergents, pharmaceuticals, dyes, and insecticides. VOC's also are found in household products such as spot removers, rug cleaners, drain cleaners, air fresheners, shoe polish, and

Table 3-Organic constituents in water samples taken July 1986 (all values in $\mu g/L$ except for DOC).

	Dakota wells (DW)											Shallow wells (SW)			Shallow (SSW) surface waters		
Constituent	#1	#2	#3	#4	#6	#7	#8	#9	#10	#11	#12	#1	#2	#3	#1	#2	#3
Chloromethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Bromomethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Vinyl chloride	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	19.7	ND	ND	ND	ND	ND
Chloroethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	19.6	ND	ND	ND	ND	ND
Dichloromethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	187	ND	ND	ND	ND	ND
1,1-Dichloroethylene	ND	ND	ND	1.7	ND	ND	ND	ND	ND	ND	ND	699	ND	ND	ND	ND	ND
1,1-Dichloroethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	263	ND	ND	ND	ND	ND
Trans &/or Cis 1,2-Dichloroethylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	122	ND	ND	ND	ND	ND
Trichloromethane (THM)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.8	1.4	1.0	ND	ND	ND	ND
1,2-Dichloroethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,1,1-Trichloroethane	ND	ND	ND	1.9	ND	ND	ND	ND	ND	ND	ND	1090	ND	ND	ND	ND	ND
Tetrachloromethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Bromodichloromethane (THM)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2-Dichloropropane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.8	ND	ND	ND	ND	ND

Table 3 Continued

Trans 1,3-Dichloropropane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Trichloroethylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	685	ND	ND	ND	ND	ND
Benzene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	2.2	ND	ND	ND	ND	ND
Dibromochloromethane (THM)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
CIS 1,3-Dichloropropane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,1,2-Trichloroethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Bromoform (THM)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,1,2,2-Tetrachloroethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Tetrachloroethylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	13.3	ND	ND	ND	ND	ND
Toluene	ND	1.9	ND	ND	ND	ND	ND	ND	ND	ND	ND	3.6	ND	ND	ND	ND	ND
Chlorobenzene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ethylbenzene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Meta-Xylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ortho &/or Para-Xylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	1.0	ND	ND	ND	ND	ND
1,3 Dichlorobenzene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2 &/or 1,4-Dichlorolobenzene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dichlorofluoromethane	ND	ND	ND	Det.	ND	ND	ND	ND	ND	ND	ND	Det.	ND	ND	ND	ND	ND

Table 3 Continued

Trichlorofluoromethane	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	Det.	ND	ND	ND	ND	ND
Dimethyl Disulfide	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	Det.	ND	ND	ND	ND
DOC, (mg/L)	0.98	0.66	0.68	1.22	0.57	0.49	0.75	0.51	0.42	0.57	0.85	2.68	5.26	3.08	4.79	8.08	10.19

NOTE: Det. = detected
ND = not detected
Samples collected 7-10-86

Table 4-Inorganic constituents in water samples taken July 1986.

	Dakota wells (DW)											Shallow wells (SW)			Shallow (SSW) surface waters		
Constituent†	#1	#2	#3	#4	#6	#7	#8	#9	#10	#11	#12	#1	#2	#3	#1	#2	#3
Silica	17	12	9.6	29	19	10	27	15	13	12	24	38	26	22	10	10	15
Calcium	76	83	58	151	82	49	143	77	74	74	71	241	158	48	26	27	34
Magnesium	12	15	14	16	13	15	15	21	19	19	6.4	24	25	8.7	2.7	3.2	4.2
Sodium	53	92	121	36	71	182	34	376	98	88	25	54	310	115	13	14	16
Potassium	3.9	5.0	5.0	5.2	3.3	6.2	4.6	8.2	3.9	3.8	2.3	6.5	4.9	5.5	5.0	8.5	10
Bicarbonate	295	305	345	255	321	337	286	216	297	271	256	466	316	315	98	102	135
Sulfate	36	80	60	38	40	106	20	176	81	115	11	31	343	57	12	10	11
Chloride	52	92	80	181	73	134	132	521	104	84	23	274	366	48	11	15	16
Nitrate	2.9	0.3	0.2	12	1.4	0.5	59	0.2	0.2	0.2	3.0	9.9	39	15	0.4	1.4	0.1
Ammonium	0.1	0.1	0.1	<0.1	<0.1	0.1	0.1	0.2	0.1	0.1	0.1	0.1	<0.1	0.1	0.1	0.1	0.1
*Phosphorous as PO_4	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	0.30	<0.05	<0.05	0.07	0.07	0.24	0.46
Field pH	7.35	7.45	7.25	6.90	7.20	7.10	6.75	7.30	6.85	7.15	7.05	6.90	7.25	7.55	8.35	7.25	8.30
Lab pH	7.55	7.60	7.65	7.45	7.55	7.65	7.50	7.95	7.45	7.65	7.60	7.40	7.70	8.00	8.10	8.05	8.30

Table 4 Continued

Field specific conductance,	µmhos/cm	750	1100	1000	1300	750	1400	1150	2500	1040	940	600	1750	3250	850	220	220	250
Lab specific conductance,	µmhos/cm	700	920	900	1060	800	1140	1000	2300	935	900	481	1600	2250	775	211	243	286
Field temp, °C		17.0	18.0	17.0	16.0	16.0	17.0	16.5	ND	17.0	17.0	16.0	16.0	16.0	18.0	30.5	28.0	30.5

Note: *Unacidified samples

[†] Concentrations of mg/L, except for pH

solvents. These chemicals do not occur naturally in ground water; thus, their presence is an indication of man-made contamination.

VOC's are becoming a new focus of concern due to their potential health effects from long-term consumption. Some VOC's have been shown to be carcinogenic and/or to have adverse effects on the central nervous system, liver, and kidneys. The degree of toxicity varies widely from chemical to chemical and long-term effects of low dosages are not known for many VOC's.

Neither KDHE nor the U.S. EPA currently have enforceable standards for VOC contamination. EPA is in the process of developing guidelines or standards for specific VOC concentrations in drinking water which will be called the Recommended Maximum Contaminant Levels (RMCLs). RMCLs are nonenforceable health goals set at levels which would result in no adverse health effects and include an adequate margin of safety. Also, the EPA plans to propose Maximum Contaminant Levels (MCLs) for volatile organic chemicals. MCL's are enforceable standards established to prevent possible adverse health effects to individuals drinking the water. Table 5 lists the nine VOC's for which RMCLs have been proposed.

For the more frequently found VOC contaminants, KDHE has established Kansas Notification Levels (KNLs) and Kansas Action Levels (KALs; table 6). The Kansas Notification Level is defined as the concentration of a VOC above which KDHE has the option to require notification to the consumers and to recommend that appropriate preventative action be taken by the well owner. Kansas Action Levels are defined as the concentration above which KDHE will require both notification to the consumers and mandatory action by the well owner (KDHE, 1985). The KAL's are levels of VOC's which do not present any

Table 5. Federal recommended maximum contaminant levels for volatile organic chemicals, proposed 40 CFR 141.50, Federal Register (6-12-84).

	VOC	Proposed RMCL μg/L
1.	Trichloroethylene	0
2.	Tetrachloroethylene	0
3.	Tetrachloromethane (carbon tetrachloride)	0
4.	1,1,1-Trichloroethane	200
5.	Vinyl chloride	0
6.	1,2-Dichloroethane	0
7.	Benzene	0
8.	1,1-Dichloroethylene	0
9.	p-Dichlorobenzene	750

Table 6. Kansas volatile organic chemical notification and action levels; all values in ug/L (ppb); KDHE, 1985.

VOLATILE ORGANIC CHEMICALS	KNL	KAL	DETECTION LIMIT
Chloromethane	+	id	5.0
Bromomethane	+	id	1.2
Vinyl chloride	1.0	10.	0.8
Chloroethane	+	id	3.7
Dichloromethane	+	150	0.9
1,1-Dichloroethylene	0.23	70	0.6
1,1-Dichloroethane	+	id	0.5
Trans 1,2-Dichloroethylene	+	270	0.5
Trichloromethane (THM)	100*	*	0.5
1,2-Dichloroethane	0.7	7.	0.6
1,1,1-Trichloroethane	16.8	168	0.7
Tetrachloromethane	0.27	2.7	0.7
Bromodichloromethane (THM)	100*	*	0.5
1,2-Dichloropropane	+	id	0.4
Trans 1,3-Dichloropropene	+	id	0.8
Trichloroethylene	1.8	18	0.6
Benzene	0.67	6.7	0.4
Dibromochloromethane (THM)	100*	*	0.7
Cis 1,3-Dichloropropene	+	id	0.9
1,1,2-Trichloroethane	0.6	6.	0.6
Bromoform (THM)	100*	*	1.5
1,1,2,2-Tetrachloroethane	0.17	1.7	0.6
Tetrachloroethylene	1.0	10.	1.1
Toluene	3.5	340	0.4
Chlorobenzene	+	id	0.4
Ethylbenzene	+	id	0.7
Meta-Xylene	+	620**	0.4
Ortho &/or Para-Xylene	+	620**	0.4
TVOC	10	id	

Note: Cis 1,2-Dichloroethylene is reported as Trans 1,2,-Dichloroethylene.
KNL - Kansas Notification Level
KAL - Kansas Action Level
THM - Trihalomethane
TVOC - Sum of all VOC, except THM's and 1,1,1-trichloroethane.
+ - Any positive detection.
* - THM is regulated by Kansas Drinking Water Standards, total THM < 100 ug/L
id - Insufficient data to establish a level.
** - Sum of all xylene isomers present.

short-term consumption health problems and are based on possible adverse health effects from long-term consumption.

The detection limit for the analysis of volatile organic chemicals is in the range of 1.0 ug/L (parts per billion). This is comparable to a drop of the contaminant in a swimming pool of water measuring 50 feet by 50 feet by 8 feet deep (McCool, 1986). The Environmental Organic Laboratory's VOC analysis detects the presence of 32 different chemicals. These chemicals mainly include chlorinated organics and components of petrochemical products. The four chemicals which make up the trihalomethane group also can be detected. Trihalomethanes are formed when natural organics in the ground water react with chlorine which is used as a disinfectant.

The main contaminants found in the landfill are chlorinated hydrocarbons common in commercial and industrial solvents (1,1,1-trichloroethane, trichloroethylene, 1,1-dichloroethylene). Also found at the landfill were lower concentrations of gasoline and petrochemical compounds such as xylene, benzene, and toluene. Sources for these chemicals are abundant in the area, especially near Great Bend, which is the regional center for industry and oil-field operations.

It is difficult to accurately predict the fate of chlorinated hydrocarbons in the landfill environment. In general, volatilization, microbial degradation, chemical hydrolysis, oxidation, and sorption can all affect the fate of chlorinated hydrocarbons in the environment. Studies by Davidson (1978) and Griffin and Chou (1980) suggest that sorption by soils may by the dominant removal mechanism for chlorinated hydrocarbons in soil/water systems. The major adsorbents were found to be clay minerals, iron and manganese hydrated oxides, and organic material. Chlorinated hydrocarbons were shown to reach equilibrium within a few hours in adsorption studies, and the adsorption process was shown to conform to the Freundlich adsorption equation. Overall, chlorinated hydrocarbons are highly resistant to aqueous phase mobility through earth materials, because of their low water solubilities and strong sorption by soils. However, they have been found to be very mobile in organic solvents (McCarty and Rittman, 1981).

In summary, VOC contamination at the landfill was generally confined to SW #1. The analyses showed that six out of the nine federally regulated VOC's from table 5 were present in SW #1 and all exceeded the proposed RMCL. The concentration of 1,1 dichloroethylene was found to exceed the proposed RMCL in DW #4. A total of 14 VOC's were present in SW #1 and, except for the trihalomethanes, all of their concentrations exceeded the Kansas Notification Level. Seven out of the 14 also exceeded the Kansas Action Level. The 1,1 dichloroethylene concentration in DW #4 exceeded the KNL but did not exceed the KAL. Although VOC concentrations were detected in DW #2 and #12 and SW #2, the concentrations were below both the EPA's proposed RMCLs and KDHE's KNLs and KALs.

Dissolved Organic Carbon

Dissolved organic carbon concentrations were determined from each of the sampling locations and are presented in table 3. In a study conducted by Leenheer et al.(1974), DOC levels were determined from 100 different sites in 27 states. The levels of the DOC in ground-water samples collected were generally very low, but values ranged from undetectable to as high as 15 mg/L, with 85% of all the values below 2 mg/L. Six samples collected in sandstone aquifers had a median value of

0.7 mg/L and a mean value of 1.0 mg/L. A recently conducted study of DOC concentrations in Kansas ground water included three samples from the Dakota Formation which had a mean concentration of 0.59 mg/L (Miller et al., 1986). The DOC levels from the present study also are generally very low with the exception of the shallow wells that penetrate the upper aquifer and the surface-water ponds, which will be discussed.

DOC levels have been shown to decrease with increasing depth of the sample in an unpolluted aquifer (Junk et al.,1980). The major sources of organic carbon in an unpolluted aquifer are from the overlying soils, surface vegetation, and surface-water infiltration. Surface waters normally contain higher levels of DOC originating from decaying organic matter on the banks and bottom sediments. The surface-water ponds downgradient from the landfill also have organic matter being added by the excretions of the livestock that use the ponds for drinking water.

DOC levels obtained from this study follow the trend of decreasing DOC with increasing depth. The surface waters show the highest levels with a mean ($\bar{x}$) value of 7.69 mg/L followed by the shallow wells ($\bar{x}$ = 3.67 mg/L) and finally the Dakota wells ($\bar{x}$ = 0.70 mg/L). Dakota wells # 1 through #4, which are located within the fenced-in portion of the active landfill, show slightly higher mean DOC levels ($\bar{x}$ = 0.89 mg/L) than do Dakota wells # 6 through #12 ($\bar{x}$ = 0.59 mg/L), which can be considered unpolluted background wells. DW #4, which is the water supply for the scale house at the landfill, had the highest DOC value of all the Dakota wells (1.22 mg/L). Poor well maintenance could account for this high value in DW #4 because the PVC casing that extends above the land surface is cracked and open to contaminants entering the well from the land surface. Overall, DOC levels in samples from the Dakota wells do not appear to have been significantly affected by the landfill.

Organic contamination often will cause an increase in DOC levels of one or more orders of magnitude over background DOC levels. Values greater than the threshold levels of DOC (> 5 mg/L) could indicate contamination and possibly organic materials moving into the aquifer from sources such as surface-water infiltration and subsurface waste-disposal operations (Malcolm and Leenheer, 1973). DOC levels in shallow wells #1, 2, and 3 are approximately one order of magnitude higher than those of the Dakota wells. All three of the shallow wells are near the active landfill. Unfortunately, no background DOC data are available for water in the upper aquifer. Therefore, it is impossible to precisely quantify the effect the landfill has had on the DOC levels in the upper aquifer. The DOC levels reported in table 3 might be normal DOC levels for the upper aquifer.

Surface-water infiltration from SSW #1 is not considered to be a factor in the DOC levels of SW #2 and #3 because the inorganic analyses showed the waters to differ greatly, with no significant infiltration and mixing evident. SW #2 (5.26 mg/L DOC) contained the greatest level of all the ground-water wells and this is possibly due to some landfill leachate migrating to the well through the upper aquifer. To provide more definitive information regarding the effects of the landfill on the DOC values in the upper aquifer, more shallow wells would be needed on all sides of the active landfill. Of these wells, at least one should be placed to provide reliable upgradient, unpolluted, background water samples.

INORGANIC CONSTITUENTS

The concentrations of inorganic constituents in ground water collected in the sampling program are given in table 4. Table 7 shows the various inorganic constituents in the landfill samples compared to concentration ranges reported for leachate and to the EPA's drinking-water standards. Shallow wells #1 and #2 seem to be slightly impacted by the landfill, as also was seen in the VOC and DOC analyses. It should be recognized that much higher concentrations of the ions shown in table 7 can occur naturally, especially in Kansas, so these concentrations per se cannot be definitively associated with landfill leachate. However, comparison of the concentrations of inorganic constituents in SW #2 with the concentrations in nearby wells (e.g., SW #3) not polluted with VOCs suggests that leachate may be present.

SW #2 and #3 are in close proximity (approximately 600 feet) and both tap the upper aquifer. These factors should normally result in very similiar water types between the two wells. The cation (Ca, Mg, Na) and anion (SO_4, Cl, NO_3) concentrations are much higher in SW #2 than SW #3 (table 4). These higher concentrations can be attributed to the location of the well, which is northeast (topographically downgradient) of the majority of the waste cells. SW #3 is located to the northwest of the majority of the waste cells, which is upgradient from the predicted direction of ground-water flow. SW #1 also contained high levels of Ca, HCO_3, and Cl, and as has already been shown, was contaminated with VOC's from the landfill.

It should be noted that Dakota wells #4, 7, 8, 9, and 10 are screened at lower depths within the Dakota Formation which contain

Table 7. Comparison of inorganic water quality to ranges for various inorganic constituents in leachate from sanitary landfills (from Freeze and Cherry, 1979, and Tchobanoglous et al., 1977).

	Leachate range	SW#1*	SW#2*	SW#3*	DW#4	DW#8	DW#9*	EPA drinking-water standards
Calcium	200-3000	241	158	22	151	143	77	None
Sodium	200-2000	54	310	115	36	34	376	None
Chloride	100-3000	274	366	48	181	132	521	250
Sulfate	100-1500	31	343	57	38	20	176	250
Nitrate	5-40	9.9	39	15	12	59	0.2	45

Note: All values in mg/L
* denotes well not presently used for drinking water

ground water with higher Cl levels. Discussions with local drillers and landowners showed that in general, the water in the Dakota Formation naturally increases in salinity with increasing depth. Also, samples from wells DW #4 and #8 contained elevated levels of nitrate that could be caused by sources other than the landfill. These alternative sources could be scale-house septic-tank leakage or poor well construction, which could allow nitrates to enter the wells at or near the land surface. A possible natural alternative could be the nitrification of ammonium ions migrating upwards from deep, anaerobic sediments.

Tin cans and ferrous metals comprise approximately 8% by weight of a typical municipal solid waste (table 1). A tin can has tin plate layered over steel. Tin is highly resistant to degradation in an anaerobic environment, but compaction of the tin cans within the landfill will break the tin surface exposing the steel. Steel is primarily composed of iron, but also contains manganese, sulfur, phosphorous, silicon, nickel, chromium and vanadium in smaller amounts (Metzler, 1981). Iron, manganese and chromium are all partially soluble at typical leachate pH values. Iron is found in leachate because landfills quickly become anaerobic after waste burial, producing a chemically reducing environment in which ferric salts are reduced to ferrous salts. Ferrous salts are more soluble and thus iron leaches from the landfill. Considering these facts, future water samples should be analyzed for metals such as iron, which would be a good indicator of the presence of leachate. Iron analysis would improve the ability to track the leachate plume because background iron concentrations should be very low (0.01 to 10.0 mg/L) in the upper aquifer.

CHAPTER 7: ASSESSMENT OF SITE LEACHATE GENERATION

It is evident from analytical results that some leachate is being generated by the landfill. This leachate has the potential to become a human-health problem by entering the local drinking-water aquifer and to threaten stock animals and wildlife by degrading the water quality in the Cheyenne Bottoms and the other surface waters between the landfill and the Bottoms. Therefore, an estimate of the amount of leachate being generated by the landfill and the directions it might migrate is important.

A quantitative description of leachate generation from the landfill is essential for determining its effects on water quality and the value of control methods. Methods of estimating leachate generation include field tests, physical and mathematical models, and monitoring of actual installations. All of these methods attempt to produce accurate estimates of leachate generation; but, due to the numerous factors involved in leachate generation, such estimates are only accurate within an order of magnitude. This lack of accuracy is due primarily to the inability to measure the surface-runoff coefficients and evaporation rates of the landfill site and adjacent area (Duval et al., 1979). Other problems include estimation of the in situ permeabilities (i.e. hydraulic conductivities) of wastes and of the underlying and cover soils. All these factors contribute to the problem of estimating leachate volume.

The volume of leachate generated at a landfill is dependent on many factors. In general, the most important factors are (Remson et al., 1968):

- Availability of water

* Landfill-surface conditions

* Solid-waste conditions, and

* Underlying soil conditions.

Current federal solid-waste legislation prohibits landfilling within a floodplain, primarily eliminating ground water as a potential water source for leachate generation. Thus, the primary contributor to the availability of water to a landfill is direct precipitation. In order to assess the effect of precipitation on the Barton County landfill, climatological data were obtained from the Great Bend (#3218) weather station, located at latitude 38°, 21' N and longitude 98°, 46' W along the Arkansas River south of the Cheyenne Bottoms Waterfowl Refuge. The average annual precipitation is 24.7 inches based on 52 years of record from 1931 to 1982. The average annual precipitation since the opening of the landfill in 1972 to 1982 was 26.74 inches; 73% of this precipitation occurred during the months of April through September. Water reaching the landfill surface by precipitation may evaporate, transpire, infiltrate through the landfill surface, or leave the site as surface runoff. Surface conditions which may affect leachate generation include vegetation, cover material, surface topography, temperature, humidity, and wind speed above the landfill.

The quantity of water that can infiltrate through the surface of the completed portions of the landfill depends on the hydraulic conductivity of the cover soils. At the present time, the cover soils at the landfill are composed of loess excavated during the construction of new trenches. Two cover soil samples were collected for hydraulic conductivity analysis from completed portions of the landfill. The samples were located northeast of the scale house near the middle of the site. Vertical hydraulic conductivities (K) were determined by the author using the constant-head permeability test in the Kansas Geological Survey's porous media laboratory. The K-values were measured to be 0.96 feet/day and 2.66 feet/day for the two samples. Permeability ranges for the loess reported in the Barton County soil survey publication (1981) were measured from 1.20 feet/day to 4.00 feet/day. The results show a fairly wide range of permeability and it is thought that permeability values of the cover soil will vary over the landfill with the clay content and the degree of compaction.

Once the field capacity of the surface cover soil is attained, leachate will percolate through the solid wastes. Field capacity is defined as the maximum moisture content which a soil or solid material can retain in a gravitational field without producing continuous downward percolation. Studies of typical municipal solid waste show that approximately 40% of the total solid waste is composed of paper products (table 1). Paper has a large water-holding capacity; thus, once the wastes are saturated, they have the potential to hold moisture between cover-soil wetting and drying cycles. Each time the field capacity of the cover soil is attained, a slug or pulse of leachate can be generated by the waste cell provided that the wastes have remained at field capacity.

MODELING OF LEACHATE GENERATION

The Versatile Budget Version 2 (VB), a soil-moisture computer model, was used to simulate the water budget in the area of the landfill. The VB program uses readily available data (soil and cropping patterns, daily temperatures, and precipitation) to calculate a simple water budget. The model produces estimates of soil-moisture deficit

(SD), runoff (RO), potential evaporation (PE), actual evaporation (AE), and deep drainage (DR). Deep drainage is considered to be the water which percolates below the rooting depth. Vegetative growth requires the usage of large amounts of water, which is the reason that any water migrating below the rooting zone is considered to be deep drainage (Baier et al., 1979). This simple budget provides enough data to make some general estimates of the water balance at the landfill. The most important parameter which affects leachate production is drainage below the rooting depth.

The input parameters used by the model were adjusted to become as site specific as possible for the landfill. Common parameters include: 1) crop (vegetation) coefficients; 2) solar energy at the top of the atmosphere; 3) soil available water capacity (AWC); 4) daily temperatures and precipitation; and 5) Z-table values (representing the relationship between the ratio of actual to potential evaporation as a function of soil moisture available to vegetation).

The Z-table chosen for the landfill simulates clayey-silt soil. The crop coefficients of Brome grass were used because they are considered to be the equivalent to natural growth. Vegetative cover for the completed portions of the landfill consists of native grasses that have rooted on the landfill. In calculating the root-zone depth parameter for the AWC, a rooting depth of 3 feet was used for the grass on the landfill. Brome grass is normally modeled using a 5-foot rooting depth but was shortened because of the landfill's cover-soil thickness of 3 feet and the fact that solid waste does not provide a suitable rooting material.

In estimating the amount of leachate generated, it was assumed that the solid waste was at field capacity and that the deep drainage values represent the amount of water that reaches the top of the solid waste. Thus, if the wastes are at field capacity, any additional infiltration will push leachate through the bottom of the waste cell. Consequently, the values shown in table 8 are the theoretical-maximum leachate estimates for the completed portions of the landfill.

The controlling factor in this scenario is the moisture content of the solid waste. If the waste is not at field capacity, then the rate and magnitude of leachate generation would be lower. Based on personal obervations, it is thought to be doubtful that field capacity of the solid wastes is very often achieved in completed portions of landfill. This is due to the climatic conditions of the area along with the initial dehydrated condition of the wastes and the intermediate-cover-soil layers that usually exist before the final cover is placed. If significant precipitation has not occurred immediately before the addition of the final cover, the wastes and intermediate-cover-soils are very low in moisture content. Of course, this is not the case if wastes are allowed to be saturated by standing water in the bottom of the trenches, or if proper drainage of the cover soil is not provided. In these cases, the initial dry conditions of the wastes would not be significant because the wastes would become saturated over time.

If field capacity of the entire thickness of the waste cells is rarely attained at the landfill, the major methods for the generation of leachate would be channeling of water through the pores of the wastes and direct leaching of wastes by standing water in trench bottoms. Allowing water to accumulate in the bottom of active trenches and the "wet pit" may be a major cause of leachate generation.

Table 8. Estimates of leachate generation.

Water year	Precipitation	Actual evapotranspiration	Runoff	Deep drainage	Acre-feet/year maximum leachate
1957	36.56 in.	28.10 in.	2.38 in.	5.55 in.	37.00
1984	23.02 in.	18.93 in.	1.50 in.	6.09 in.	40.00
1956	12.21 in.	15.92 in.	0.76 in.	0.48 in.	3.20

Computer runs were made for water years with precipitation that was average (figure 10, 1984-23.02 inches), above average or wet (figure 11, 1957-36.56 inches), and below average or dry (figure 12, 1956-12.21 inches). In table 8, deep drainage for the three different years is 5.55 inches/year (wet), 6.09 inches/year (average), and 0.48 inches/year (dry).

It should be noted from table 8 that deep drainage for the wet year (1957) is slightly less than the average year (1984). It would be expected that greater precipitation would result in higher drainage through the soil, but this is not the case in this situation. Three wet years were modeled and all created less deep drainage than the average year.

Two conditions can be proposed to account for less deep drainage occurring in a wet year. The first condition that affects the amount of deep drainage is the times during the year that precipitation occurs. The months between November and March usually have low precipitation, but the average water year (1984) had above-average precipitation in November. This brought the soil near field capacity for the winter months. Additional precipitation during December through February when the majority of the cover soil was saturated and frozen, allowed deep drainage to begin with the start of the spring rains (figure 10). Secondly, during the modeled wet years, the majority of the yearly precipitation occurred in large storms (> 2 inches) during the spring and summer months (figure 11). This results in a high run-off situation and coincides with the period that plant use of water and direct evaporation are greatest. Thus, a lower drainage value could be expected.

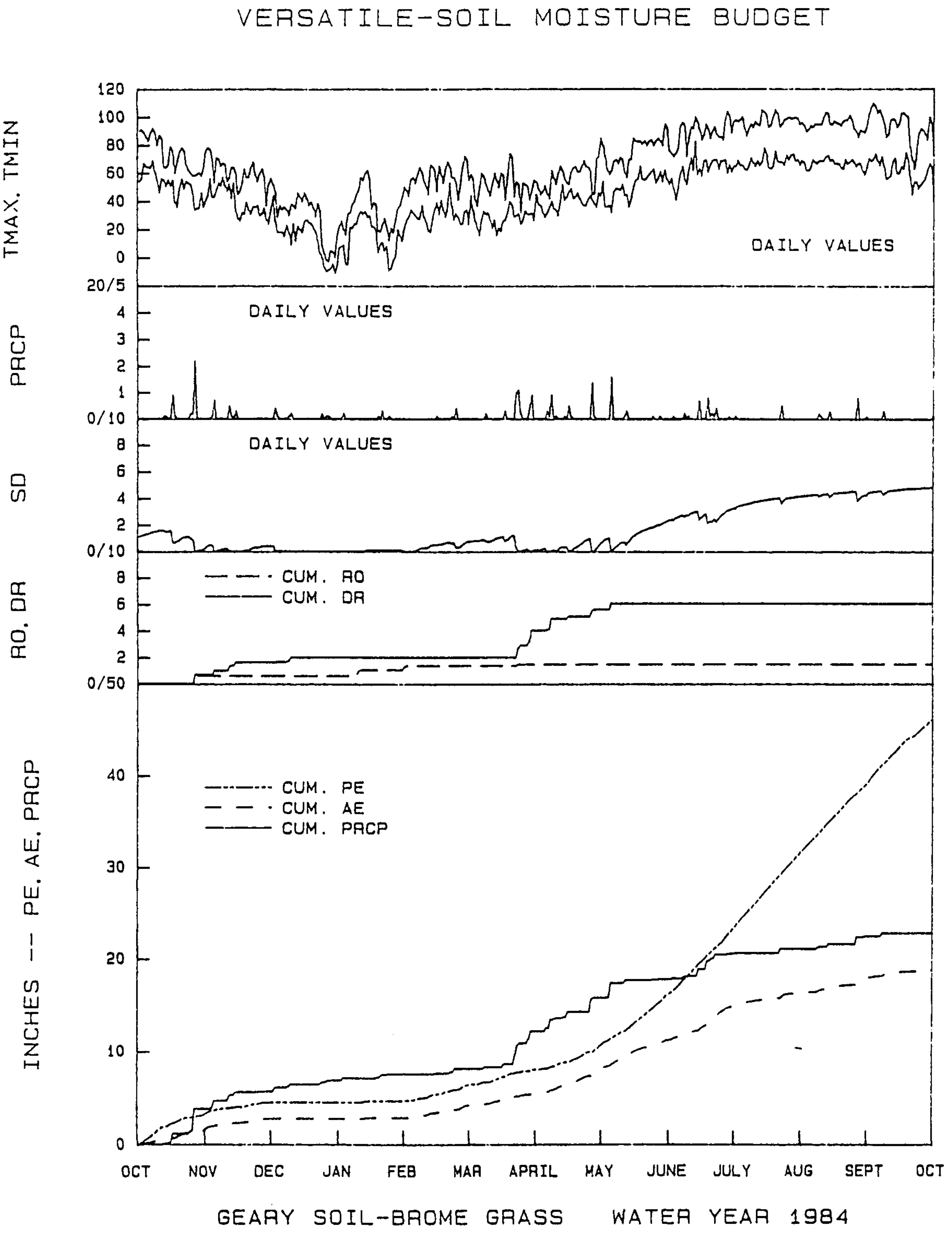


Figure 10. Cumulative landfill water balance - average year.

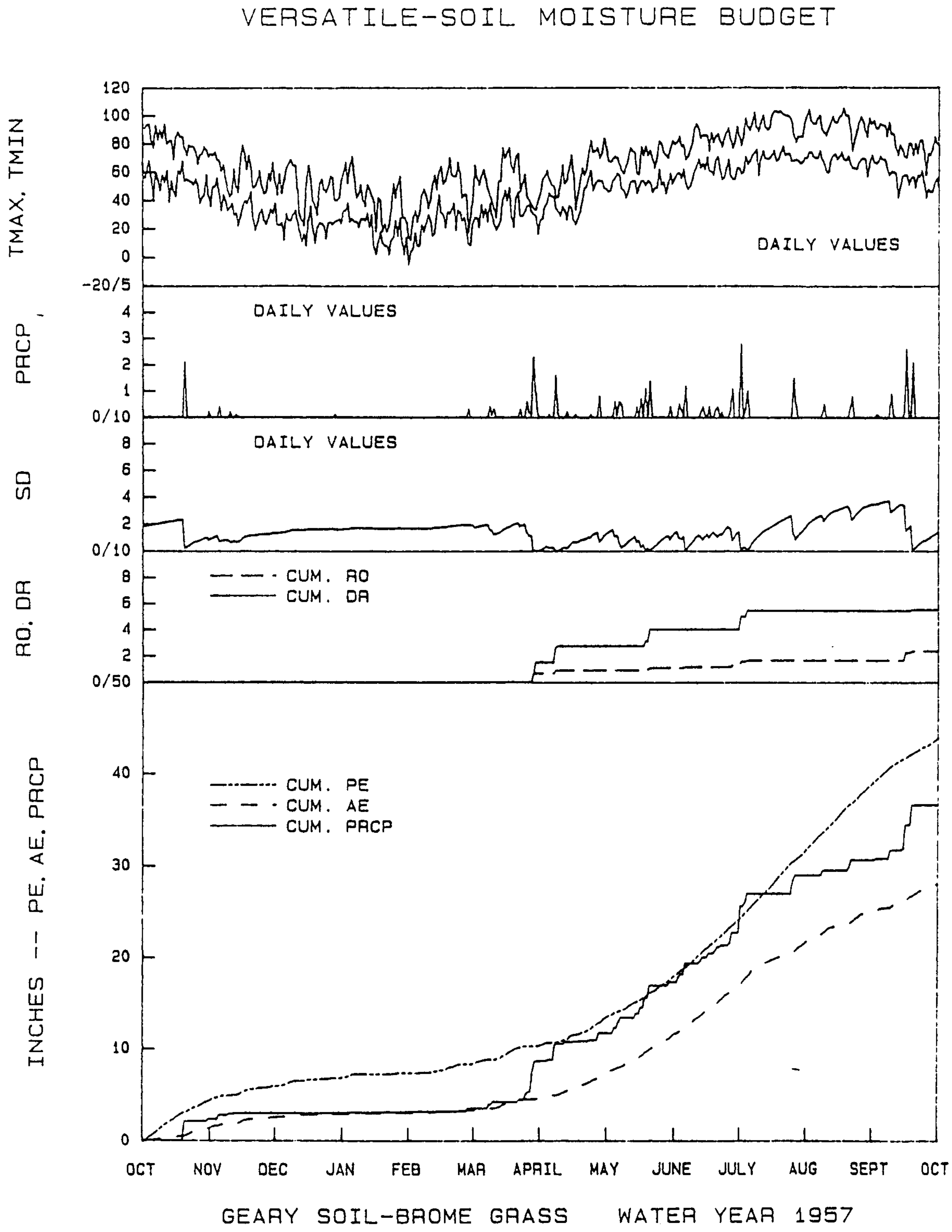


Figure 11. Cumulative landfill water balance - wet year.

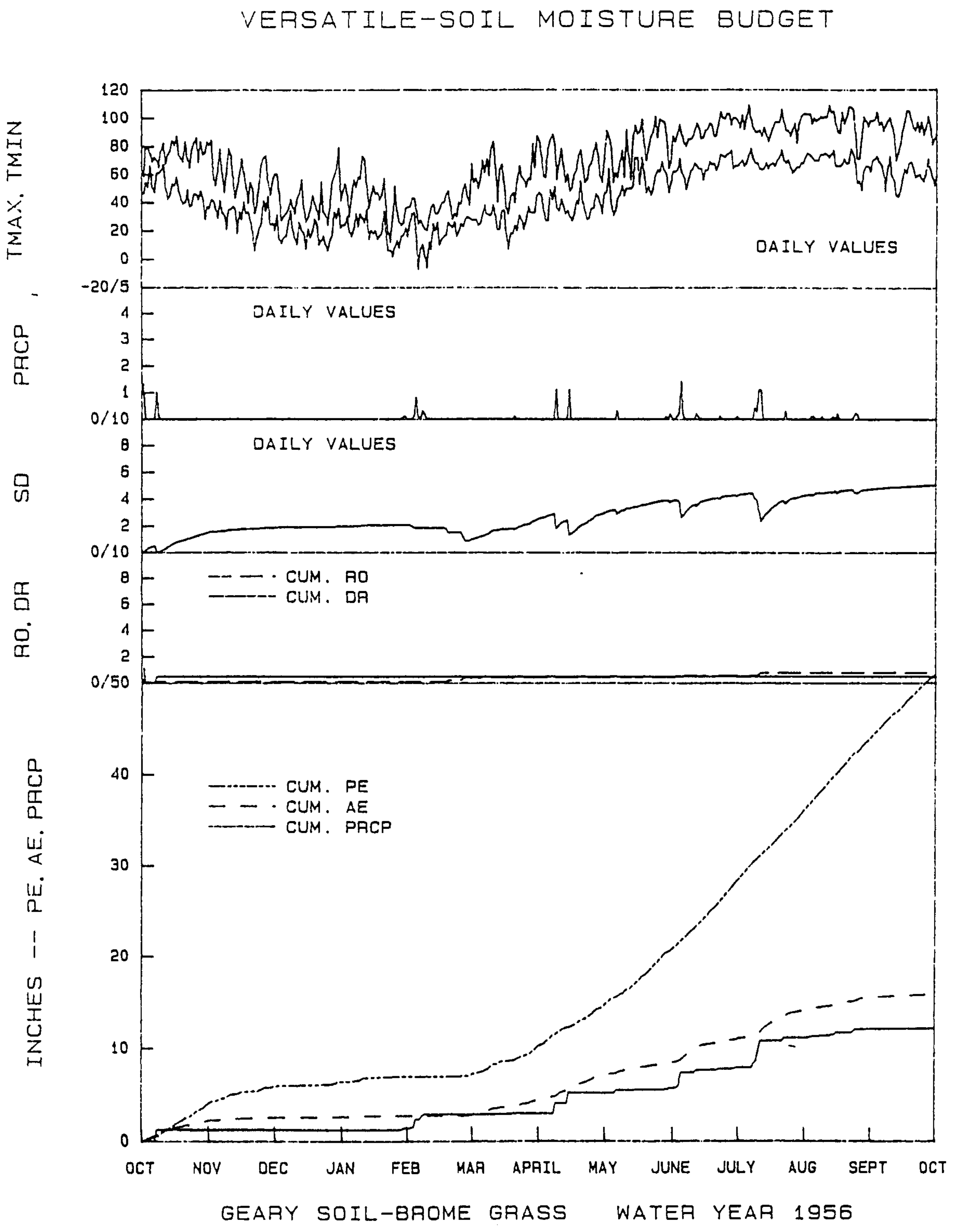


Figure 12. Cumulative landfill water balance - dry year.

The leachate values generated by this model are estimates for the completed portion of the landfill that have the 3-foot cover-soil mantle in place. The "wet pit" and the active trenches will have different values for leachate generation than the completed portions of the landfill. These two areas are either uncovered or have a 6-inch layer of soil over the solid waste. Also, these trenches receive surface run-on water until the time they are completely filled and graded to the proper slope. These conditions allow for a higher rate and magnitude of leaching from the wastes that could only be quantified by a leachate-collection system.

CHAPTER 8: ASSESSMENT OF SITE LEACHATE MIGRATION

The water-quality data collected from the Dakota wells indicate that the leachate presently is confined to the upper aquifer. The Dakota wells, which draw water from the shallowest local drinking-water aquifer, show no impact from the landfill leachate. DW #4 does show a small amount of VOC contamination as well as elevated nitrogen levels, but this cannot positively be connected with leachate percolating downward through the Dakota clay. Due to the poor well-head condition, leachate in the upper aquifer could be flowing into the gravel packed well annulus.

One concern about the concentration and direction of the contaminants in the upper aquifer is that there are several local domestic wells near the landfill. These wells are generally gravel packed from 10 feet below land surface to the bottom of the well. If the contamination plume reaches the wells by lateral movement, pollutants could enter the well annulus from the upper aquifer and flow down into the drinking-water aquifer.

Another concern is that the upper aquifer possibly discharges towards the stock ponds and the Cheyenne Bottoms. In the event that contaminated water were to seep from the upper-aquifer outcrop on the north slope of the divide, downgradient surface-water bodies could be contaminated. This situation might already exist at the landfill as the water quality slightly decreases in SSW #2 and #3, which are at a greater distance from the landfill.

It would be expected that if only polluted runoff water from the active landfill were flowing into SSW #1, then water quality would improve with increasing distance from the landfill because of dilution effects and biological stabilization of the contaminants in the landfill-runoff water. The inorganic and DOC analyses actually show decreasing surface-water quality with increasing distance from the landfill. No VOC's were found in the ponds; but this was expected because the pond water is exposed to the atmosphere, which would permit volatilization of any VOC's present.

It is possible for leachate to enter SSW #2 and #3 without entering SSW #1, and the decreasing water quality could be the result of leachate from the upper aquifer discharging into the lower ponds from the outcrop in the landfill ravine. In the northern portion of the landfill ravine, the upper aquifer crops out near the northern edge of SSW #2 and on all sides of SSW #3. Thus, discharges into the ravine from the upper aquifer could be degrading the water quality of SSW #2 and the downgradient SSW #3. Nevertheless, the water-quality data do not indicate the presence of significant amounts of leachate.

Since no runoff samples were collected from landfill drainage, it was not proven that runoff water from the landfill is heavily polluted. If the runoff water is relatively pure, its quality would naturally degrade with time and increasing distance from the landfill, as reflected in the surface-water analyses (tables 3 and 4). With increasing water age, DOC concentrations in the ponds would increase due to algal productivity and run-off water from the surrounding pasture. Also, the concentration of salts would increase due to evaporation and increased dissolution of minerals from the surrounding sediments. Considering the minimal data that are currently available, the natural processes are a more likely cause of the surface-water degradation than upper-aquifer discharge. A more conclusive answer could be provided if the direction of transport of the pollutants in the upper aquifer were investigated in greater detail.

PERMEABILITY OF THE DAKOTA CLAY

Pollutants in the upper aquifer can either migrate vertically through the Dakota clay, or horizontally along the top of the clay. In order to determine if the pollutants will migrate downward towards the drinking-water aquifer, two soil samples were collected to determine vertical hydraulic conductivities of the Dakota clays underlying the landfill. The samples were analyzed by the author in the Kansas Geological Survey's porous-media laboratory. Hydraulic conductivities were determined using the constant-head permeability test. The samples were taken at depths of 1 foot and 6 feet from the top of the Dakota Formation. The samples consisted of red sandy clay and were relatively undisturbed by the sampling procedure. Hydraulic conductivities were measured to be 1.56 feet/day and 2.25 feet/day for the 1-foot and 6-foot samples respectively.

The measured values are higher than the average values for Dakota clay due to the sand content in the clay in the upper portion of the Dakota in this area coupled with the fact that sampling always affects the in situ permeabilities somewhat. Samples of Dakota clay inspected during drilling showed the sandy clay present only at the top of the formation, with lower samples consisting of hard clay. Literature reports indicate that the confining clay layers of the Dakota Formation have vertical hydraulic conductivities ranging from 0.001 foot/day to 0.00001 foot/day (Butler, 1982).

Below the landfill, the thickness of the Dakota clay separating the contaminated upper aquifer from the shallowest sandstone drinking-water aquifer ranges from 17 feet below DW #1 to 60 feet below DW #3. The clay thicknesses were determined from the drilling and gamma logs. The conservative calculation of only 17 feet of clay thickness below DW #1 was made with the assumption that contaminants are completely dispersed throughout the 23-foot-thick discontinuous sandstone lens. The thicknesses of these Dakota clays, along with the low vertical hydraulic conductivity of the confining clay layers, indicate a lateral direction of contaminant transport as opposed to vertical.

Figure 7 shows the elevation of the erosional surface of the Dakota Formation, indicating a slope in an east-southeasterly direction below the landfill. Considering the low permeability characteristics of the underlying clay, the contaminants are expected to migrate in a similiar direction. This could possibly account for the poor water quality of DW #4. This well is located downgradient (in reference to the upper aquifer) from the "wet pit" and the earliest waste cells that were constructed at the landfill. Upper-aquifer leachate flowing towards the well could be affecting the water quality. As with the potentiometric-surface map, the limited amount of data allows only a very generalized contour map of the clay.

CONTAMINANT-TRANSPORT PROCESSES

The physical processes that control the transport of dissolved substances (solutes) in a ground-water-flow system are advection and hydrodynamic dispersion. Advection is the process by which solutes are transported by the bulk motion of the flowing ground water. Hydrodynamic dispersion is the process that causes the solutes to spread out from their expected advection-flow paths and also causes dilution of the solutes. It occurs because of mechanical mixing during fluid advection and because of molecular diffusion due to the thermal-kinetic energy of the solute particles. At low ground-water velocities, molecular diffusion is the controlling factor in the dispersion of the contaminants. At high velocities, mechanical mixing is the dominant dispersive process. Ground-water velocities were not measured or estimated in this study; thus, the dominant dispersive process in the upper aquifer is unknown. For the initial study, it is sufficient to conclude that the contaminants in the upper aquifer will probably migrate horizontally along the top of the Dakota clay.

It is possible that the upper aquifer as described may be a perched water table occurring only in the vicinity of the landfill. The upper aquifer could be recharged through the bottom of the "wet pit" as well as through the active trenches. This would result in a limited quantity and areal extent of water in the upper aquifer which will vary over time with the amount of precipitation that the landfill receives. In a perched water table, the ground-water velocities would be low, allowing molecular diffusion to become the controlling factor in the dispersion of the contaminants. It is also not known if the discontinuous upper-sandstone lens penetrated in the boreholes of DW #1 and SW #1 was previously contaminated along with the upper aquifer which rests on top of it. Currently, SW #1 is screened through both the upper aquifer and the sandstone lens; thus, determining if the sandstone lens was contaminated prior to drilling is impossible. A well screened to isolate the upper-sandstone lens would be needed to determine if contamination has invaded the sandstone. It is possible that the upper discontinuous sandstone lens is now locally contaminated by the vertical pathway created by SW #1.

Considering the high permeability of the sediments in this zone and the fact that only 7 feet of Dakota sandy clay separate the loess and the sandstone lens, it is thought that the sandstone lens was at least partially contaminated before SW #1 was drilled. As a precaution, SW #1 should be drilled out with an oversized bit and plugged with grout to eliminate the vertical pathway for contaminants. These points reinforce the need for more shallow wells or auger probes to better define the areal extent of water and contaminated water, along with a better understanding of the water-level gradient in the upper aquifer.

CHAPTER 9: SUMMARY AND CONCLUSIONS

The results from the water-quality analyses indicate that some contamination by landfill leachate has already occurred in the upper aquifer. The major contaminants found were VOCs which are potential human carcinogens. The deeper drinking-water aquifer and the surface water ponds seem to be unaffected by landfill leachate. A more detailed monitor-well network (chapter 10) should be installed in the upper aquifer to more precisely delineate the extent and degree of contamination.

The Barton County sanitary landfill is located in a geologic environment favorable for minimizing the environmental impacts on the drinking-water aquifer underlying the landfill. However, modeling shows that significant quantities of leachate can be generated; therefore, the landfill has the potential to produce undesirable effects on the downgradient surface-water bodies through surface runoff and possible discharges from the upper aquifer. Also, local domestic wells could potentially be contaminated by lateral leachate flow along the top of the Dakota clay. The considerable thickness and low permeability of the Dakota clays have protected the drinking-water aquifer from contamination up to this point and should continue to do so.

In regard to possible effects on the Cheyenne Bottoms Waterfowl Refuge, it appears that the landfill has had no impact on water quality in the Bottoms. The contaminated upper aquifer does not underlie the Bottoms and crops out approximately 2 miles to the south of Pool 5. In the event that the upper aquifer accumulates enough water to begin discharging substantial quantities of leachate towards the Bottoms, microbial activity, natural aeration and evaporation (for VOC's), dilution effects, and clay adsorption and ion exchange would help stabilize any contaminants before they reach the Bottoms.

The use of the computer-based soil-moisture model provided theoretical maximum values for leachate generation in completed portions of the landfill. Leachate generation values for the exposed active trenches and the "wet pit" would be much greater than the modelled values for completed trenches. Model results indicate that significant quantities of leachate can theoretically be generated in average and above-average precipitation years, while little or no leachate will be formed in below-average precipitation years. The model also showed that leachate generation is heavily dependent on the time of year precipitation occurs and on the intensity of the precipitation events.

In general, the management and operational practices at the landfill follow accepted sanitary-landfill guidelines. The one major weakness is the failure to provide a daily soil cover for the solid wastes in the active trenches as well as the "wet pit." This practice may be a major contributor to leachate generation. If, in the future, the solid wastes are covered daily with soil, leachate generation may be greatly curtailed.

CHAPTER 10: RECOMMENDATIONS

Several options are available to attempt to prevent any further contamination of the water resources surrounding the landfill. The first step would be to install more shallow wells on the east, west, and south sides of the landfill to define the extent of contamination in the upper aquifer.

To facilitate correlation of water quality in the upper aquifer to water quality in the drinking-water aquifer, it would be desirable for two of the new shallow wells to be placed near Dakota wells #2 and #3. One shallow well could be placed on the west side of the landfill near DW #2 and the other on the east side near DW #3. A distance of at least 15 feet separating the wells (e.g., SW #1 and DW #1) would be advisable to minimize cross-contamination possibilities. On the east side of the landfill where contamination has been found, at least three more shallow wells should be installed. This would provide better coverage paralleling the wastes in the general direction of ground-water flow. It is recommended that two of the wells be located at least 500 feet to the east of DW #1 and #3. The third well could be located 500 feet east of the southeast corner of the current landfill fenceline (figure 13). A shallow well or wells could be placed along the south boundary between the scale house and east fenceline. Considering the dip of the Dakota clay, wells on the south and east sides should have the greatest probability to show contamination. If contamination also is found in these proposed wells, additional wells would be needed at a further distance from the landfill's fenceline until the limit of the plume can be determined.

Contamination is expected to move laterally over the top of the

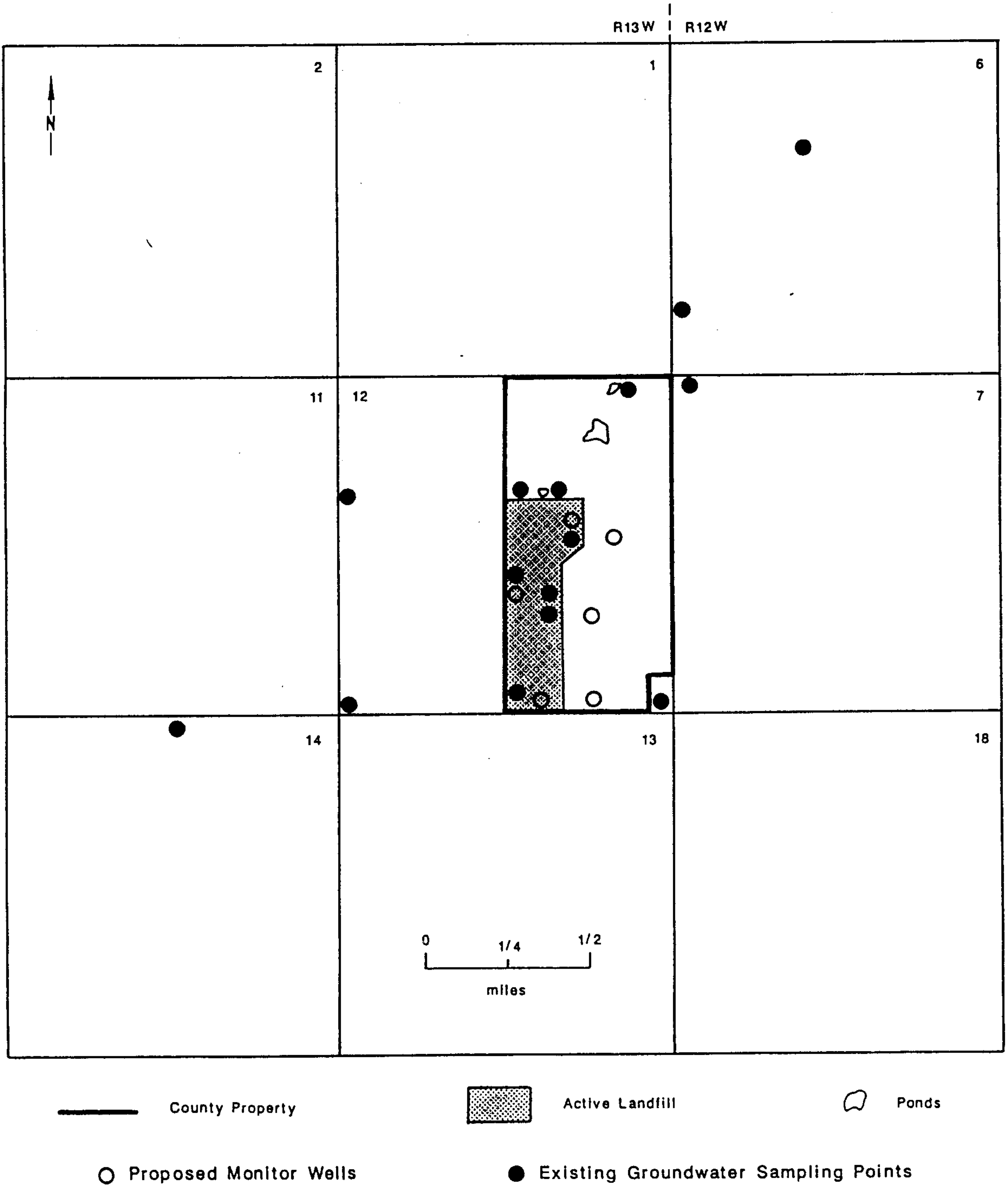


Figure 13. Location of proposed monitor wells.

relatively impermeable Dakota clay. Since the plume could eventually reach the local domestic wells and contaminate the drinking-water aquifer by flowing down the well annulus (see chapter 8), any new domestic wells drilled in the vicinity of the landfill should be screened and gravel-packed only in the producing zones greater than 105 feet. A grout seal should be placed starting above the well screen and extending up to land surface.

If the extent of contamination is determined to be widespread by the chemical analysis provided by the proposed shallow wells, remedial alternatives must be considered. Alternatives could include: 1) an impermeable surface liner for the completed portions of the landfill, or liners placed in the bottom of waste trenches prior to filling; 2) installation of a leachate-collection and treatment system; 3) improved site drainage and seeding of completed areas with a high-evapotranspiration crop; 4) provision of a daily soil cover for waste cells and the "wet pit,"; and 5) pumping of standing water in the bottom of active trenches or the "wet pit" as soon as possible following a precipitation event to minimize infiltration into the underlying wastes. Alternatives 3, 4, and 5 together would probably be less expensive than either alternative 1 or 2, but an economic analysis of the above options should be made if widespread contamination is discovered. It should be noted that alternatives 3 and 4 are considered normal sanitary-landfill practices, but they were not consistently being implemented at the time this study was completed.

It is recommended that future trenches be constructed in such a way so as to minimize the quantity of surface run-on occurring. Also, each day's solid waste should be placed in a small enough area in the active trench so the proper thickness (6 feet) can be attained before the daily soil cover is placed. These two recommendations should greatly reduce the amount of water coming into direct contact with the wastes, thus reducing leachate production.

As was mentioned earlier, SW #1 should be plugged to prevent contaminants from migrating downwards through the upper discontinuous sandstone lens and coming that much closer to the drinking-water aquifer. Another option that should be considered is the future use of DW #4 as a water supply for the scale house. This well seems to be downgradient from the upper aquifer, and some contamination was evident in the chemical analyses. More samples should be analyzed from the well, with the results being compared to those of this study. If contamination is again detected, a new water supply for the scale house should be considered. At the time of this study, DW #4 was being used as a source of drinking water as well as for other uses.

This study has provided a general indication of the extent of local water-resource contamination from the Barton County landfill. It has also provided an assessment of future effects pollution from the landfill could have on surrounding ground-water supplies and the Cheyenne Bottoms Waterfowl Refuge. Presently, the landfill has not significantly affected any usable water supplies. Unfortunately, as is the case with every subsurface waste-disposal operation, the landfill does hold potential for future adverse environmental impacts, especially if the management and operation of the site fail to meet recommended sanitary-landfill guidelines.

CHAPTER 11: CITED REFERENCES

Baier, W., Dyer, J.A. and Sharp, W.R., 1979, The versatile soil moisture budget: Technical Bulletin 87, Agrometeorology Section, Research Branch, Agriculture Canada, Ottawa, Ont., 52pp.

Bayne, C.K., 1977, Geology and structure of Cheyenne Bottoms, Barton County, Kansas: Kansas Geological Survey, Bulletin 211, part 2, 12pp.

Butler, R.D., 1982, Hydrogeology of Lower Cretaceous rocks, Williston Basin, North Dakota: Proceedings of the C.V. Theis Conference on Geohydrology, Dakota Aquifer Symposium, October 1982, Lincoln, Nebraska.

Crawford, J.F., and Smith, P.G., 1985, Landfill technology: Butterworths Publishing Company, 159pp.

Davidson, J.M., 1978, Adsorption, movement, and biological degradation of high concentrations of selected pesticides in soils: Proceedings of the Fourth Annual Research Symposium. EPA-600/9-78-016. U.S. Environmental Protection Agency, Cincinnati, Ohio. pp. 233-244.

Duval, W.A., Jr., Atwood, R.A., Gallagher, W.R., Knight, R.G., and McLaren, R.I., 1979, FDD sludge-disposal manual: Research Project 786-1, Electric Power Research Institute, Palo Alto, California.

Federal Register, 1984, Recommended maximum contaminant levels for VOC's: U.S. GPO, Proposed 40 CFR, 141.50.

Freeze, A.R., and Cherry, J.A., 1979, Groundwater: Prentice-Hall, 604pp.

Griffin, R.A., and S.F.J. Chou., 1980, Disposal and removal of halogenated hydrocarbons in soils: Proceedings of the Sixth Annual

Research Symposium, EPA-600/9-80-010, U.S. Environmental Protection Agency, Cincannati, Ohio. 290pp.

Hem, J.D., 1985, Study and Interpretation of the Chemical Characteristics of Natural Water: United States Geological Survey Water-Supply Paper 2254, 263pp.

Houghton, R.L., and Berger, M.E., 1984, Effects of well casing composition and sampling method on apparent quality of ground water: Proceedings of the Fourth National Symposium and Exposition on Aquifer Restoration and Ground Water Monitoring, pp. 203-213.

Junk, G.A., Spalding, R.F., and Richard, J.J., 1980, Areal, vertical, and temporal differences in ground water chemistry: II. Organic constituents: Journal of Environmental Quality, v. 9, no. 3, pp. 479-483.

Kansas Department of Health and Environment, 1985, Program strategy addressing volatile organic chemicals in Kansas groundwater: KDHE Procedural Memorandum 85-1, 12pp.

Kansas Department of Health and Environment, 1986, Monitoring well-sampling guidelines: 9pp.

Keene, K.M., and Bayne, C.K., 1977, Groundwater from Lower Cretaceous rocks in Kansas: Kansas Geological Survey, Chemical Quality Series 5, 18pp.

Latta, B.F., 1950, Geology and ground-water resources of Barton and Stafford counties, Kansas: Kansas Geological Survey, Bulletin 88, 228pp.

Leenheer, J.A., Malcolm, R.L. McKinley, P.W., and Eccles, L.A., 1974, Occurrence of dissolved organic carbon in selected ground-water samples in the United States: U.S. Geological Survey, Journal of

Research, v. 2, no. 3, pp. 361-369.

Lin, Y.H., 1966, Acid and gas production from sanitary landfills: Ph.D. Thesis, West Virginia University, Morgantown, West Virginia. 95pp.

Malcolm, R.L., and Leenheer, J.A., 1973, The usefulness of organic carbon parameters in water-quality investigations: Proceedings no. 19, Institute of Environmental Science, Anaheim, California, pp. 336-340.

McCarty, P.L., and B.E. Rittman, 1981, Trace organics in ground water: Environ. Sci. and Tech. v. 15, pp. 40-49.

McCool, P., 1986, Testing for volatile organic chemicals: The Kansas Lifeline, Kansas Rural Water Association, pp. 20-21.

Metzler, S.M., 1981, Sanitary landfill management for ground-water quality protection: M.S. Thesis, University of Kansas, 158pp.

Miller, R.F., Randtke, S.J., Denne, J.E., and Hathaway, L.R., 1986, Evaluation of the trihalomethane formation potential of Kansas groundwaters: University of Kansas, research project in progress.

Remson, J.A., Fungaroli, A., and Lawrence, A.W., 1968, Movement in an unsaturated sanitary landfill: Journal of the Sanitary Engineering Division, ASCE, v. 94, no. SA2, pp. 307-317.

Tchobanoglous, G., Theisen, H., and Eliassen, R., 1977, Solid wastes-Engineering Principles and Management Issues: McGraw-Hill Book Company, 621pp.

United States Department of Agriculture-Soil Conservation Service, 1981, Soil survey of Barton County, Kansas: 117pp.

Walsh, J.J. and Kinman, R.N., 1979, Leachate and gas production under controlled moisture conditions. In: Municipal Solid Waste: Land

Disposal, EPA-600/9-79-023a, U.S. Environmental Protection Agency, Cincinnati, Ohio. pp.41-57.

CHAPTER 12: APPENDICES

Appendix 1-Geologic logs of monitor and domestic wells.

DW #1 19-13-12dbd. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1886 feet.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Silt, clayey, red-brown	8	8
Silt, clayey, brown, some red	12	20
Silt, clayey, tan-gray; contains some caliche nodules and coarse sand and fine gravel	15	35
Silt, clayey, tan-gray; contains some caliche nodules	21	56
Silt, clayey, sandy, tan-gray; contains some caliche nodules and a sandstone lens at 57.5 feet	3	59
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, silty, red-tan; contains abundant coarse sand and fine gravel	3	62
Clay, multicolored; contains some free sand	6	68
Sandstone, weathered; contains some gray sandy clay	12	80
Sandstone, gray, very porous	11	91
Clay, multicolored; contains sandstone lens at 96.5 feet	17	108
Sandstone, weathered, yellow-gold; contains some multicolored clay	4	112
Sandstone, white, tight	1	113
Clay, multicolored; contains bits of sandstone	8	121

DW #2 19-13-12bbd. Sample log of borehole; drilled by KGS in June 1986. Surface elevation 1882 feet.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Road fill	1	1
Silt, clayey, sandy; red-brown	27	28
Silt, clayey, sandy, gray and red-brown; contains a few caliche nodules	9	37
Silt, tan-brown; contains caliche and coarse sand and fine gravel	26	63

CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, red, sandy; contains fine gravel	2	65
Clay, white-buff	2	67
Clay, red-tan.............................	6	73
Clay, red.................................	2	75
Clay, tan-white...........................	6	81
Clay, brown...............................	6	87
Clay, sandy, brown........................	2	89
Clay, multicolored........................	18	107
Sandstone, weathered, fine grained, yellow-gold; contains some multicolored clay....	15	122

DW #3 19-13-12acd. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1886 feet.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Silt, clayey, gray-tan.....................	9	9
Silt, clayey, dark-brown...................	6	15
Silt, clayey, red-brown....................	4	19
Silt, clayey, sandy, tan-brown; contains caliche nodules beginning at 23.0 feet...	32	51
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, multicolored; contains fine to medium gravel in upper 5.0 feet................	6	57
Clay, tan-yellow...........................	7	64
Clay, gray.................................	5	69
Sandstone, weathered, yellow-gold; contains some multicolored clay..................	2	71
Clay, sandy, multcolored...................	5	76
Clay, multicolored.........................	5	81
Clay, sandy, multicolored..................	5	86
Clay, multicolored; contains sand lenses at 100 ft, 103 ft and 109 ft.............	20	106
Clay, sandy, multicolored..................	8	114
Sandstone, weathered, yellow-gold; contains multicolored clay......................	7	121

DW #4 19-13-12dcc. Driller's log of domestic well.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Topsoil....................................	2	2
Silt, clayey, brown........................	62	64
Silt, clayey, brown; contains broken sandstone.............................	9	73

CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, multicolored	37	110
Clay, multicolored; contains sandstone bits	46	156
Clay, multicolored	30	186
Sandstone; contains some clay streaks	44	230

DW #5 19-13-12aac. No log available; no water-quality sample taken.

DW #6 19-12-06ccb. No log available; domestic well.

DW #7 19-12-07bbb. Driller's log of domestic well.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Silt, clayey	150	150
CRETACEOUS-GULFIAN		
Dakota Formation		
Sandstone	23	173

DW #8 19-13-12ddd. No log available; domestic well.

DW #9 19-12-06bdd. Driller's log of domestic well.

	Thickness ft	Depth ft
CRETACEOUS-GULFIAN		
Dakota Formation		
Topsoil with multicolored clay	80	80
Sandstone, brown, tight	55	135
Sandstone, brown, soft	33	168
Pyrite, hard	8	176
Sandstone, brown, soft	19	195

DW #10 19-13-12bdd. Driller's log of domestic well.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Clay	10	10
CRETACEOUS-GULFIAN		
Dakota Formation		
Shale	70	80
Sandstone	44	124

DW #11 19-13-12ccd. Driller's log of domestic well.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		

Topsoil	3	3
Clay, brown; contains limestone	54	57
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay	63	120
Sandstone and clay	10	130
Clay	30	160
Clay and sandstone streaks	18	178
Clay	10	188
Sandstone and clay streaks	52	240

DW #12 19-13-14abb. Driller's log of domestic well.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Topsoil and brown clay	65	65
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, yellow-white	1	66
Sandstone, brown, soft	33	99
Clay, red-gray	9	108

SW #1 19-13-12dbd. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1886 feet.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Silt, clayey, dark brown	2	2
Silt, clayey, gray-brown	14	16
Silt, clayey, tan and light-brown; contains caliche beginning at 25.0	27	43
Silt, clayey, tan and light-brown; contains some fine gravel	14	57
Silt, clayey, tan and light-brown; contains some sand and fine gravel	2	59
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, sandy, multicolored; contains some broken sandstone and sand	2	61
Clay, sandy, multicolored; contains less broken sandstone and sand	5	66
Sandstone, yellow-white; contains some sandy clay	6	72

SW #2 19-13-12aca. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1862 feet.

	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Topsoil	2	2
Silt, clayey,yellow-brown	3	5
Silt, light tan	11	16
Silt, clayey, yellow	6	22
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, multicolored	4	26

SW #3 19-13-12acb. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1870 feet.

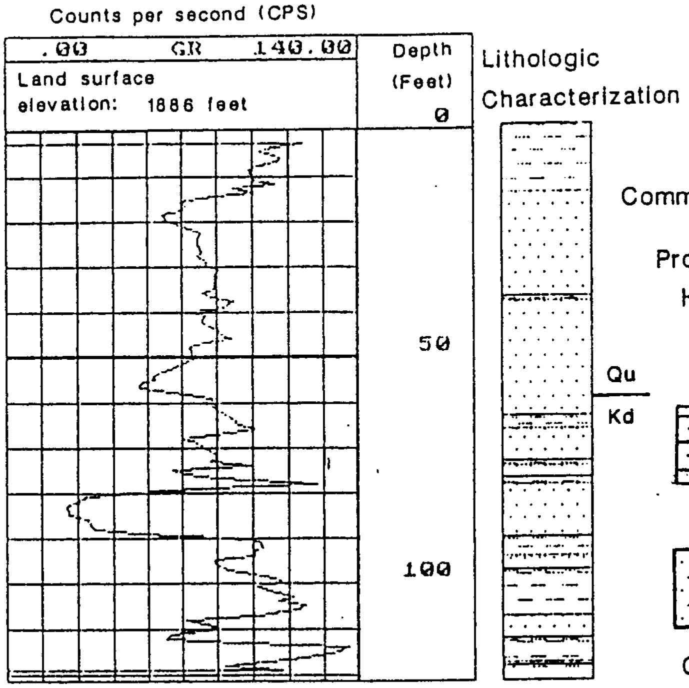
	Thickness ft	Depth ft
QUATERNARY-PLEISTOCENE		
Undifferentiated loess		
Topsoil	2	2
Silt, clayey, yellow-brown	6	8
Silt, clayey, light-tan; contains caliche nodules	1	9
Silt, clayey, light-tan; contains caliche nodules and occasional gravel with iron oxide at 23.5 ft	14	23
CRETACEOUS-GULFIAN		
Dakota Formation		
Clay, red	4	27

SW #4 19-13-12aab. Sample log of borehole; drilled by KGS in June 1986. Surface elevation, 1835 feet. Hole plugged from 20.0 ft to 50.0 ft with grout.

	Thickness ft	Depth ft
CRETACEOUS-GULFIAN		
Topsoil	1	1
Clay, red-brown	2	3
Clay, sandy, red-brown	2	5
Clay, red-brown with gray stringers	3	8
Clay, light-gray	5	13
Clay, multicolored	37	50

Appendix 2–Natural gamma ray logs.

DW#1



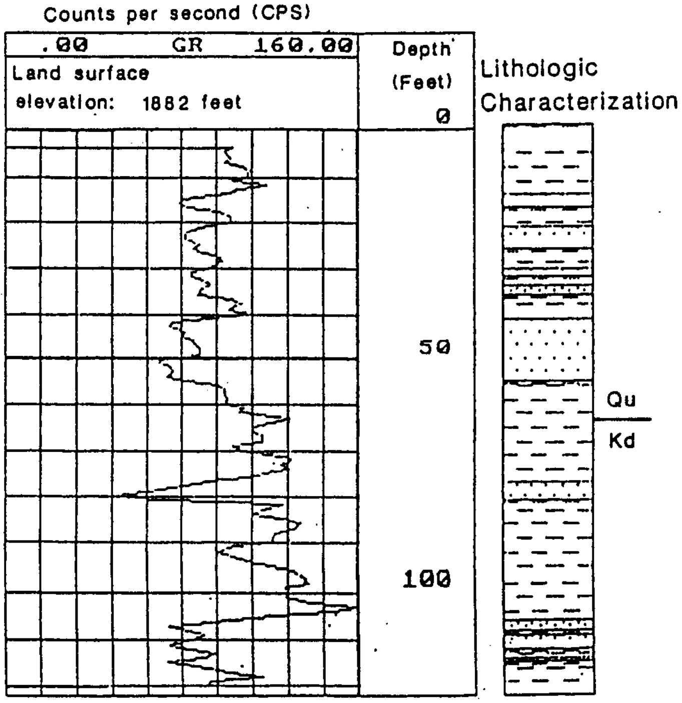
Common parameters for all logs:

Probe speed = 10 feet/minute
Horizontal displacement = 0
Time constant = 2 seconds

▤	Clayey zone
░	Sandy zone

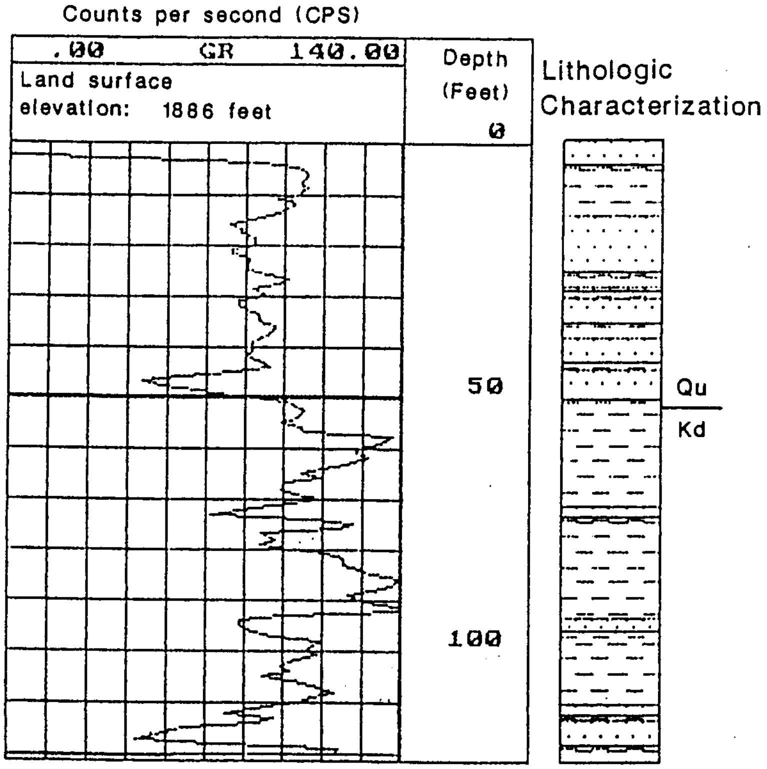
Qu = Undifferentiated Loess
Kd = Dakota Formation

DW#2

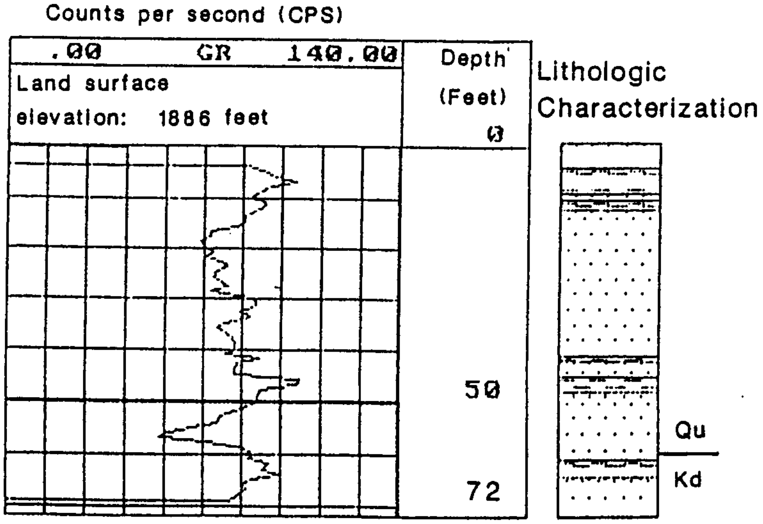


Note: The computer generated lithologic characterization columns are generalized and may not consistently agree with the geologic logs for the boreholes.

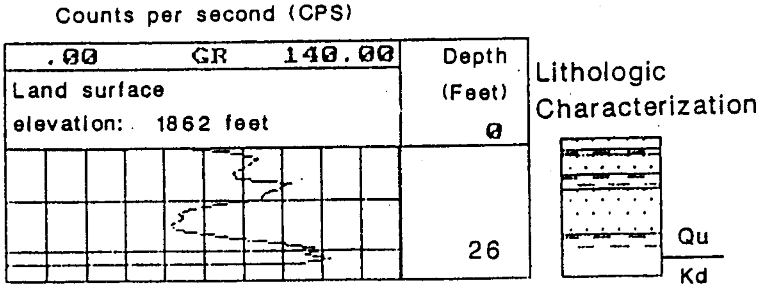
DW#3


Counts per second (CPS)
.00
GR
140.00
Land surface
elevation: 1886 feet
Depth
(Feet)
0
50
100
Lithologic
Characterization
Qu
Kd

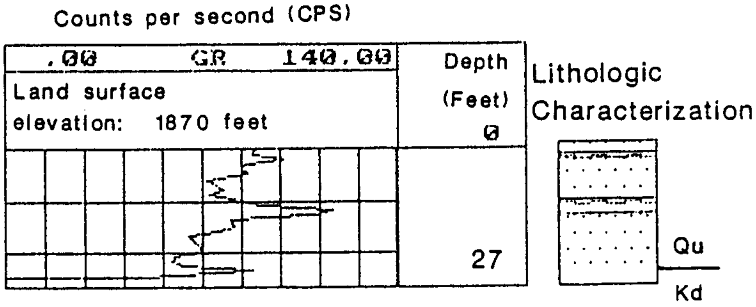
SW#1


Counts per second (CPS)
.00
GR
140.00
Land surface
elevation: 1886 feet
Depth
(Feet)
0
50
72
Lithologic
Characterization
Qu
Kd

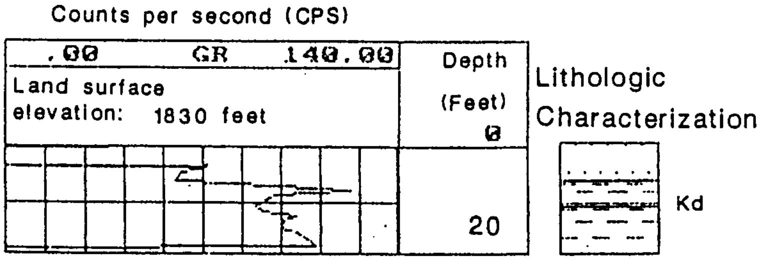
SW#2


Counts per second (CPS)
.00 GR 140.00
Land surface
elevation: 1862 feet
Depth
(Feet)
0
26
Lithologic
Characterization
Qu
Kd

SW#3


Counts per second (CPS)
.00 GR 140.00
Land surface
elevation: 1870 feet
Depth
(Feet)
0
27
Lithologic
Characterization
Qu
Kd

SW#4


Counts per second (CPS)
.00 GR 140.00
Land surface
elevation: 1830 feet
Depth
(Feet)
0
20
Lithologic
Characterization
Kd

Appendix 3–Monitor- and domestic-well construction specifications.

Well number	Well type	Well depth (feet)	Land surface elevation (feet)	Static water level (feet)	Water-level measurement date	Screen interval (feet)	Gravel-pack interval (feet)	Bentonite interval (feet)	Grout interval (feet)	Casing diameter (inches)	Casing material	Water-bearing zone
DW #1	M	121	1886	1827	July 1986	101-121	96-121	94-96	0-94	2	PVC	Dakota Sandstone
DW #2	M	122	1882	1816	July 1986	102-122	90-122	89-90	0-89	2	PVC	Dakota Sandstone
DW #3	M	121	1886	1821	July 1986	101-121	90-121	88-90	0-88	2	PVC	Dakota Sandstone
DW #4	D	230	1888	1827	May 1986	180-230	40-230	None	0-40	5	PVC	Dakota Sandstone
DW #6	D	NL	1830	NL	NL	NL	NL	NL	NL	NL	NL	Dakota Sandstone
DW #7	D	173	1875	1777	May 1986	153-173	?	None	4-14	5	PVC	Dakota Sandstone
DW #8	D	NL	1887	1817	NL	NL	NL	NL	NL	NL	NL	Dakota Sandstone
DW #9	D	195	1799	1792	October 1985	135-168 176-195	10-195	None	0-10	8	PVC	Dakota Sandstone
DW #10	D	124	1850	1804	May 1986	104-124	10-124	None	0-10	5	PVC	Dakota Sandstone
DW #11	D	240	1900	1801	November 1976	160-240	?	None	0-10	5	Styrene	Dakota Sandstone

DW #12	D	108	1910	1827	January 1986	80–100	74–108	None	0–10	5	PVC	Dakota Sandstone
SW #1	M	72	1886	1833	July 1986	52–72	47–72	45–47	0–45	2	PVC	upper aquifer
SW #2	M	26	1862	1852	July 1986	16–26	14–26	12–14	0–12	2	PVC	upper aquifer
SW #3	M	27	1870	1851	July 1986	17–27	15–27	14–15	0–14	2	PVC	upper aquifer
SW #4	M	20	1835	None	July 1986	10–20	8–20	6–8 20–23	0–6 23–50	2	PVC	Dakota clay no water

NL = No log available
PVC = Polyvinyl chloride
M = Monitor well
D = Domestic well

Appendix 4-Well-construction requirements; KDHE, 1986.

1. Six-inch steel or PVC protective casing with water-proof cap will be placed over the well. The casing should be set in a concrete pad and extend 36 inches above the ground line.

2. A 2-foot zone of bentonite pellets is to be placed on top of the gravel pack.

3. Gravel pack for wells may be sand, gravel, chat, or crushed sandstone meeting the requirements of Type U.D.-1. Source area for gravel pack must be identified.

4. Manufactured 4-inch well screen or horizontal-slotted schedule 40 or manual-slotted PVC schedule 40 shall be used. Maximum slot size 1/8 inch. Maximum vertical distance between slots is 6 inches.

5. Pipe specifications: Schedule-40 threaded PVC casing or better. No joint solvent should be applied.

6. Upper 10 feet of hole shall be grouted.

7. Prior to the drill rig entering the area of investigation, the drill rig, bits, rods, and any other hardware which could come into contact with the borehole or drilling fluid must be washed with hot potable water to remove any contaminants. The water pump and drilling-rig plumbing system should be thoroughly flushed with potable water to flush any contaminants present.

8. This decontamination procedure shall be used after the completion of each monitoring well to eliminate cross contamination between wells.

9. All casing used to complete monitoring wells shall be decontaminated by washing (inside and out) with biodegradable soap (i.e. Alconox), followed by a thorough rinsing with a potable-water supply.

10. During the construction of the monitoring wells, the use of lubricating oils or solvents on drill-rod threads should be eliminated or very minimal to avoid contamination of the ground water by hydrocarbon or organic solvents.

11. All drilling fluids (water) should be obtained from an offsite potable-water supply of known chemical quality. Samples should be collected from the drilling water source prior to intial drilling and submitted to the Kansas Department of Health and Environment for chemical analysis.

12. Flush-joint thread, 4-inch Schedule-40 PVC casing, has been specified to be used in all monitoring wells. The casing should be connected by the threads, hand-tight, with no solvent or cementing material used on the casing joints.

13. All drilling-fluid muds or additives shall be approved by the KDHE prior to use in the construction of the wells.

14. All wells shall be developed by pumping and flushing by the drilling contractor to sufficiently remove drilling fluids and muds. The wells must be developed until the pump-stream discharge is characteristic of the aquifer waters.

15. All wells shall be completed to the shale contact (40-45 feet in depth). All wells should be screened from the static water level to the shale contact. Any changes in the screening interval of the wells shall be made by the Department's representative present as the wells are being constructed.

16. The source of the gravel pack used to complete the monitoring wells must be approved by KDHE.

17. All grouting materials and grout mixtures must meet specifications defined in Article 30, Water Well Contractor License and Water Well Construction Regulations.