CHEMICAL EFFECTS OF SELECTED TRACE-METALS FROM SANITARY LANDFILL LEACHATES ON GROUND WATER QUALITY

by

Eric G. Vogl

B.S., University of New Hampshire, 1978

Submitted to the Department of
Geology and the Faculty of the
Graduate School of the University
of Kansas in partial fulfillment
of the requirements for the
degree of Master of Science

Thesis Committee

Ernest E. Angino

Chairman and Professor in Charge

L. R. Hathaway

Michael J. Brady

Committee Members

1981

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

ACKNOWLEDGEMENTS

At this point, I would like to thank my chief advisor and true friend, Dr. E. E. Angino for his encouragement, professional advice, and genuine interest in this project. Throughout the duration of my research, he has contributed significantly toward my personal and intellectual growth.

Without the support of my parents, Dr. Otto and Jane Vogl, the completion of my thesis would have been impossible. Throughout my life, they have exhibited continued concern in all my endeavors. They are truly the finest parents anyone could ever ask for.

I would also like to thank Drs. L. Hathaway, M. J. Brady, and R. A. Robison, the remaining members of my committee, for their guidance in my quest for a graduate degree in geology. They always found time for constructive criticism and helped enormously in the editing of my thesis. I owe my thanks to Dr. R. L. Kaesler for his help in all my work with the computer.

I thank Dr. A. M. Diaz, Mike Pope, and the U.S.G.S. ground water section for lending me essential equipment and offering helpful advice. I thank Dr. J. D. Zimbrick and the entire Radiation Biophysics Dept. for allowing me to use their E.P.R. instrument and for helping in the interpretation of the data.

The City of Lawrence was most kind in allowing me to collect ground water samples from both the upland and floodplain sanitary landfills.

Finally, I would like to dedicate this thesis to my fiancee and future wife, Lisa Smith. She aided me by typing my drafts and by providing emotional support throughout my two and one-half years at Kansas University.

ABSTRACT

Nationwide use of sanitary landfills as disposal sites for municipal wastes has increased concern for their capacity to contaminate local ground water systems. This matter stems from the fact that potentially dangerous leachate ultimately forms in and permeates regions adjacent to a landfill. Under favorable geohydrologic conditions, this solution may enter nearby aquifers used for drinking water. Two sanitary landfills, both in Lawrence, Kansas, were investigated as to the actual or potential impact of selected trace-metals (Cu, Co, Cr, Ni, Pb, Cd, Mn, Fe, and Zn) accumulated in these leachates on local ground water quality. In addition, pH, specific conductance, temperature, and dissolved oxygen were used to further identify potential hazards at these two landfills.

One landfill, southwest of Lawrence, is in upland area and has been out of use since 1969. The effluent from this landfill has general chemical characteristics and trace-metal content indicating an aerobic environment. There appears to be no contamination by this landfill of surrounding local ground water.

The other landfill, north of Lawrence, is in the Kansas River floodplain and has been in use since 1970. The leachate generated typically contained less dissolved oxygen, more dissolved solids, more soluble trace-metals, and is more acidic than leachate derived from the upland landfill. At the floodplain landfill, an anaerobic-aerobic

environment exists immediately surrounding the area of buried refuse. There is no evidence based on chemical constituents investigated that the floodplain landfill poses any threat to local ground water quality or to the water quality of the adjacent Kansas River.

INTRODUCTION

Sanitary landfills are superior to open dumps with regards to controlling fire, odor, vermin, and allowing site reuse. The effects of landfilling operations must be carefully monitored, because landfills can be potential sources of health and pollution hazards due to the character of the leachate that ultimately forms in and permeates the regions adjacent to such disposal sites. Leachate is defined as the aqueous solution containing decomposition products including gases and soluble compounds, which are accumulated in water moving through a landfill.

Contamination of ground water from refuse leaching can occur when: 1) a landfill is located adjacent to or over an aquifer; 2) a leachate is produced, and it enters the aquifer, and 3) a landfill is supersaturated with respect to soil moisture in comparison to the surrounding region. Supersaturation is caused by the flow of ground water derived from precipitation and surface runoff through the landfill site. A minor source of moisture is the water derived from the chemical decomposition of material in the landfill. It follows, therefore, that for a landfill to become a leachate generator, water must enter the landfill.

There are four ways by which water may enter a landfill (Degner, 1974): 1) water reaching the surface over the landfill percolates vertically down through the soil cover, 2) water from an adjacent source moves horizontally through the side walls of the landfill, 3) water enters from the bottom of the landfill due to a rise in the ground water table or by capillary action, and 4) water is present in the fill prior to or during placement of refuse material.

The magnitude of any contamination largely depends on the volume of leachate, concentration of chemical components in leachate, chemical characteristics of leachate, extent of migration of leachate, and the nature of the soil profile. In turn, these factors are controlled by the amount of refuse, the age of the landfill, the hydrogeology of the site, and the climate (Farquhar *et al.*, 1972).

The sanitary-landfill method of waste disposal began in the 1930's as an alternative to the open dump. Since then it has won acceptance by health authorities as the only viable means for land disposal (Salvato, 1971). To avoid any confusion, the American Society of Civil Engineers (1959) defined sanitary-landfilling as a:

> "...method of disposing of refuse on land without creating nuisances or hazards to public health or safety, by utilizing the principles of engineering to confine the refuse to the smallest practical area, to reduce it to the smallest practical volume, and to cover it with a layer of earth at the conclusion of each day's operation or at such more frequent intervals as may be necessary."

Evidence exists, however, that many operations do not fit these guidelines for a sanitary landfill.

Currently in the United States, sanitary landfills play a major role in the disposal of solid wastes. In fact, approximately 90% of all municipal solid wastes generated in the continental U.S. are discarded on land.

About half goes to open dumps and the remainder goes into landfills (Apgar and Langmuir, 1971). As landfills become more widely used in the future, they may become the major means for waste disposal. While providing adequate service and having an economic advantage over other means of disposal, they can at the same time be designed to handle nearly every common municipal and industrial waste.

During the past two decades, considerable and increasing attention has been focused on gaining knowledge of the trace element chemistry of both surface and subsurface waters. However, in many modern communities, aquifers used as water sources are potentially subject to either a drop in water quality or complete degradation as a result of man's activities.

The investigation reported here (made from August 3, 1979-August 12, 1980) involved field and subsequently laboratory studies of leachates derived from two sanitary-landfills. One is located southwest of Lawrence, Kansas and has been out of use for nearly a decade. The other is located north of Lawrence and has been used since 1970. The former is in an upland area, whereas the latter is in the Kansas River floodplain.

It is generally accepted that in aquatic environments, metals such as Cu, Co, Cr, Ni, Pb, Cd, Mn, Fe, and Zn in trace amounts play a significant role in inorganic and biologically controlled reactions. In trace quantities (less than 1.0 mg/l) many of these metal ions are essential micronutrients for living organisms (Korte et al, 1976), but in unnaturally high concentrations, they may be inhibiting or toxic to these same organisms. Toxic levels for selected metals may be as low as 1.0 mg/l or even less.

The objectives of this study were to measure and evaluate the actual or potential impact of selected trace-metals (Cu, Co, Cr, Ni, Pb, Cd, Mn, Fe, and Zn) accumulated in these leachates on local ground water quality in the vicinity of both the currently used Lawrence-Douglas County Landfill (floodplain site) and the abandoned Lawrence-Douglas County Landfill (upland site).

Comparisons of these trace-metal data are necessary to determine differences, if any, between leachate generated in a floodplain site as compared to one in an upland site. Another result obtained in the study from such a comparison is a hint as to some of the effects of landfill age on the trace-metal concentration of leachate produced from a 10-12-year-old landfill versus that from a younger one. A direct comparison is impossible, because the ground water chemistry at a landfill is site specific. In addition, an attempt was made to define the state of chemical equilibrium in each landfill system using data defining the chemical speciation of manganese. Finally seasonal and climatic variation in leachate production and chemistry were examined.

PREVIOUS WORK

Large scale studies of sanitary landfills as potential sources of environmental contamination began in the late 1960's and early 1970's. The United States Geological Survey, the Environmental Protection Agency (E.P.A.) and its predecessor agencies, and various state agencies were instrumental in initiating such studies in this area (e.g. Fungaroli, 1971; Boggess, 1975; and Robson, 1971). Together with the work of Langmuir (1972), these studies provide an overview of the general chemistry of landfill generated solutions (leachates) and provide criteria for location of future landfills.

Some of the most useful investigations within the past decade of landfill-site selection and evaluation have been conducted by the Illinois Geological Survey. Griffen (1976, 1977), made a detailed laboratory and field examination of the effects of clay minerals on the attenuation of pollutants (particularly metals) in landfills. Hughes (1976) summarized the methods for evaluating a landfill site considering the manner of disposal and the hydrologic environment. Lindorff and Cartwright (1977) wrote a helpful publication concerning problems and possible remedial actions for ground water pollution.

Rovers (1973, 1974) and Farquhar (1972) examined a number of landfills in Canada in an attempt to determine the relative influences of the geology of the site, ground water conditions, effect of age of a fill on ground water, and factors that affect leachate and gas production in sanitary-landfills in Canada.

Soil type may also profoundly affect leachate production and chemistry. Korte (1976) and Fuller (1976) examined the composition of soils of different compositions in relation to the migration of certain trace-elements (Cu, Fe, Mn, Zn). Olsen (1974) offered some insight for solutions to existing problems created by waste disposal in soils of Kansas. He also proposed guidelines for improving waste disposal in specific soils.

Baedecker (1979) summarized the chemical reactions and hydrogeologic processes occurring in a landfill, while Apgar (1971) described the potential for ground water pollution of a landfill located above the water table. In connection with this problem, Langmuir (1972) described the affect of pollution concentration of subsurface waters.

Little has been reported in the literature on the effects of bacteria and organic compounds on leachate formation and its chemistry at a landfill.

Minor research has been done on the landfills near Lawrence, Kansas. Brown (1971) provided a political and historical overview of establishing a waste-disposal site for Lawrence, Kansas. Degner (1974) examined the general and major element chemistry of the leachate at the Lawrence-Douglas County landfill (floodplain site) and speculated on their effect on ground water quality. Hathaway and Carr (1976) extended investigations conducted by McKinney and

Blackman (1973-1977) on the impact of buried solid waste on the chemical quality of ground water in the vicinity of the floodplain landfill.

U.S.E.P.A. (1979) and Skougstad et al. (1979) prescribed methods for the chemical analysis of inorganic components in water and wastes. Maienthal and Becker (1976) outlined sample handling and sampling techniques. Salvato (1971) discussed factors controlling and preventing sanitary-landfill leaching.

In summary, relatively little research has been done on the chemistry and oxidation state variation of trace metals in leachates from a sanitary-landfill. Furthermore, there has been no previous description of the physicochemical and hydrologic conditions at a landfill based on trace-metal data.

HISTORY OF SANITARY LANDFILL OPERATIONS IN LAWRENCE

Until 1967, the City of Lawrence, Kansas has used various methods at different sites for disposal of its solid wastes. Early in 1967, the City purchased a 120-acre tract of land west of Dragstrip Road for landfilling purposes. Much of this land had previously been used for quarrying. No plans for operation of this sanitary landfill were developed prior to the time when landfilling began in October of the same year. Due to poor planning, problems immediately arose. The refuse was not covered on a daily basis as required for a sanitary landfill. Moreover, rather than confining the refuse to a small area where it easily could be controlled, the refuse was spread over a large area. Complaints concerning blowing paper and loose trash soon became common and frequent. In short, it took less than a year for the City to degrade the new landfill site to such an extent that a total reevaluation of its operation was required.

Early in 1970, it was decided to move the landfill operation to a new site adjacent to the Kansas River. On July 20, 1970, the City of Lawrence began placing its municipal refuse in the floodplain-landfill north of Lawrence. At first, disposal was limited to a region that included an area referred to as a test cell. The City disposed of wastes to this area over a three week period. Excavation for the original test cell produced a pit, 42.2 m (145') long, 22.9 m (75') wide, and 2.4 m (8') deep, which was designed to contain approximately 1300 tons of waste. The function of the test cell was to provide a means for observation of a landfill operation situated in an environment of fluctuating ground water level and floods. The City stopped placing wastes at this site in August, 1970, at the request of the Corps of Engineers. After six months of observation, the Corps granted permission to resume landfilling operations in February, 1971 (McKinney and Blackman, 1973-1979).

With improved planning, stricter guidelines, and better organization, this landfill site has been in continuous use since 1971. At the present usage rate it appears that a new site will be needed within three years. Brown (1971) provided a more detailed account of the history of landfill operations in Lawrence, Kansas.

LOCATION AND DESCRIPTION

Upland Landfill:

The upland Lawrence-Douglas County landfill, which was abandoned in 1969, is located on a 120-acre tract of land. The location of this site is the N 3/4 SW 1/4, sec. 4, T. 13 S., R. 19 E., just west of Dragstrip Road. This area is about 12 km southwest of the Lawrence business district and is situated in what is physiographically termed an upland area. It is underlain by flat-lying beds of limestone with thin interbeds of shale. Thin clay-soil layers covering the bedrock distinguish the land of this region from either the floodplain or the terraces that are common in the Lawrence area. The elevation of the land ranges from 268 m (880') to 299 m (980') above mean sea level. Figure 1 shows the location of a few major cities and Douglas County in Kansas. Figure 2 indicates the location of the upland landfill in relation to Lawrence.

Floodplain Landfill:

The Lawrence-Douglas County floodplain landfill is located in an area best described as "a borrow pit" in a floodplain. Alluvial soils in the area have been excavated to build the North Lawrence Levee (Brown, 1971). The site contains approximately 242 acres in the S$\frac{1}{2}$ SW$\frac{1}{4}$, sec. 2, T. 12 S., R. 19 E., the W$\frac{1}{2}$ sec. 11, T. 12 S., R. 19 E., and the NE$\frac{1}{4}$ sec. 10, T. 12 S., R. 19 E. The landfill is located approximately eight km northwest of the Lawrence

Figure 1. Index map indicating the location of a few major cities and Douglas County in Kansas. Longitude and latitude in degrees (°).

LEGEND

— State line ○ Major city

--- County line

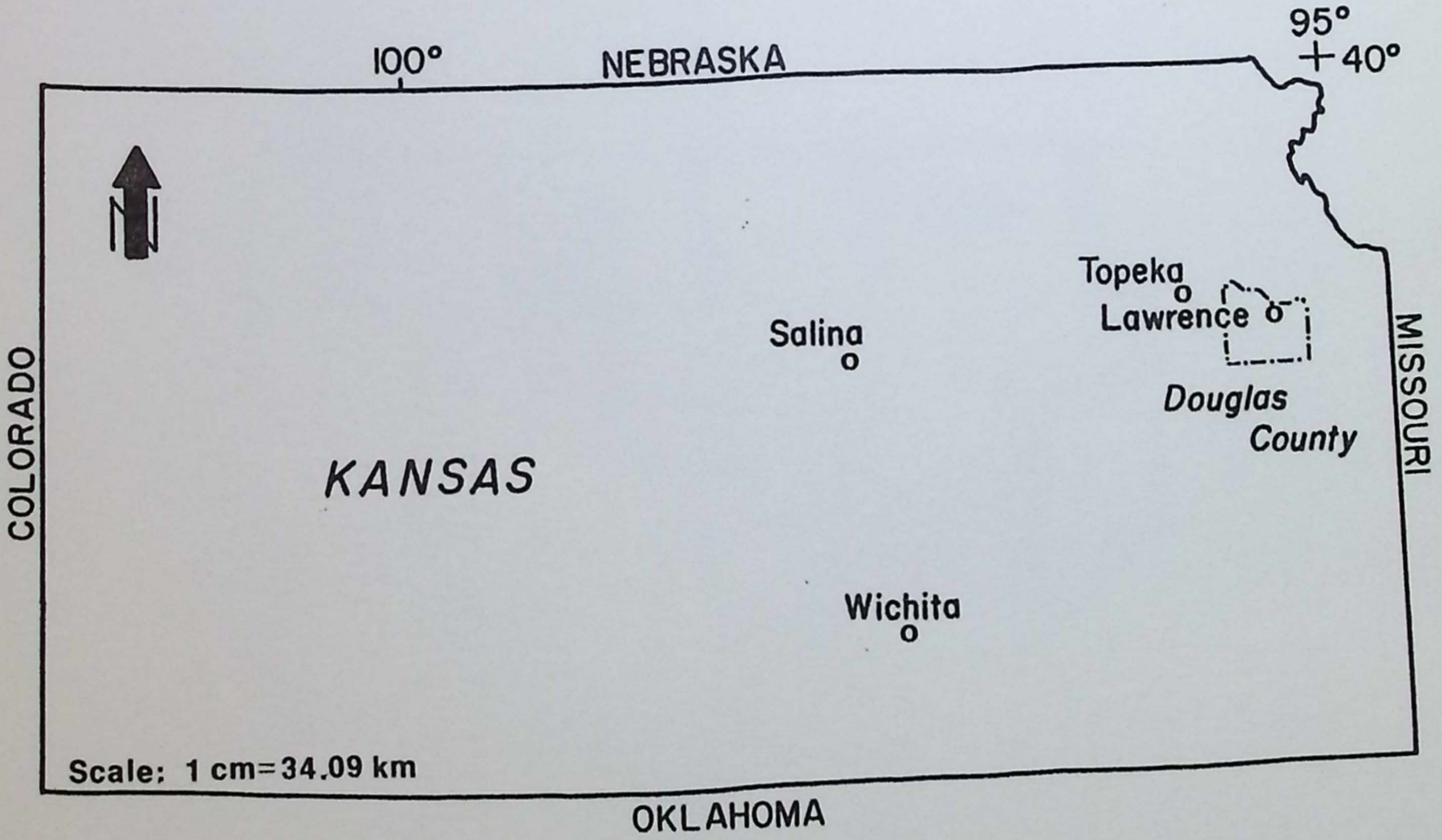

95°
100°
NEBRASKA
+40°
COLORADO
N
Topeka
Lawrence
Salina
MISSOURI
Douglas County
KANSAS
Wichita
Scale: 1 cm=34.09 km
OKLAHOMA

Figure 2. Locations of the upland and floodplain landfills in relation to Lawrence. City limits represented by solid line.

LEGEND

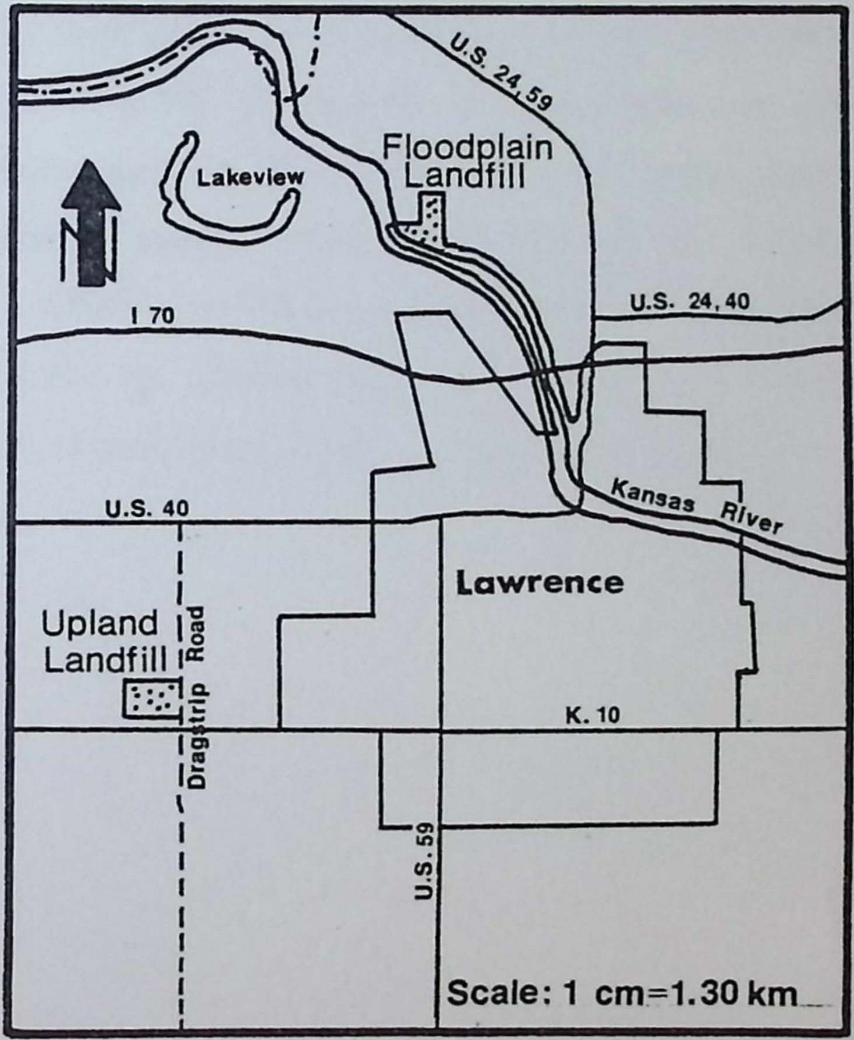
—Interstate

·—County line

--Dirt road

░ Landfill site

—Paved road


U.S. 24, 59
Floodplain Landfill
Lakeview
N
I 70
U.S. 24, 40
Kansas River
U.S. 40
Lawrence
Upland Landfill
Dragstrip Road
K. 10
U.S. 59
Scale: 1 cm=1.30 km

business district. Figure 2 defines the location of this disposal site relative to Lawrence, and Figure 3 depicts its location relative to U.S. Highways 24 and 59, the Jefferson and Douglas county boundaries, the Kansas River, and the Union Pacific Railroad.

This site, located in Grant Township, Douglas County, is owned by the City of Lawrence. It is bounded on the north by cultivated farmland, on the south and west by the Kansas River, and on the east by a drainage canal along the North Lawrence Levee. The elevation of the terrain ranges from 247 m (812') to 254 m (832') above mean sea level. The floodplain is characterized by recent stream deposits of unconsolidated clay, silt, sand, and gravel.

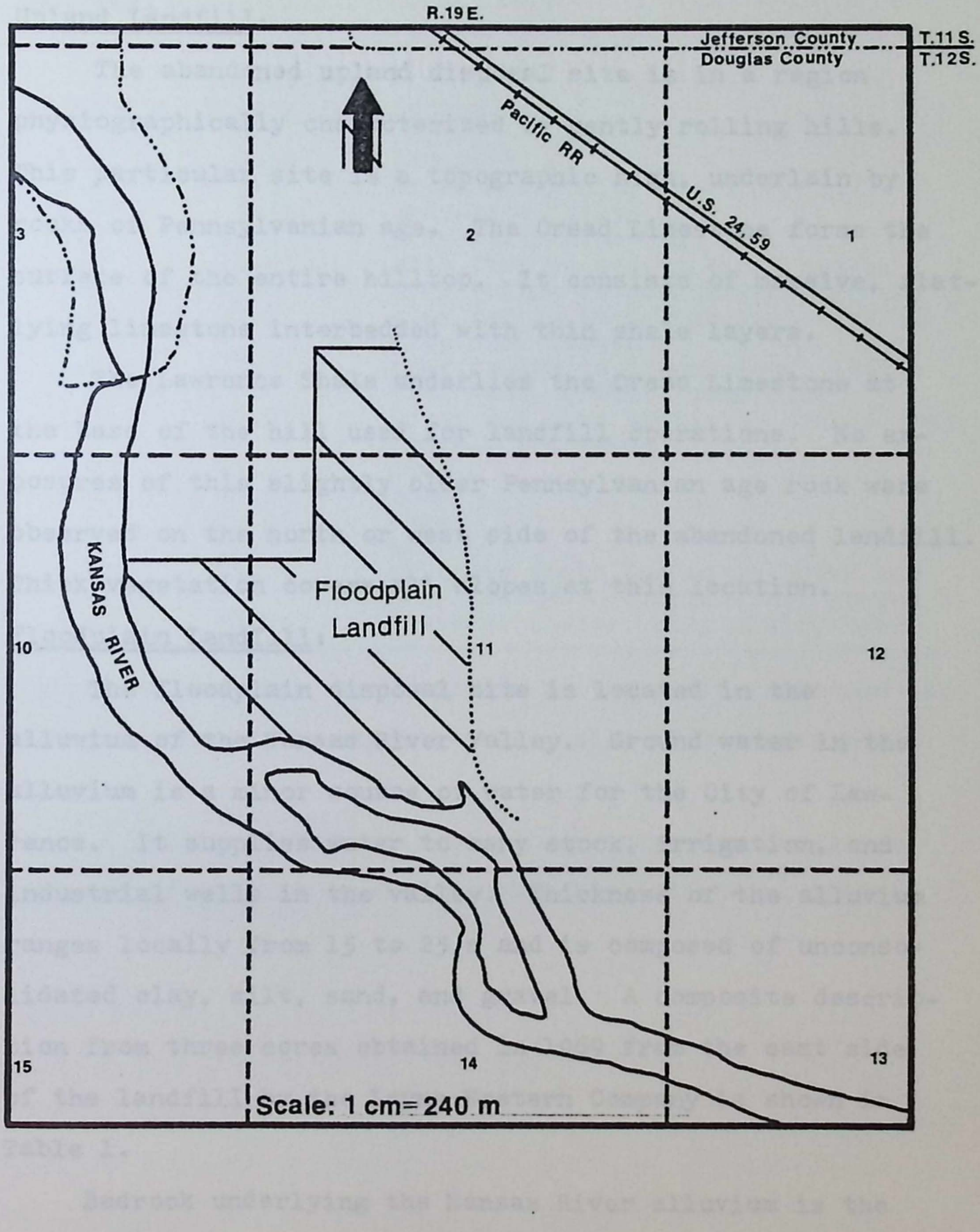
Figure 3. Location of the floodplain landfill relative to U.S. Highways 24 and 59, the Jefferson and Douglas County boundaries, the Kansas River, and the Union Pacific Railroad.

LEGEND

—U.S. 24 & 59   --Section line

-..County line   ·····Levee

↦Union Pacific Railroad


R.19 E.
Jefferson County
Douglas County
T.11 S.
T.12 S.
Pacific RR
U.S. 24, 59
3
2
1
KANSAS RIVER
Floodplain Landfill
10
11
12
15
14
13
Scale: 1 cm= 240 m

GEOLOGY

Upland Landfill:

The abandoned upland disposal site is in a region physiographically characterized by gently rolling hills. This particular site is a topographic high, underlain by rocks of Pennsylvanian age. The Oread Limestone forms the surface of the entire hilltop. It consists of massive, flat-lying limestone interbedded with thin shale layers.

The Lawrence Shale underlies the Oread Limestone at the base of the hill used for landfill operations. No exposures of this slightly older Pennsylvanian age rock were observed on the north or west side of the abandoned landfill. Thick vegetation covers all slopes at this location.

Floodplain Landfill:

The floodplain disposal site is located in the alluvium of the Kansas River Valley. Ground water in the alluvium is a minor source of water for the City of Lawrence. It supplies water to many stock, irrigation, and industrial wells in the valley. Thickness of the alluvium ranges locally from 15 to 25 m and is composed of unconsolidated clay, silt, sand, and gravel. A composite description from three cores obtained in 1969 from the east side of the landfill by the Layne Western Company is shown in Table 1.

Bedrock underlying the Kansas River alluvium is the Lawrence Shale, a part of the Douglas Group of Pennsylvanian age. The hills both north and south of the Kansas River

Table 1. Composite core description from three cores drilled on the east side of the floodplain disposal site in 1969.

Thickness	Cumulative Thickness	Description
0.5'	0.5'	top soil
5.0'	5.5'	brown silty clay
4.0'	9.5'	gray-brown silty clay
5.5'	15.0'	gray silty fine sand
5.0'	20.0'	gray fine to medium sand, gravel
5.0'	25.0'	gray medium to course sand, gravel
5.0'	30.0'	gray medium to course sand, gravel gray clay lenses
5.0'	35.0'	gray medium to course sand, little fine sand, gravel, some clay
5.0'	40.0'	brown and gray medium to course sand, gravel, boulders
5.0'	45.0'	same
5.0'	50.0'	same
5.0'	55.0'	same
5.0'	60.0'	same
5.0'	65.0'	thin tan shale lenses in brown and gray medium to course sand, gravel
5.0'	70.0'	Lawrence Shale: gray shale, hard

Valley at Lawrence are underlain by the Oread Limestone and younger beds of the Shawnee Group, also of Pennsylvanian age.

A detailed study of the geology and ground water resources of Douglas County is available (O'Connor, 1960).

SAMPLING SITE DESCRIPTIONS

Upland Landfill:

The locations of surface water sampling sites 1 to 5 for the abandoned upland-landfill are shown in Figure 4. Site 1 is a standing body of water at the northeast section of the landfill. Site 2 is located at the head of the drainage ditch that runs along the northwestern and western perimeter of the landfill. Its purpose is to catch any leachate draining from the upland landfill in a westerly direction and route it away from Yankee Tank Lake. Site 3 is located in a small, marshy area east of the drainage ditch where leakage of water from the upland landfill is evident. Site 4 is situated on the northeastern shore of Yankee Tank Lake, and site 5 is located at the southwestern corner of the abandoned landfill, near the end of the drainage ditch.

Floodplain Landfill:

Specific sample and control well locations used for sampling at the floodplain landfill are shown in Figure 5. In addition, locations of sample and control wells in relation to the test cell are shown in this figure. Descriptions of sample wells S-3, S-4, S-5, S-6, S-9, S-10, S-11, and S-13 are from Degner (1974). Since 1974, several wells have been damaged by nearby construction equipment used at the landfill. These wells are no longer suitable for use for studies of this type. Furthermore, new sample wells have been added. For these two reasons,

Figure 4. Location of surface water sampling sites 1 to 5 at the upland landfill.

LEGEND

--	Dirt road	~	Contour line (ft)
░	Water body	⁙	Water-filled pit
≈	Dirt path	●	Sampling site
-----	Creek	▩	Clinton Reservoir Water Treatment Facility

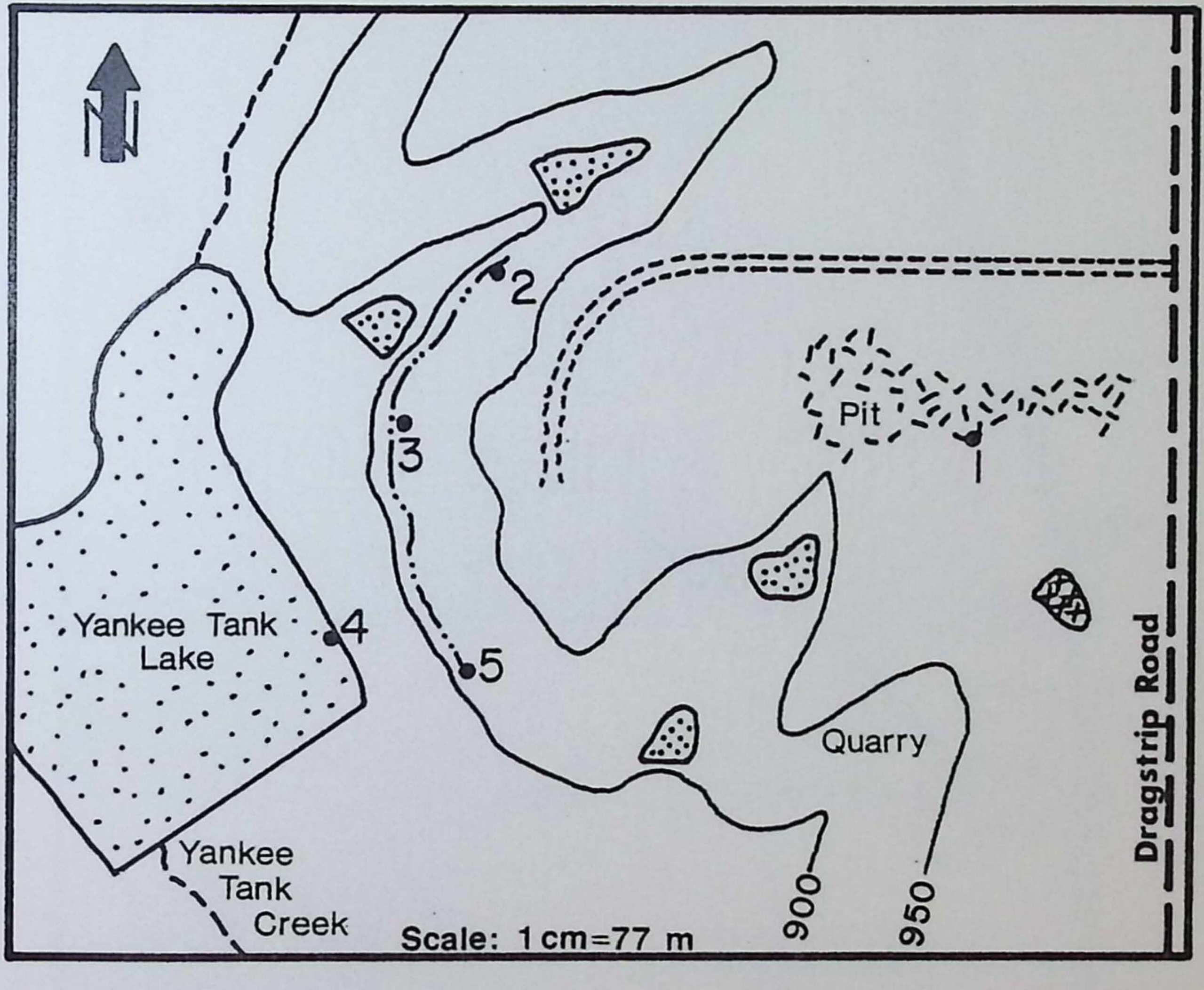
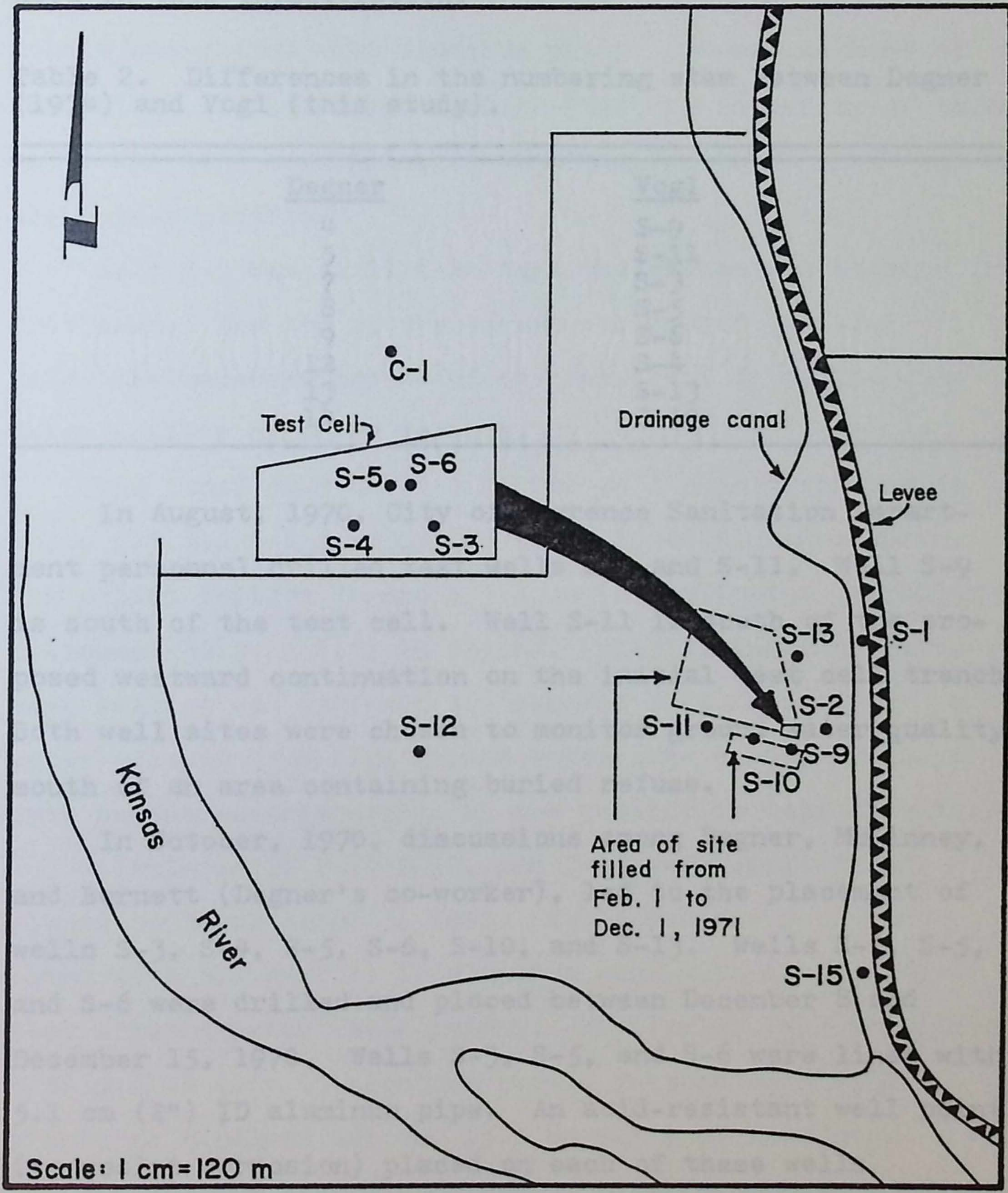

N
2
3
Pit
1
Yankee Tank Lake
4
5
Quarry
Dragstrip Road
Yankee Tank Creek
Scale: 1 cm=77 m
900
950

Figure 5. Location of test wells used for sampling at the floodplain landfill. "●" indicates test well location.


C-1
Test Cell
S-5
S-6
S-4
S-3
Drainage canal
Levee
S-13
S-1
S-2
S-11
S-9
S-10
S-12
Kansas River
Area of site filled from Feb. 1 to Dec. 1, 1971
S-15
Scale: 1 cm=120 m

a new numbering system has been employed in this study. Table 2 lists the differences in the numbering schemes of the wells between that given in Degner (1974) and the one used in this investigation.

Table 2. Differences in the numbering stem between Degner (1974) and Vogl (this study).

Degner	Vogl
4	S-9
5	S-11
7	S-3
8	S-5
9	S-6
12	S-4
13	S-13
14	S-10

In August, 1970, City of Lawrence Sanitation Department personnel drilled test wells S-9 and S-11. Well S-9 is south of the test cell. Well S-11 is south of the proposed westward continuation on the initial test cell trench. Both well sites were chosen to monitor ground water quality south of an area containing buried refuse.

In October, 1970, discussions among Degner, McKinney, and Burnett (Degner's co-worker), led to the placement of wells S-3, S-4, S-5, S-6, S-10, and S-13. Wells S-3, S-5, and S-6 were drilled and placed between December 8 and December 15, 1970. Wells S-3, S-5, and S-6 were lined with 5.1 cm (2") ID aluminum pipe. An acid-resistant well point (to resist corrosion) placed on each of these wells measure 61 cm (2') in length. The well point consists of four different metallic materials. The nose of the well point is forged steel. The inside shell of the well point is 5.1 cm (2") ID galvanized steel pipe. The gauze screening is brass, and the jacket surrounding the gauze is type 304 stainless steel. The ungalvanized portion of the well points was coated with aluminum paint. It was assumed at the outset of this investigation that the integrity of these wells has been maintained so that samples taken from them are representative.

Well S-3 was drilled so that the bottom was beneath the test cell. The top of the screen was placed 9.1 cm (.3') below the measured bottom of the test cell to monitor the water quality directly beneath.

The lower 30.5 cm (1') of the well points for sample wells S-5 and S-6 were sealed with liquid aluminum paste. The sealed portion formed a 5.1 cm (2") diameter, circular cylinder, 30.5 cm (1') in height. This cylinder or catchment section was designed to trap and hold flood water or water trapped from high ground water levels when the test cell became supersaturated at the respective well elevations.

Sample wells S-4, S-10, and S-13 were drilled and placed on March 4 and 5, 1971. Wells S-4 and S-10 were lined with 5.1 cm (2") ID aluminum pipe with 61 cm (2')-long acid-resistant well points. Similarly, S-13 was lined with 5.1 cm (2") ID aluminum pipe. Instead of a well point, an aluminum cup and 63.5 cm (25") of fine mesh (8 openings per linear centimeter) aluminum screen was fabricated onto the bottom end of a 61-cm (2')-long slotted section of aluminum pipe.

Sample wells C-1, S-1, S-2, S-12, and S-15 were drilled and cased by the Kansas Geological Survey during 1974 and 1975. Their casing consists of 10.2 cm (4") ID PVC pipe (Hathaway and Carr, 1976).

Table 3 summarizes the depth drilled, depth cased, and interval slotted or screened for the floodplain landfill wells.

Table 3. Summary of drilling data for floodplain test wells.

Well Number	Depth Drilled m.	ft.	Depth Cased m.	ft.	Depth of Interval Slotted or Screened m.	ft.
C-1	11.73	38.5	11.43	37.5	2.29-11.43	7.5-37.5
S-1	11.74	38.5	11.74	38.5	5.64-11.74	18.5-38.5
S-2	14.64	48.0	13.79	45.2	8.30- 9.52	27.2-31.2
S-3	4.58	15.0	4.58	15.0	3.97- 4.58	13.0-15.0
S-4	4.91	16.1	4.91	16.1	4.30- 4.91	14.1-16.1
S-5	3.72	12.2	3.72	12.2	3.11- 3.72	10.2-12.2
S-6	3.08	10.1	3.08	10.1	2.47- 3.08	8.1-10.1
S-9	6.10	20.0	6.10	20.0	5.19- 6.10	17.0-20.0
S-10	6.10	20.0	6.10	20.0	5.49- 6.10	18.0-20.0
S-11	7.35	24.1	7.35	24.1	6.44- 7.35	21.1-24.1
S-12	5.49	18.0	5.49	18.0	2.44- 5.49	8.0-18.0
S-13	6.44	21.1	6.44	21.1	5.83- 6.44	19.1-21.1
S-15	9.46	31.0	9.46	31.0	3.26- 6.19	10.7-20.3

SAMPLING PROCEDURE

This investigation of the abandoned upland and active floodplain landfills covered the period from August 3, 1979 to August 12, 1980. Attempts were made to sample once every three weeks, but some sampling occurred at four- or five-week intervals. These breaks in the collection interval were due to incliment weather, unavailability of field equipment, or malfunctions of field equipment.

Water-sample-collection and storage techniques described by Skougstad et al. (1979) and U.S.E.P.A. (1979) were followed. Polyethylene bottles (150 ml) were chosen as sample containers. To minimize contamination, each bottle was soaked for at least 48 hours in distilled 7N hydrochloric acid, then rinsed profusely with redistilled water prior to sample collection.

Ground water temperature measurements were made on all leachate samples immediately after withdrawal from each test well at the floodplain landfill. The same determinations were made in situ for ground water samples collected at the upland landfill.

Specific conductance measurements were made with a Lab Line Mark 4 portable meter subsequent to each temperature measurement for samples collected at the floodplain landfill. The same determinations were made upon arrival to the laboratory for upland landfill samples.

Hydrogen ion activity (pH) measurements were made in the laboratory for upland landfill samples and in the field for floodplain landfill samples with a Leeds and Northrup 7417 pH/specific ion/mV portable instrument.

Dissolved oxygen (D.O.) determinations were made _in situ_ before making any other measurements at all sampling sites on the four sampling dates from June through August, 1980. A Yellow Springs Instrument (Y.S.I.) Model 51 Dissolved-Oxygen Meter coupled with an oxygen-temperature sensing probe with a 15.24 m (50') lead was employed to make field measurements. The saturated-water-standardization procedure recommended on the back of the instrument were followed prior to each of the sampling days when dissolved oxygen data were recorded.

To minimize contamination the field equipment was thoroughly rinsed with a dilute HCl solution before a sample was collected. Subsequent to this the same equipment was rinsed with the sample leachate solution to be taken, before that sample was collected.

Upland Landfill:

At each of the five sampling sites, temperature and dissolved oxygen (D.O. only from June-August, 1980) were measured, appearance and odor observations were made, and a water sample was taken. Specific conductivity and pH measurements were made upon arrival at the laboratory. Subsequently each sample was filtered in the laboratory through a 0.45 u membrane filter to assure that only dissolved trace-metal constituents were measured. According to several state and federal agencies that collect surface and ground water data, filtrate passing through a 0.45 u membrane filter is assumed to represent dissolved components. Yet some colloidal material will still pass through membranes of this site. Since an examination of certain dissolved inorganic constituents was the purpose of this study, a 0.45 u membrane filter was chosen.

Once filtered, two subsamples were taken from each sample filtrate and put in sealed sample bottles (150 ml). One subsample remained unacidified to be used for analysis by a Varian Electron Paramagnetic Resonance (E.P.R.) unit. Addition of acid to this subsample at this point would defeat the purpose of the E.P.R. analysis. The other subsample was acidified with 1.0 ml of approximately 7N redistilled nitric acid. This fixed the solution at a pH of approximately 1.5 and also oxidizes the sample.

<u>Floodplain Landfill</u>:

For the floodplain disposal site, samples were taken from existing test wells. Because anoxic conditions exist within the subsurface zone penetrated by most of these wells, precautions were taken to avoid contamination and minimize any physicochemical changes that might occur between the time of sampling and laboratory examination. The following field sampling procedure was employed:

1. The well cap was removed, and the distance to the ground water was measured with a clean tape measure.

2. Well water was withdrawn with a clean hand bailer. The bailer was constructed from a 61 cm (2') section of

2.54 cm (1")-diameter polyvinyl chloride pipe with a 2.54 cm (1") check valve solidly molded to one end. The bailer was attached to a 9.15 m (30'), 0.64 cm ($\frac{1}{4}$")-diameter nylon rope to complete the assembly. One (310 m) or two full bailers (620 ml) of water were taken from each well before the samples for chemical analyses were taken.

3. Immediately after the sample was withdrawn, the temperature, pH, and specific conductivity were measured (prior to filtration). Temperature determinations were made when D.O. measurements were taken before the water sample was withdrawn.

4. In order to assure that only dissolved trace metal constituents were measured, each sample was then filtered through a 0.45 u membrane filter.

5. Once filtered, two 150-ml-subsamples were aliquoted again, one acidified with 1.0 ml of 7N redistilled nitric acid, the other was not acidified.

6. Appearance and odor observations were made on the well water and then recorded.

7. The well cap was replaced.

This entire procedure was repeated for each test and control well on each sampling date.

ANALYTICAL TECHNIQUE

Atomic Absorption:

A Perkin-Elmer Model 360 atomic absorption spectrometer was employed in the flame mode to determine the concentrations of iron, manganese, and zinc (mg/l concentration range). This same unit coupled with a Perkin-Elmer HGA high-temperature graphite furnace was used to measure ug/l concentration range levels of copper, chromium, cobalt, nickel, lead, and cadmium in the leachate samples. For clarity, 1 mg/l equals 1000 ug/l.

The analytical parameters employed for each element measured are summarized in Table 4. Standard operating conditions and instrument settings were used throughout the study (E.P.A., 1979 and Perkin-Elmer, 1973).

Electron Paramagnetic Resonance:

E.P.R. was employed to determine manganese speciation, because the instrument only detects Mn in the reduced Mn^{+2} form. The magnetic field (measured in gauss), time constant, scan range, scan time, and modulation were set at 3250, 0.5, 1000, 4 min., and 8, respectively. The gain was varied, depending on the Mn^{+2} concentration present in the sample. The detection limit for Mn^{+2} using this technique is 1.0 mg/l. Operating parameters for the E.P.R. were obtained experimentally. An attempt was made to detect Cu^{+2}, but its concentration in the leachate samples proved to be below the Cu^{+2} detection limit. Fe^{+2} was also measured, but the data obtained proved unreliable and

Table 4. Analytical operating parameters*. "--" indicates not calculated. "RSD" indicates Relative Standard Deviation in percent at the given concentration level.

Element	Standard Concentration Range	Technique	Sample Size	Detection Limit**	RSD 1 ug/l	RSD 25 ug/l
Cd	1.0-10.0 ug/l	AA-furnace	20 ul	0.1 ug/l	61%	--
Co	5.0-50.0 ug/l	AA-furnace	20 ul	1.0 ug/l	38%	2%
Cr	5.0-50.0 ug/l	AA-furnace	20 ul	1.0 ug/l	51%	2%
Cu	5.0-50.0 ug/l	AA-furnace	20 ul	1.0 ug/l	32%	1%
Pb	5.0-50.0 ug/l	AA-furnace	20 ul	1.0 ug/l	69%	3%
Ni	5.0-50.0 ug/l	AA-furnace	20 ul	1.0 ug/l	47%	2%
Fe	0.5- 5.0 mg/l	AA-flame	5 ml	0.03 mg/l	--	--
Mn	0.5- 3.0 mg/l	AA-flame	5 ml	0.01 mg/l	--	--
Zn	0.1- 0.5 mg/l	AA-flame	5 ml	0.005 mg/l	--	--

*For details, refer to U.S.E.P.A. (1979) or Perkin Elmer (1973).

**Detection limits from U.S.E.P.A. (1979).

erroneous. This was due to the rapid rate at which Fe^{+2} oxidized and precipitated from some sample solutions after exposure to surface conditions, temperature, pressure, and atmospheric oxygen concentrations.

Blanks:

Control blanks were run for each trace metal analysis, to account for any contamination of samples from chemical preparation, from any of the equipment, or from the washing procedure. Lead and cadmium were the only metals that yielded blank concentrations greater than the detection limit. Those values greater than the detection limit were subtracted from sample concentrations obtained from the appropriate calibration curve.

Construction of the Calibration Curve:

Initially, the standard method of additions (see Perkin-Elmer, 1973) was employed to obtain concentration levels of the selected trace metals in the sample solutions. When these values were compared to those obtained from the construction of graphs of absolute concentration versus peak height or absorption, there was no difference. Therefore, standard curves were constructed without using the standard method of additions.

For atomic-absorption analyses, three runs were made for each element per standard solution. The mean of the three values obtained was recorded. Later graphs of the atomic absorption signal mean-peak height for the graphite furnace mode (or mean-absorption value for the flame mode) versus the standard concentrations were plotted for each element studied. To illustrate the uncertainty of concentrations at the 1 ug/1 concentration level compared with that at the 25 ug/1 concentration level, relative standard deviations were calculated and are shown in Table 4. For each element the maximum standard deviation of peak height after three runs was used in these calculations. Concentration levels of 1 ug/1 and 25 ug/1 were chosen arbitrarily. The maximum standard deviation observed for each element during this one year investigation was divided by the concentration level and multiplied by 100. Concentrations for Cd were not observed at the 25 ug/1 concentration level; this calculation has been omitted from Table 4. Three readings at a given concentration level using the atomic absorption flame mode could not be significantly differentiated from the scale readings on the available instrument. Because the standard deviation of these three readings was zero, the absorption values plotted as points against known concentrations of Mn, Fe, and Zn. A typical calibration curve of peak height versus known cobalt concentration for the graphite furnace mode is shown in Figure 6. Error bars are also indicated. A typical calibration curve for flame mode absorption versus known iron concentration is shown in Figure 7.

The calibration curves were constructed with a Hewlett-Packard 9862A calculator plotter. Once these linear curves were obtained, the calculator was programmed to

Figure 6. Typical calibration curve for Co. Error bars are shown. Peak height in mm and concentration in ug/l.

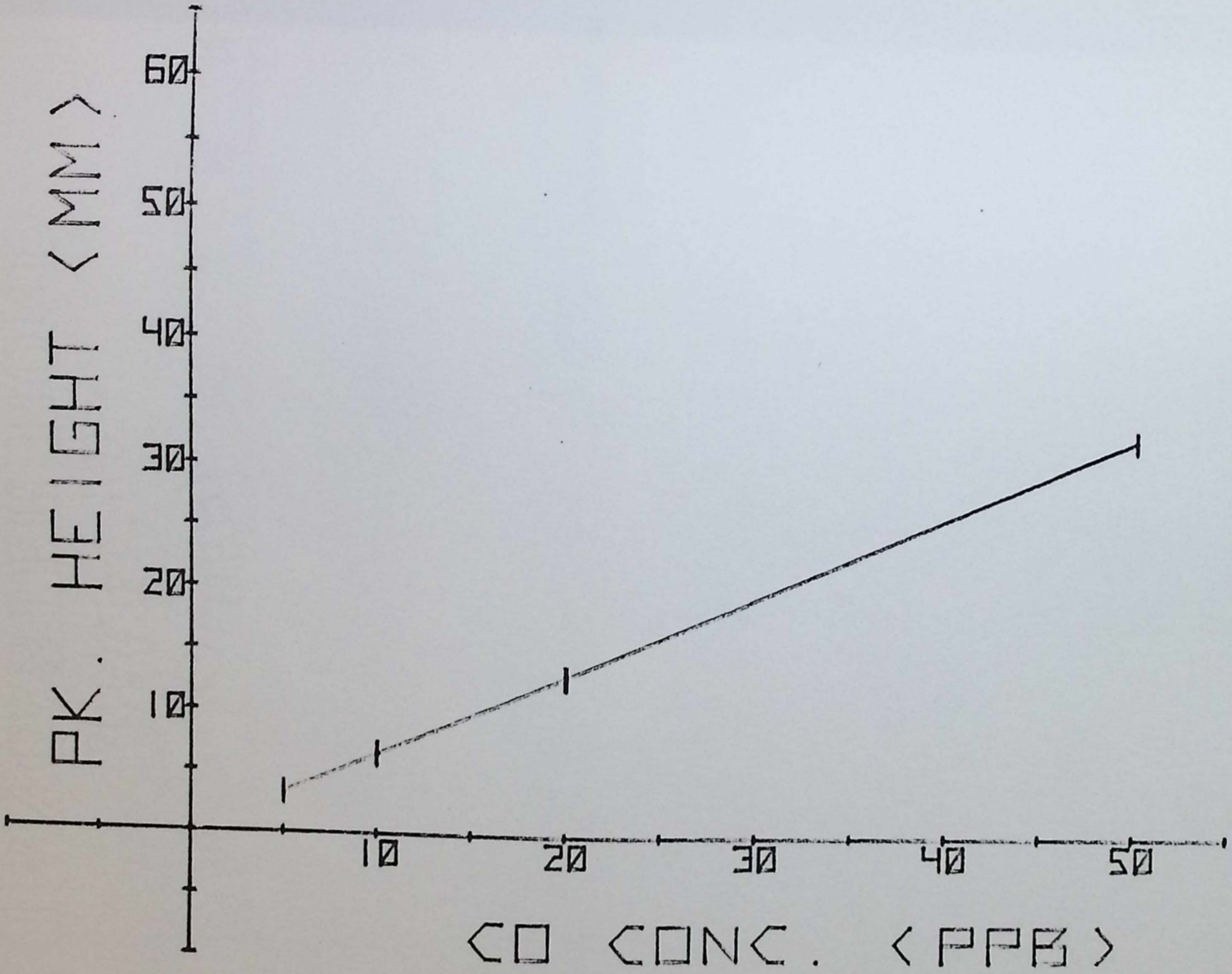
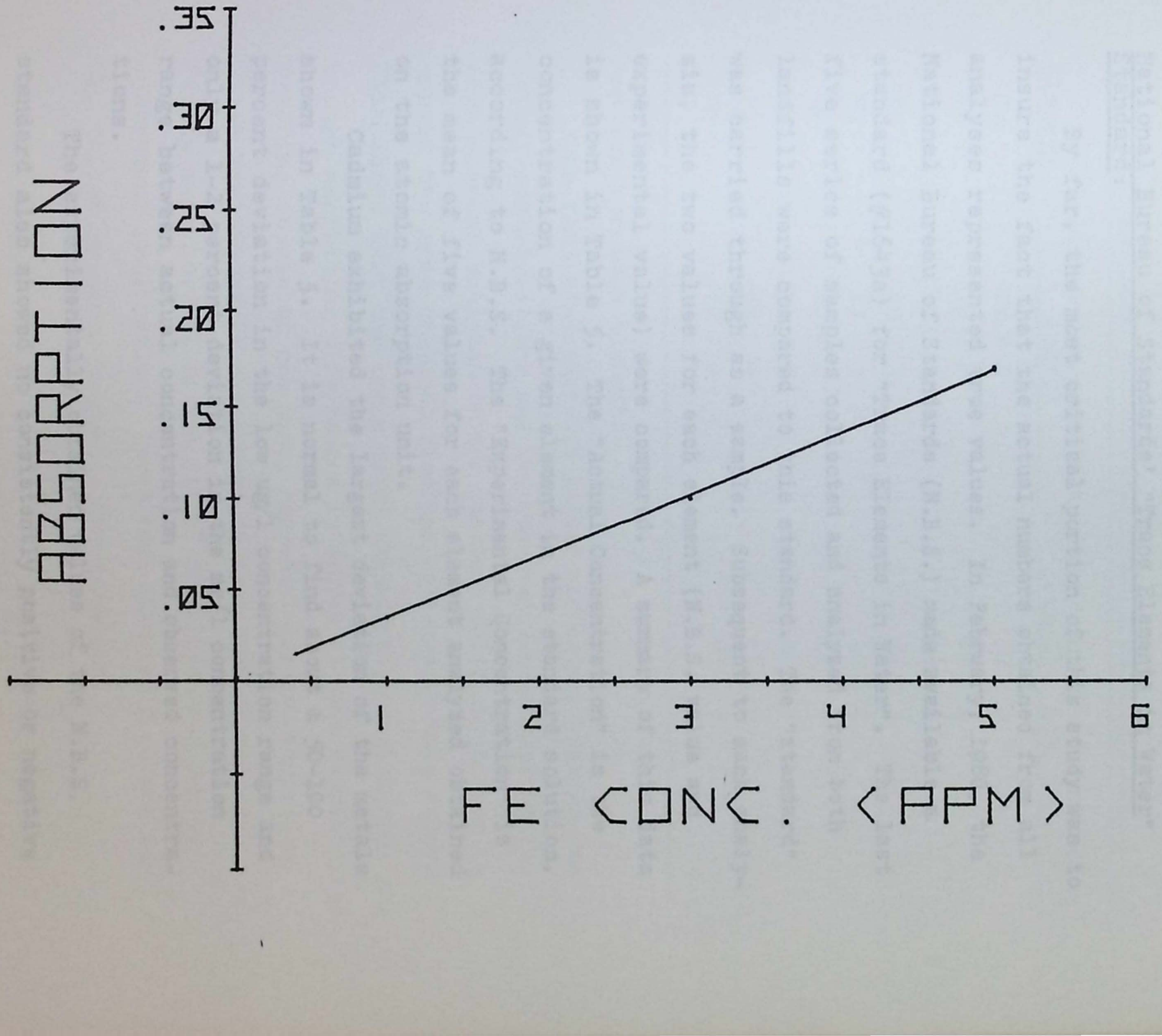

PK. HEIGHT (MM)
60
50
40
30
20
10
10
20
30
40
50
CO CONC. (PPB)

Figure 7. Typical calibration curve for Fe. Absorption in absorption units and concentration in mg/l.


.35
.30
.25
.20
.15
.10
.05
ABSORPTION
1
2
3
4
5
6
FE CONC. (PPM)

output unknown concentrations, given the known peak heights or absorptions.

National Bureau of Standards' "Trace Elements in Water" Standard:

By far, the most critical portion of this study was to insure the fact that the actual numbers obtained from all analyses represented true values. In February, 1980, the National Bureau of Standards (N.B.S.) made available a standard (#1643a) for "Trace Elements in Water". The last five series of samples collected and analyzed from both landfills were compared to this standard. The "standard" was carried through as a sample. Subsequent to each analysis, the two values for each element (N.B.S. value and experimental value) were compared. A summary of this data is shown in Table 5. The "Actual Concentration" is the concentration of a given element in the standard solution, according to N.B.S. The "Experimental Concentration" is the mean of five values for each element analyzed obtained on the atomic absorption unit.

Cadmium exhibited the largest deviation of the metals shown in Table 5. It is normal to find about a 50-100 percent deviation in the low ug/l concentration range and only a 1-2 percent deviation in the mg/l concentration range between actual concentration and observed concentrations.

The experimentally obtained values of the N.B.S. standard also showed no consistently positive or negative deviation. In fact, the deviations appeared randomly high

Table 5. Comparison of N.B.S. standard solution actual versus experimental metal concentrations. Actual value provided by N.B.S. Experimental value determined by atomic absorption in the laboratory.

Element	Actual Concentration (ug/l)	Experimental Concentration (ug/l) ± Std. Dev.
Cu	18 ± 2	17 ± 2
Co	19 ± 2	19 ± 3
Cr	17 ± 2	16 ± 2
Ni	55 ± 3	56 ± 3
Pb	27 ± 1	27 ± 2
Cd	10 ± 1	11 ± 4
Mn	31 ± 2	31 ± 3
Fe	88 ± 4	89 ± 3
Zn	72 ± 4	73 ± 3

and low for each element.

The fact that there is little difference between the actual concentrations and the observed concentrations for each element analyzed in the standard solution perhaps is one of the most critical findings obtained in this study. This shows that the trace metal concentrations obtained, although exhibiting wide ranges in values at most sample sites over the course of one year, represent natural variation. These variations do not appear to have been caused by faulty experimental or analytical technique or a faulty instrument calibration. Occasional detectable values for Cr, Co, or Pb could be due to contamination.

RESULTS

So few leachate samples were taken from sample wells S-3 (two samples) and S-12 (one sample) that no further mention will be made of these wells. The raw data that were obtained from these wells are shown in Appendices A and B. Well S-3 was dry all but two times during sampling. Well S-12 was damaged by landfilling equipment after the first sampling date and was not available thereafter.

Throughout the remainder of this report, groupings of sample sites will be used for convenience. The logical choice of groups is as follows: upland landfill sites 1, 2, 3, and 5; upland landfill control site 4 (Yankee Tank Lake); floodplain landfill wells C-1 and S-15 (both controls); floodplain landfill wells S-4, S-5, S-6, S-9, S-10, and S-11 (within the landfill where refuse has been buried); and floodplain landfill wells S-1, S-2, and S-13 (east of and adjacent to the region of buried refuse).

Specific Conductance:

The raw specific conductance data are shown in Appendix A. Conductance measures the ability of salts dissolved in water to transmit electrical energy. The more salts in the water, the greater is its conductance. The salts leached from the sanitary landfills tend to reflect the impact of leachate, by increasing the conductance.

Conductance shows a wide range of fluctuation, both among sampling sites and over a time span of approximately one year. The mean specific conductance (measured in umhos at 25°C for the upland landfill samples) from August 3, 1979 to August 12, 1980, and the standard deviations for each well are shown in Table 6.

The upland landfill sampling sites (1-5) clearly reflect influence of surface shallow-ground water that contains relatively low total dissolved solids. Sites 1, 2, 3, and 5 show slightly higher conductance values than those obtained from Yankee Tank Lake (site 4). This implies that leaching from the upland landfill may be affecting local ground water quality although on a small scale.

Control wells C-1 and S-15 reflect the lowest specific conductance values measured in the floodplain region. Two possible explanations exist as to why these values are not considerably lower. First, because both these wells are among the deeper wells at the floodplain landfill, the ground water from them could have had longer contact with minerals at depth, thus increasing the quantity of salts in solution. Second, these two wells are located near farming areas, and the high conductivity could reflect the impact of poor agricultural practices on ground water quality.

Within the floodplain landfill leachate from sample wells S-4, S-5, S-6, S-9, S-10, and S-11 show a marked increase in contamination on the basis of conductance. All these wells are located in the area that was filled with refuse within the past decade. This clearly indicates that even after ten years soluble material is still being mobilized in this landfill.

Table 6. Summary of specific conductance and pH data; "n" refers to the number of measurements taken at a given site.

n	Sampling Site	Mean Specific Conductance (umhos)	Standard Deviation	Mean pH	Standard Deviation
13	1	368	48	7.9	0.5
11	2	768	216	7.3	0.3
4	3	970	42	7.5	0.2
4	5	620	154	7.5	0.4
13	4	264	36	7.7	0.2
13	C-1	1653	379	6.9	0.3
9	S-15	1948	1189	7.1	0.2
13	S-1	2002	1316	7.0	0.3
13	S-2	2402	1162	7.1	0.3
10	S-4	2544	1114	6.8	0.3
13	S-5	3328	1127	7.0	0.3
13	S-6	3517	1500	6.9	0.2
13	S-9	2651	782	7.0	0.1
13	S-10	2298	976	7.0	0.2
13	S-11	2310	983	7.1	0.2
13	S-13	2202	1324	7.6	0.6

Adjacent to and east of this previously filled area, leachate from wells S-1, S-2, and S-13 exhibit intermediate conductance values. It appears that some leachate has reached and contaminated these wells. The form of the soluble salts is unknown, but their volume and concentration is certainly not as great as within the landfill. Dilution by the drainage canal running on the east side of the landfill could contribute to lower these values.

Examination of all sampling site conductance data indicates a tendency for conductance to increase following precipitation. Clearly some contaminants enter the migrating ground water system more easily at high-water levels.

pH:

Theoretically, sanitary landfills tend to become strongly anaerobic with significant acid production when wet. Sometimes, perhaps like at the floodplain landfill, depressed pH values may be masked by the generation of ammonia. The production of this gas tends to raise the pH slightly. Because the ground water in the region of both the upland and the floodplain landfills is slightly basic (pH 7.3-7.6), due to the presence of limestone extending throughout the area, pH measurements may be an indicator of contamination in the ground water both in and around the vicinity of the sanitary landfills. It should be realized that limestone acts to rapidly neutralize acidity away from the landfill.

It is noteworthy that natural variations in pH occur.

This is most likely related to exposure to air and the time required before measurements are made. As the temperature in the sample rises, CO_2 is released. This raises the pH and leads to the precipitation of Fe oxides and hydroxides that tend to scavenge other metals such as Mn.

Appendix A shows the pH values for all the sampling sites from both landfills. Mean annual pH values for surface and shallow ground water the upland landfill ranged from 7.3 to 7.9, as seen in Table 6. This suggests little, if any acid contamination from this landfill. It could also indicate that any acid present is being buffered by the natural carbonate in the system. In either case, there is probably little possibility that any metal contaminants, as simple cations, are being leached into the ground water from the buried refuse. This projection is based on data derived from surface - shallow ground water samples.

Ground water from control wells C-1 and S-15, at the floodplain landfill have mean annual pH values of 6.9 and 7.1 respectively. Although not exceptionally acidic, these relatively low values could result from entrapped CO_2 and the subsequent formation of carbonic acid at depth in these two relatively deep wells.

The shallower wells (S-4, S-5, S-6, S-9, S-10, and S-11), within the landfill show leachate with mean annual pH values as low as 6.8. This strongly suggests some acid contamination.

Leachate samples from S-1, S-2, and S-13 show mean annual pH values of 7.0, 7.1, and 7.6, respectively. This may suggest slight acid contamination in the region adjacent to and just east of the buried refuse.

In general, pH values in the vicinity of the floodplain landfill are not exceptionally low, indicating the possibility that chemical conditions in the ground water passing through the floodplain landfill are not strongly acidic as one would initially speculate. In fact, at certain distinct sampling sites and at certain times during the year, slightly basic ground water was sampled.

Temperature:

Temperature data for each sampling site from both the upland and floodplain landfills are summarized in Appendix A. According to Zanoni (1972), "usually temperature is not a restricting factor even in northern latitudes since the interior of the refuse mass is insulated from the ambient temperature and the degradation process is exothermic in character." In order to speed up microbial activity, water temperature in the range of 15° to 25°C are necessary. As the temperature drops, bacteria are less active, thus consuming less oxygen. As a result, the leachate formed is more aerobic with decreasing temperature for a given moisture content. In these respects, temperature determinations can aid in evaluating a sanitary landfill.

Thus, ground water temperatures are directly related to seasons of the year. Warmest ground water temperatures were obtained in late August, coldest temperatures in late

February and early March, with intermediate temperatures obtained in fall and spring.

Physical Appearance:

All ground water collected from the upland landfill and from wells C-1, S-15, S-1, and S-2 at the floodplain landfill was clear and odorless at the time of collection. Water from well S-13 showed some turbidity, mostly silt, but was odorless. Water collected from wells S-4, S-5, S-6, and S-9 invariably were very turbid and had a distinct odor of H_2S and NH_3. Often the water from some of these wells had a slight grayish character upon sampling but produced "iron red" precipitates upon being returned to the laboratory and allowed to stand at room temperature. Leachate from wells S-10 and S-11 exhibited slightly less turbidity and odor.

These physical characteristics further confirm the fact that surface-shallow ground water at the upland landfill is relatively free of contamination. However some contamination is clearly leaching from the floodplain landfill. It does not appear to be adversely affecting the ground water quality adjacent to and surrounding the landfill, at least on the basis of ground water characteristics examined thus far.

Trace-Metals: Cu, Co, Cr, Ni, Pb, Cd, Mn, Fe, and Zn:

Trace metal data are given in Appendix B. Once the complete set of data was collected for the period August 3, 1979 to August 12, 1980, an attempt was made to statistically analyze the data for significant differences among and within the certain groups of wells mentioned earlier. Variability in trace metal concentrations over the course of one year for each metal at each site proved to be great. In fact, it was so great that it exceeded the basic assumptions for use of the statistical tests available. This variability is likely due to seasonal changes in precipitation and temperature, but is not otherwise accounted for.

Metal concentrations in the leachate samples fluctuated throughout the year. The most marked activity occurred following major precipitation. Not all metals behaved identically and some not even similarly. This is realistic because a peak in all metal concentrations would not be reached at the same time at all sampling sites in any given landfill. Furthermore leachate composition is not vertically or laterally homogeneous in composition, because of the inhomogeneity of the buried refuse. Therefore, due to the inhomogeneity of the leachate and the irregular occurrence of precipitation, leachate slugs of highly variable metal concentrations pass through the subsurface at a landfill. Such a situation precludes a thorough statistical investigation.

At this point it is necessary to reemphasize the uncertainty of those metal concentrations that fall at or near the detection limit. Refer to Table 4 and observe the high relative standard deviation (RSD) of concentrations at 1 ug/l and the lower RSD of concentrations at the 25 ug/l level. Table 7 shows the minimum and maximum concentrations for selected trace metals and each sampling site during one year. The total number of samples taken is shown by "s", and "d" refers to the number of times a concentration was read above the detection limit. A mean and standard deviation was calculated only when no values for a specified element at a single sample site fell below the detection limit. Median values are listed for metal species where at least one value was observed to fall below the detection limit. The data must be treated this way to minimize bias which might otherwise be introduced. Table 7 does not include trace metal data for Co, Cr, Ni, and Pb from all sites at the upland landfill and for Cr from well C-1 at the floodplain landfill. In these cases, there were fewer than four values that read above the detection limit. As a result, a mean was not calculated. Instead, histograms were constructed (see Figure 8) to illustrate the frequency distribution of concentrations of these elements.

Copper:

Mean concentrations for Cu from the upland landfill surface water ranged from 6 ug/l (site 2) to 8 ug/l (median at site 1 and mean at site 3), only slightly higher than the median concentration of 6 ug/l at the control site. Mean concentrations for Cu from the floodplain landfill well water also exhibit little difference between the control wells (5 ug/l and 6 ug/l), wells within the landfill (6 ug/l to 11 ug/l), and wells adjacent to the landfill (9 ug/l to

Table 7. Summary of trace metal data based on sampling during the year August 3, 1979 to August 12, 1980. Means, medians and standard deviations rounded to nearest significant place. "--" indicates not calculated. "ND" indicates below the detection limit. Cu, Co, Cr, Ni, Pb, and Cd concentrations in ug/l. Mn, Fe, Zn concentrations in mg/l.

Sample	Element	s	d	Min. Conc.	Max. Conc.	Mean Conc.	Standard Deviation	Median
1	Cu	13	12	ND	24	--	--	8
	Cd	13	11	ND	3	--	--	1
	Mn	13	13	0.02	0.55	0.30	0.18	
	Fe	13	13	0.03	0.30	0.15	0.08	
	Zn	13	13	0.02	0.38	0.14	0.14	
2	Cu	11	11	2	18	6	5	
	Cd	11	11	1	2	2	1	
	Mn	11	11	0.02	0.95	0.25	0.27	
	Fe	11	11	0.11	0.30	0.21	0.06	
	Zn	11	11	0.03	0.32	0.09	0.08	
3	Cu	4	4	4	12	8	4	
	Cd	4	4	1	2	2	1	
	Mn	4	4	0.05	0.25	0.10	0.09	
	Fe	4	4	0.06	0.30	0.16	0.12	
	Zn	4	4	0.01	0.05	0.03	0.02	
5	Cu	4	4	5	11	7	3	
	Cd	4	4	1	4	3	2	
	Mn	4	4	0.02	0.05	0.04	0.02	
	Fe	4	4	0.11	0.22	0.18	0.05	
	Zn	4	4	0.01	0.15	0.05	0.07	
4	Cu	13	12	ND	9	--	--	6
	Cd	13	13	1	2	2	1	
	Mn	13	13	0.03	0.07	0.04	0.01	
	Fe	13	13	0.06	0.25	0.12	0.07	
	Zn	13	13	0.01	0.17	0.06	0.06	
S-1	Cu	13	13	5	42	10	10	
	Co	13	6	ND	8	--	--	1
	Cr	13	8	ND	6	--	--	1
	Ni	13	11	ND	47	--	--	11
	Pb	13	12	ND	37	--	--	10
	Cd	13	13	1	3	2	1	
	Mn	13	13	0.02	0.19	0.08	0.05	
	Fe	13	13	0.06	0.85	0.32	0.24	
	Zn	13	13	0.03	1.65	0.27	0.46	
S-2	Cu	13	13	5	24	11	6	
	Co	13	10	ND	9	--	--	3
	Cr	13	9	ND	8	--	--	1

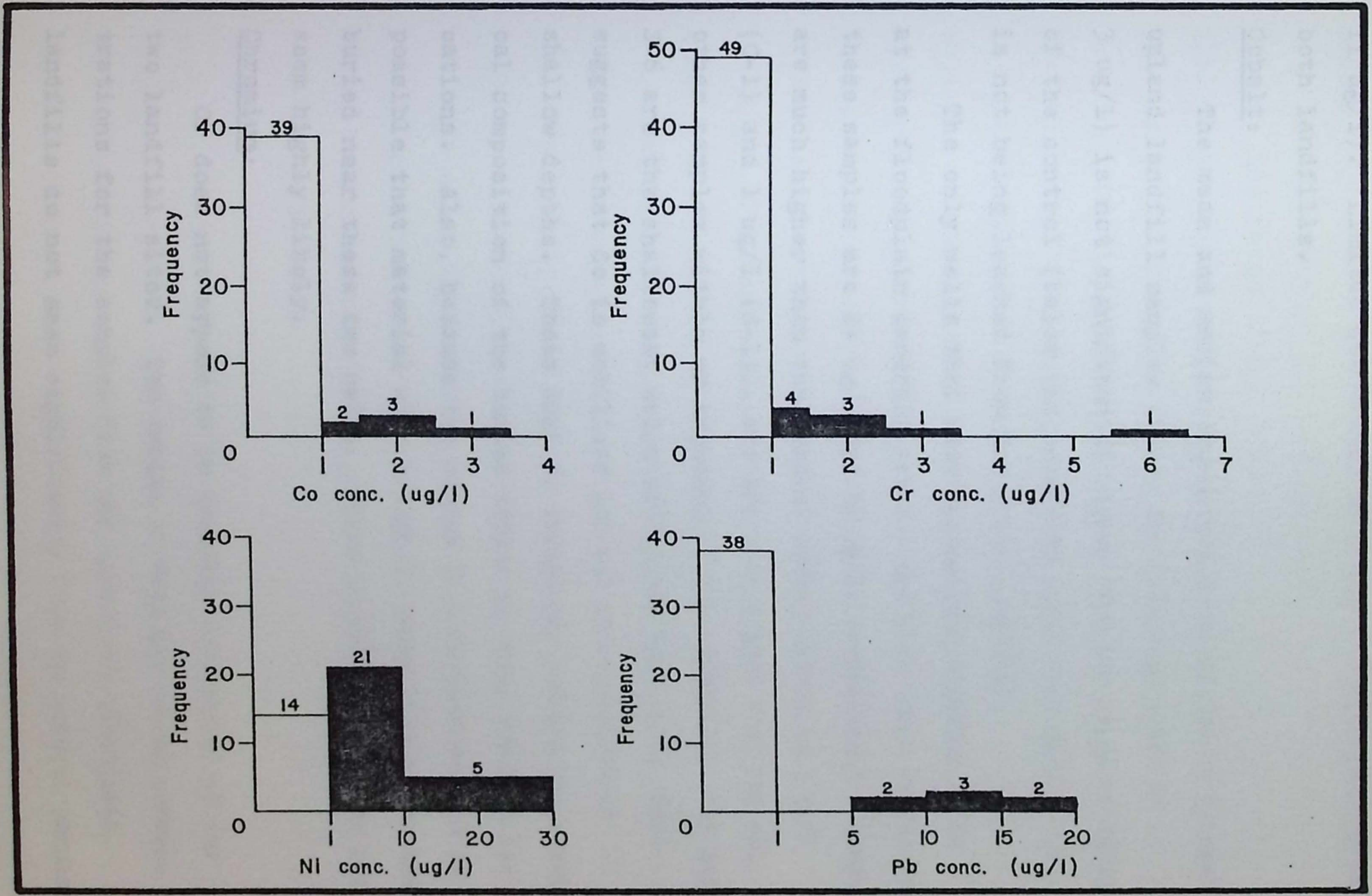
Table 7 (cont'd.)

Sample	Element	s	d	Min. Conc.	Max. Conc.	Mean Conc.	Standard Deviation	Median
S-2	Ni	13	13	3	46	20	11	
	Pb	13	13	3	41	20	17	
	Cd	13	13	1	2	2	0	
	Mn	13	13	0.11	1.32	0.76	0.45	
	Fe	13	13	0.05	0.60	0.25	0.15	
	Zn	13	13	0.03	0.95	0.13	0.25	
S-13	Cu	13	13	4	18	9	5	
	Co	13	5	ND	3	--	--	1
	Cr	13	4	ND	7	--	--	1
	Ni	13	13	3	35	12	8	
	Pb	13	13	3	25	13	9	
	Cd	13	13	1	4	2	1	
	Mn	13	13	0.47	1.21	1.03	0.19	
	Fe	13	13	0.23	6.40	2.76	2.36	
	Zn	13	13	0.01	0.71	0.14	0.21	
S-4	Cu	10	10	5	24	10	5	
	Co	10	10	4	9	6	2	
	Cr	10	6	ND	2	--	--	1
	Ni	10	10	4	65	29	21	
	Pb	10	10	7	34	15	9	
	Cd	10	10	1	4	2	1	
	Mn	10	10	1.03	2.96	2.00	0.65	
	Fe	10	10	1.22	46.00	17.60	12.02	
	Zn	10	10	0.32	1.76	0.89	0.43	
S-5	Cu	13	13	3	19	9	4	
	Co	13	13	8	39	24	9	
	Cr	13	10	ND	6	--	--	2
	Ni	13	13	14	88	33	21	
	Pb	13	13	2	29	14	9	
	Cd	13	13	1	4	2	1	
	Mn	13	13	0.68	2.55	1.20	0.48	
	Fe	13	13	6.30	46.00	22.46	13.08	
	Zn	13	13	0.01	1.10	0.32	0.36	
S-6	Cu	13	13	1	28	11	8	
	Co	13	13	17	39	29	7	
	Cr	13	13	1	5	3	1	
	Ni	13	13	14	57	33	17	
	Pb	13	13	5	19	12	6	
	Cd	13	13	1	5	2	1	
	Mn	13	13	0.71	2.32	1.60	0.45	
	Fe	13	13	6.30	75.09	38.80	24.17	
	Zn	13	13	0.01	0.63	0.15	0.18	

Table 7 (cont'd.)

Sample	Element	s	d	Min. Conc.	Max. Conc.	Mean Conc.	Standard Deviation	Median
S-9	Cu	13	13	2	12	6	3	
	Co	13	11	ND	6	--	--	3
	Cr	13	7	ND	3	--	--	1
	Ni	13	13	3	62	16	17	
	Pb	13	13	3	30	15	9	
	Cd	13	13	1	3	2	1	
	Mn	13	13	1.30	7.04	5.15	1.63	
	Fe	13	13	13.95	77.60	45.17	17.43	
	Zn	13	13	0.02	0.15	0.06	0.03	
S-10	Cu	13	13	1	13	8	4	
	Co	13	12	ND	9	--	--	3
	Cr	13	11	ND	3	--	--	1
	Ni	13	13	5	57	19	16	
	Pb	13	13	2	45	19	17	
	Cd	13	13	1	3	2	1	
	Mn	13	13	1.32	2.34	1.79	0.39	
	Fe	13	13	15.60	59.41	44.05	10.72	
	Zn	13	13	0.04	0.54	0.17	0.13	
S-11	Cu	13	13	1	19	8	6	
	Co	13	8	ND	3	--	--	2
	Cr	13	9	ND	2	--	--	1
	Ni	13	13	3	38	16	13	
	Pb	13	13	4	26	13	6	
	Cd	13	13	1	5	2	1	
	Mn	13	13	1.41	2.25	1.89	0.24	
	Fe	13	13	1.54	44.59	23.30	12.79	
	Zn	13	13	0.02	0.15	0.07	0.04	
C-1	Cu	13	13	1	12	5	4	
	Co	13	10	ND	11	--	--	4
	Ni	13	13	3	20	10	4	
	Pb	13	11	ND	20	--	--	9
	Cd	13	13	1	2	1	0	
	Mn	13	13	0.11	4.72	2.02	1.53	
	Fe	13	13	1.12	8.42	3.59	2.37	
	Zn	13	13	0.01	0.20	0.05	0.05	
S-15	Cu	9	9	1	12	6	3	
	Co	9	5	ND	5	--	--	1
	Cr	9	5	ND	3	--	--	1
	Ni	9	9	6	31	12	8	
	Pb	9	9	4	22	15	6	
	Cd	9	9	1	1	1	0	
	Mn	9	9	0.09	2.23	0.69	0.89	
	Fe	9	9	0.06	2.11	0.62	0.79	
	Zn	9	9	0.01	0.06	0.04	0.02	

Figure 8. Frequency distribution histrograms for metals (Co, Cr, Ni, Pb) whose concentrations were above the detection limit fewer than four times from August 3, 1979 to August 12, 1980. These values incorporate all the upland landfill sample metal concentrations and floodplain sample C-1 for Cr, where means could not be calculated.


Frequency
40
30
20
10
0
39
2
3
1
1
2
3
4
Co conc. (ug/l)
Frequency
50
40
30
20
10
0
49
4
3
1
1
1
2
3
4
5
6
7
Cr conc. (ug/l)
Frequency
40
30
20
10
0
14
21
5
1
10
20
30
Ni conc. (ug/l)
Frequency
40
30
20
10
0
38
2
3
2
1
5
10
15
20
Pb conc. (ug/l)

11 ug/l). Limited quantities of copper is being leached at both landfills.

Cobalt:

The mean and median concentration range for Co in the upland landfill samples (below the detection limit to 3 ug/l) is not significantly higher than the range of values of the control (below the detection limit to 1 ug/l). Co is not being leached from the upland landfill.

The only wells that show appreciable leaching of Co at the floodplain landfill are S-5 and S-6. Mean values for these samples are 24 ug/l and 29 ug/l, respectively. These are much higher than the control median values of 4 ug/l (C-1) and 1 ug/l (S-15), and are much higher than for all other samples within or adjacent to this landfill. S-5 and S-6 are the shallowest wells within the test cell. This suggests that Co is mobilized and put into solution at shallow depths. These high Co values may reflect the chemical composition of the buried refuse at these specific locations. Also, because the refuse is inhomogeneous, it is possible that material with a high Co concentration has been buried near these two wells. While possible, this does not seem highly likely.

Chromium:

Cr does not appear to be leaching from either of the two landfill sites. The ranges of mean and median concentrations for the samples from the upland and floodplain landfills do not seem significantly from the control samples.

Nickel:

It appears that Ni is being leached from both landfill sites. The range of Ni concentrations for the upland landfill sample sites are higher than the range for the control. This indicates slight leaching of Ni at the upland landfill.

Ni contamination is also apparent within the floodplain landfill with mean concentration values ranging from 16 ug/l to 33 ug/l. Contamination is evident in well S-2, adjacent to the landfill, with a mean concentration of 20 ug/l. Ground water from wells S-1 (median of 11 ug/l) and S-13 (mean of 11 ug/l) show no contamination when compared to the mean concentrations of 10 ug/l (C-1) and 12 ug/l (S-15) in the control samples.

Lead:

Lead is not being leached at the upland landfill. Pb was detected only seven times in samples from this location.

Concentration for Pb in water samples from control wells at the floodplain landfill are median of 9 ug/l (C-1) and mean of 15 ug/l (S-15). In general, well samples taken within and adjacent to the floodplain landfill are somewhat higher in Pb concentration, indicating slight contamination from Pb leaching.

Cadmium:

Cd concentrations for both landfill sites proved to be the most invariable through time and among sampling sites. There is no indication of Cd leaching or Cd contamination in water samples from either landfill.

Manganese:

Manganese appears to be leaching from the area of sample sites 1 to 3 of the upland landfill. Mean values ranged from 0.10 mg/l to 0.30 mg/l, compared to 0.04 mg/l from the control site 4.

Mean Mn concentrations within the floodplain landfill (1.20 mg/l to 5.15 mg/l) and even adjacent to this landfill (0.08 mg/l to 1.03 mg/l) are higher than from the control well S-15 (0.69 mg/l). Ground water from well C-1 had a high mean value for Mn of 2.02 mg/l, possibly related to the depth of the well, which is 12.04 m (38.5'). This well was drilled within the deep channel of the Kansas River floodplain. It is likely that sufficient CO_2 is available to form a weak carconic acid solution. This would partially dissolve minerals and oxide coatings in Mn-Fe-rich alluvium, thereby yielding relatively high Mn concentrations to the ground water.

Iron:

As expected, Fe behaved much like Mn. Some Fe leaching is occurring at the upland landfill. Mean concentrations for water from all four sampling sites (0.15 mg/l to 0.21 mg/l) are higher than the mean value for the control site (0.12 mg/l).

The only evidence for Fe leaching at the floodplain landfill existed from the ground water sampled from wells within the region of buried refuse. Mean concentration values here ranged from 17.60 mg/l to 45.17 mg/l compared with the values of 3.59 mg/l (C-1) and 0.62 mg/l (S-15) at the control wells. C-1 had higher values in Fe for the same reasons it had high Mn values. No evidence exists for Fe leaching or contamination adjacent to the floodplain landfill.

Zinc:

There is evidence for minimal leaching of Zn at sample sites 1 (0.14 mg/l) and 2 (0.09 mg/l), compared to samples from the control site (0.06 mg/l) at the upland landfill.

At the floodplain landfill, control wells C-1 and S-15 yielded mean concentrations of 0.05 mg/l and 0.04 mg/l, respectively. These values are significantly lower than those from within the landfill (0.06 mg/l to 0.89 mg/l) and adjacent to the landfill (0.13 mg/l to 0.27 mg/l). This indicates that Zn is being leached from the floodplain landfill, and some Zn contamination exists in ground water adjacent to it.

DISCUSSION

Significance of Data:

Data collected from sampling sites at the upland and floodplain landfills during the period August 3, 1979 to August 12, 1980 demonstrate that it is possible to monitor leachate composition, migration and contamination potential. This can be accomplished by relatively simple analyses for such characteristics as pH, specific conductance, and trace metal composition. Table 8 compares metal concentrations with drinking water standards published by U.S.E.P.A. (1976). This table illustrates the contrast between the ground water quality at both landfills and drinking water standards, based on trace metal data.

The upland landfill has not been used for over 10 years. No acid contamination in surface-shallow ground water flowing from this site could be detected. Conductivity values remained low, as did trace metal concentrations, when compared with data from Yankee Tank Lake and data from the floodplain landfill. This suggests that a landfill, in an upland geohydrologic environment, may not adversely affect surface-shallow ground water quality in and around the landfill site after being closed for ten years. The fact remains that the pollution potential of a landfill is greatest in the first years of use, so only after this time is it safe to suggest that aging helps to reduce the contamination potential of a sanitary landfill site.

It is noteworthy that a direct comparison between the

Table 8. Mean trace metal concentrations compared with drinking water standards, in ug/l. "+" indicates no published value. "--" indicates mean not calculated. "*" indicates median value.

Sample	Cu	Co	Cr	Ni	Pb	Cd	Mn	Fe	Zn
Std.	1,000	+	50	+	50	10	50	300	5,000
1	8*	--	--	--	--	1*	300	150	140
2	6	--	--	--	--	2	250	210	90
3	8	--	--	--	--	2	100	160	30
5	7	--	--	--	--	3	40	180	50
4	6*	--	--	--	--	2	40	120	60
S-1	10	1*	1*	11*	10*	2	80	320	270
S-2	11	3*	1*	20	20	2	760	250	130
S-13	9	1*	1*	12	13	2	1,030	2,760	140
S-4	10	6	1*	29	15	2	2,000	17,600	890
S-5	9	24	2*	33	14	2	1,200	22,460	320
S-6	11	29	3	33	12	2	1,600	38,800	150
S-9	6	3*	1*	16	15	2	5,150	45,170	60
S-10	8	3*	1*	19	19	2	1,790	44,050	170
S-11	8	2*	1*	16	13	2	1,890	23,300	70
C-1	5	4*	--	10	9*	1	2,020	3,590	50
S-15	6	1*	1*	12	15	1	690	620	40

upland and floodplain landfills on the basis of aging cannot be made. The two have extremely different hydrologic, geologic, and soil characteristics. The only true measure of aging can be obtained from continuous, regular monitoring of a specific landfill over a long period of time.

Based on an assessment of pollution potential for inorganic ions in solution, the most significant finding of this investigation is that the floodplain sanitary landfill does not appear to pose a threat to ground water quality in those areas surrounding the landfill. From previous work of Degner (1974), the ground water flow direction has been postulated to be from the north-northwest toward the south-southeast. Observation of ground water chemistries by Hathaway and Carr (1976) suggest that the ground water flow may be toward the west-southwest. Flow through alluvial soil is laminar. The spread of contamination is by molecular diffusion (McKinney and Blackman, 1973-1977). Moreover contaminated leachate is flowing only at shallow depths with the deeper ground water remaining uncontaminated. The deeper ground water acts only to dilute already contaminated ground water. As contaminants move in solution through the soil and alluvium, test wells penetrating the area should reflect the passage of contaminated water with depressed pH and increase in conductance and an increase in trace metal concentration.

Control well C-1 is northwest of the floodplain landfill, and control well S-15 is southeast. Both wells, during the one year studied, showed no evidence of contamination. Ground water from wells S-1, S-2, and S-13, adjacent to the landfill, on the eastern fringe, showed only a slight amount of metal, acid, and dissolved solids contamination. All wells within the landfill, particularly the shallow ones, yielded leachate showing significant contamination. This indicates that leachate moving in the subsurface has not reached the eastern fringes of the landfill and certainly not the subsurface area penetrated by the southeastern most well (S-15).

Degner (1974) found the mean horizontal velocity of ground water at the floodplain landfill to be 2.1 ft/day. He based his calculations on the ground water system's flow rate, saturated thickness, width of saturated zone, and porosity. Using five years (1825 days) as a time frame and Degner's ground water velocity, the leachate generated by refuse buried in the original test cell should be about 3800 ft. away. Well S-15 is less than 2000 ft. away, yet it has not yielded contaminated and potentially hazardous leachate. Thus, either leachate movement is much slower than Degner previously suggested, its movement has been slowed by absorption and adsorption onto clays, or it is moving in a more southerly or westerly direction than Degner believed. If the flow regime is more southerly or even westerly, the leachate will likely enter the Kansas River; however, tremendous dilution by the river will likely prevent an environmental hazard.

Degner's work (1974) is not entirely consistent with that of Hathaway and Carr (1976) of the Kansas Geological Survey. In this latter study, they sampled ground water from existing wells on the west side of the landfill. These wells have been destroyed by landfilling equipment and could not be used in this investigation. Early in 1975, they found NH_4^+ concentrations as high as 2.3 mg/l, Fe concentrations as high as 48 mg/l, and NO_3^- concentrations as low as 0.9 mg/l in ground water at the southwest corner of the landfill. These values indicate the presence of leachate west of the region of buried refuse.

After more than nine years of operation, there is no evidence of pollution of ground water in the area surrounding the floodplain sanitary landfill. It is evident that as leachate produced from landfilling operations passes into the surrounding soils, microbial degradation, absorption and adsorption onto clays, and dilution tend to reduce the impact of this leachate on the quality of the ground water and the adjacent Kansas River. There has, of course, been some contamination of the ground water in the immediate area of the landfill. The amount of contamination released, in an easterly direction, has been low with respect to the contaminant concentrations within the landfill. Continued testing at the well sites, particularly wells S-1, S-2, S-13, and S-15, will help to monitor ground water quality in the areas surrounding the landfill.

Chemical Interpretation:

There are many data that, when collectively examined, may aid in defining the chemistry of the leachate from both the upland and floodplain sanitary landfills. They are pH, specific conductance, dissolved oxygen, concentration of metals, and the concentration of the manganous ion (Mn^{+2}).

Upland Landfill:

The mean pH values at the upland landfill sampling sites ranged from 7.3 to 7.9 with no evidence for acid contamination. The mean conductance measurements ranged from 368 to 970 umhos at $25^{o}C$. A rough estimate of total dissolved solids (T.D.S.), in mg/l, can be obtained from specific conductance data, in umhos, by multiplying by 0.7 (Davis and De Wiest, 1966). This yields T.D.S. values of approximately 260 to 680 mg/l, which are not excessive. Many natural waters contain T.D.S. concentrations within this range. Drinking waters standards state that T.D.S. should not exceed 250 mg/l (U.S.E.P.A., 1976).

Dissolved oxygen concentrations (measured from June through August, 1980) at the sampling sites averaged 6.4 mg/l, compared with 6.8 mg/l in Yankee Tank Lake, the control. Values in this range are also common in natural waters. Oxygen at those concentrations is readily available to aid in the rapid and complete breakdown of organic matter into soluble species and gases. At these levels, an aerobic environment is present.

Metal concentrations at the sampling sites were consistently low, indicating only small quantities of metals mobilized into solution. Furthermore, there appears to be insufficient organic matter and moisture content at the landfill to produce acid conditions, and those metals that leach from the site are not now in mobile forms. They are precipitated as oxides and hydroxides. Finally, the relative concentration of Mn^{+2} (the reduced form of Mn) was always below the detection limit of the instrument (1.0 mg/l). These data are summarized in Table 9.

In conclusion, these observations indicate that the upland sanitary landfill does not at this time (October, 1980) pose any environmental threat, based on surface-

Table 9. Mean Mn^{+2} concentrations and percent Mn^{+2} values based on eight sampling days. "ND" indicates below the detection limit of the E.P.R. instrument (1.0 mg/l).. "--" indicates no calculation.

Sample	Mn^{+2} (mg/l)	Total Mn (mg/l)	%Mn^{+2}
1	ND	0.30	-
2	ND	0.25	-
3	ND	0.10	-
5	ND	0.04	-
4	ND	0.04	-
S-1	ND	0.08	-
S-2	ND	0.76	-
S-4	1.80	2.00	90
S-5	1.15	1.20	96
S-6	1.40	1.60	88
S-9	2.79	5.15	54
S-10	ND	1.79	-
S-11	1.09	1.89	58
S-13	ND	1.03	-
C-1	ND	2.02	-
S-15	ND	0.69	-

shallow ground water data collected during this investigation.

A computer determined correlation matrix and cluster-analysis among trace metals studied showed correlation coefficients ranging from -0.15 to 0.64. These data are in Appendix C. For the elements examined, no meaningful correlations were detected.

Floodplain Landfill:

Of the two landfills investigated, leachate from the floodplain landfill appears to be chemically the most complex. Many characteristics of the leachate from this area indicate the presence of a anaerobic system within the landfill, a borderline aerobic-anaerobic zone on the eastern perimeter of the landfill, and an aerobic zone surrounding the landfill.

Oxygen deficiency and the availability and type of organic matter are probably the most important controls on biological activity in subsurface environments, even though the controls on these reactions are not well understood. Within this landfill, there appears to be sufficient moisture (especially during the late winter and early spring) and organic matter to favor bacterial activity, which quickly depletes the existing oxygen supply. This is shown by mean dissolved oxygen concentrations averaging 0.3 mg/l since June, 1980 within the landfill compared with mean concentrations of 0.9 mg/l at the control wells.

Depressed pH values also are characteristic of anaerobic conditions, owing to the decomposition of organic matter. Although this quality is not overly pronounced at this landfill (suggesting conditions might not be strongly anaerobic) on an annual basis, individual values for certain wells attained pH levels as low as 6.4.

Under anaerobic conditions, many trace metals, especially Fe and Mn are more mobile. This has been shown by the characteristically high metal concentrations within the landfill itself. When compared with data in the literature, the concentrations of metals in the leachate at the floodplain landfill are lower, again suggesting limited mobility from incomplete anaerobic activity.

Table 9 shows the mean concentration of Mn^{+2} and percent Mn^{+2} in solution in the ground water sampled from both landfills on the eight sampling days Mn^{+2} was measured. Most manganese reduction occurs within the original test cell, at the shallowest depths. The percent Mn^{+2} was calculated by dividing the Mn^{+2} concentration by the total Mn concentration and multiplying by 100. The significance of this data is questionable, because of the way in which the samples were handled after collection and before the E.P.R. analysis. Too much time elapsed before analysis, allowing Fe oxides and hydroxides to precipitate. Most likely this scavenged Mn. Therefore, some of the Mn that was in solution upon sample collection formed insoluble oxides and hydroxides along with Fe or was simply trapped with precipitating iron compounds. Mn^{+2} was detected in the analysis indicating that some Mn^{+2} still remained in solution, though the original concentration of Mn^{+2} directly after collection is unknown. Sample collection and storage in an inert gas environment could alleviate some of this problem. Determination of total Mn and Mn^{+2} from the same unacidified sample might have served as a better test for speciation differences for Mn. In leachate samples other than those collected within the landfill, Mn^{+2} was absent or below the detection limit of the instrument. No precipitation of "iron red" material (Fe-Mn-oxy-hydroxides) were observed in these samples.

Correlation data revealed a correlation coefficient of 0.65 between Fe and Mn, the only significant correlation. In a low pH, highly reducing environment, a correlation coefficient closer to 1.0 would have been expected. The coefficient value of 0.65 is significant because it suggests that the stability of Fe and Mn is similar but not identical in the system. Although much of the available Mn is in solution, Eh and pH of the leachate will not allow reduction of all the oxides and hydroxides of Fe. When and if the pH and oxidation-reduction conditions for Fe become low enough, it will all be converted to soluble Fe^{+2}.

SUMMARY OF CONCLUSIONS

1. On the basis of chemical components reviewed, there is no evidence that the upland sanitary landfill poses any threat as a contaminant source to the surrounding ground water system or to Yankee Tank Lake.

2. On the basis of chemical components reviewed, there is no evidence that the floodplain sanitary landfill poses any threat as a contaminant source to the surrounding ground water system or to the Kansas River.

3. Chemically, the effluent from the upland landfill indicates the presence of an aerobic system in the landfill itself.

4. The floodplain landfill is anaerobic within and borderline aerobic-anaerobic at its eastern edge.

5. In 10 years, measurable contamination has not reached well S-15, southeast of the floodplain landfill and approximately 300 m from the region of buried refuse. This implies that ground water movement in this area is 1) more southerly or westerly than previously documented or 2) extremely slow (less than 10 m per year). Also, possibly much of the leachate material is trapped by absorption and adsorption onto alluvial material in the landfill area.

RECOMMENDATIONS

1. It is not recommended that a new landfill operation be initiated in a rock quarry of any sort, especially in bedrock consisting of jointed limestone. In formations of this type, there is less time available for degradation to take place within the area of leachate generation. Ground water velocities are much higher, therefore only partially degraded and poorly diluted leachate may show up at some distances from the landfill. Lining of quarry walls and floor may help to alleviate this problem. Calcium carbonate treatment of leachate would help neutralize leachate and precipitate undesirable trace elements.

2. Wells C-1, S-1, S-2, S-13, and S-15 should be regularly sampled at the floodplain landfill to monitor any future leachate movement and to help detect potential environmental hazards.

3. A natural or synthetic seal should be devised as to suit any new sanitary landfills' geohydrologic conditions in order to prevent vertical and horizontal leakage.

4. Knowledge of local hydrologic and geologic conditions is essential during the selection of a landfill site in order to determine the potential for attenuation of the leachate, surface water percolation rates, and ground water velocity.

5. Ground water monitoring wells should be placed in and around any landfill site. They should be sampled regularly to test for leachate arrival and character.

6. A more detailed study as to the oxidation state variation of certain trace metals at a landfill is necessary to further determine the chemistry of a leachate.

BIBLIOGRAPHY

Anderson, J. R. and Dornbusch, J. M. (1967), Influence of Sanitary Landfill on Ground Water Quality. J. Am. Water Works Assoc. 59: 457-470.

Apgar, M. A. and Langmuir, D. (1971) Ground Water Pollution Potential of a Landfill Above the Water Table. Ground Water 9: 76-94.

Baedecker, M. J. and Back, W. (1979) Hydrogeological Processes and Chemical Reactions at a landfill. Ground Water 17: #5.

Boggess, D. H. (1975) Effects of Sanitary Landfill Operations on Water Quality Near Fort Myers, Florida, U.S. Dept. of Interior, Geological Survey, Miami, Florida.

Brown, S. M. (1970) The Establishment of a Sanitary Landfill For Lawrence, Kansas. Ph.D. diss., Kansas University.

Camp, T. R. (1963) Water and its Impurities. Reinhold Publishing Corp., New York, pp. 245-247.

Cartwright, K., Griffen, R. A., and Gilkeson, R. H. (1956) Migration of Landfill Leachate Through Unconsolidated Porous Media. Advances in Ground Water Hydrology, pp. 215-227.

Chain, E. S. K. (1976) Sanitary Landfill Leachates and Their Treatment. J. Environ. Eng. Div. 102: 411-431.

Davis, S. W. and Carlson, W. A. (1952) Geology and Ground Water Resources of the Kansas River Valley Between Lawrence and Topeka, Kansas. Kansas Geol. Survey Bull. 96: 201-276.

Davis, S. N. and DeWiest, R. J. M. (1966) Hydrogeology. John Wiley & Sons, Inc., New York-London-Sydney.

Degner, D. A. (1974) A Chemical Investigation of the Effect of a Floodplain Solid Waste Disposal Site on Ground Water Quality. Ph.D. diss., Kansas University.

Dobson, A. A. (1964) Microbial Decomposition Investigation in Sanitary Landfills. Ph.D. diss., West Virginia University.

Dufford, A. E. (1958) The Quaternary Geology and Ground Water Resources of Kansas River Valley Between Bonner Springs and Lawrence, Kansas. Kansas Geol. Survey Bull. 130: part 1.

Farquhar, G. J. *et al*. (1972) Sanitary Landfill Studies Final Report. Vol. 1, Waterloo Research Institute Project 8083.

Frye, J. C. and Leonard, A. B. (1952) Pleistocene Geology of Kansas. *Kansas Geol. Survey Bull*. 99: 230 pp.

Fuller, W. H. *et al*. (1976) Contribution of the Soil to the Migration of Certain Common and Trace Elements. *Soil Sci*. 122: 223-235.

Fungaroli, A. A. (1971) Pollution of Subsurface Water by Sanitary Landfills. Vol. 1, Pub. #SW-12rg, U.S.E.P.A. Solid Waste Management Office, Cincinnati, Ohio.

Gibbs, R. J. (1973) Mechanisms of Trace Metal Transport in Rivers. *Science*. 180: 71-73.

Griffen, R. A. et al. (1976 and 1977) Attenuation of Pollutants in Municipal Landfill Leachate by Clay Minerals: Part 1 and Part 2. *Environ. Geol. Notes.* Illinois Geological Survey, #78 and #79.

Hathaway, L. R. and Carr.(1976) Influence of the Lawrence Alluvial Sanitary Landfill Upon Local Groundwater Quality. Open File Report.

Hughes, G. M., Schleicher, J. A., and Cartwright, K. (1976) Supplement to the Final Report on the Hydrogeology of Solid Waste Disposal Sites in Northeastern Illinois. *Environ. Geol. Notes*. Illinois Geological Survey, #80.

Korte, N. E. *et al*. (1976) Trace Element Movement in Soils: Influence of Soil Physical and Chemical Properties. *Soil Sci*. 122: 350-359.

Korte, N. E., Nieble, E. E., and Fuller, W. H. (1976) The Use of CO_2 to Sample and Preserve Natural Leachates. *J. Water Poll. Control Fed*. 48: 959-961.

Langmuir, D. (1972) Controls on the Amounts of Pollutants on Subsurface Waters. *Earth and Min. Sci*. 42: 9-13.

Lindorff, D. E. and Cartwright, K. (1977) Ground Water Contamination: Problems and Remedial Actions. *Environ. Geol. Notes*. Illinois Geological Survey, #81.

Lohman, S. W. (1941) Ground Water Conditions in the Vicinity of Lawrence, Kansas. *Kansas Geol. Survey Bull*. 38: 17-64.

Maienthal, E. J. and Becker, D. A. (1976) A Survey on Current Literature on Sampling, Sample Handling for Environmental Materials and Long Term Storage. *Interface*. vol. 5, #4.

McKinney, R. E. and Blackman, D. A. (1973-1977) Lawrence Sanitary Landfill Water Quality Testing Program. Submitted to the City of Lawrence, Kansas. Dept. of Civil Engineering, Lawrence, Kansas.

O'Connor, H. G. (1960) Geology and Ground Water Resources of Douglas County, Kansas. *Kansas Geol. Survey Bull.* 148: 200 pp.

Olsen, G. W. (1974) Using Soils of Kansas For Waste Disposal. *Kansas Geol. Survey Bull.* 208: 51 pp.

Perkin-Elmer (1973) *Analytical Methods for Atomic Absorption Spectrometry*. Perkin-Elmer, Norwalk, Connecticut.

Robson, S. G. (1977) Effects of Land Disposal of Sludge and Sanitary Landfills on Ground Water Quality. Dept. of Interior, Geological Survey, Denver, Colorado.

Rovers, F. A., Nuan, J. P., and Farquhar, C. J. (1979) Landfill Contaminant Flux, Surface and Subsurface Behavior. Proceedings Ontario Industrial Waste Conference, v. 21.

Rovers, F. A. and Farquhar, G. J. (1973) Infiltration and Landfill Behavior. *Environ. Eng. Div. J.*, 99: 2084-2100.

Salvato, J. (1971) Sanitary Landfill Leaching Prevention and Control. *Water Poll. Control Fed.* 43: 2084-2100.

Skougstad, M. W. *et al*. (1979) *Methods for Determination of Inorganic Substances in Water and Fluvial Sediments*. Techniques of Water Resources Investigations of the U.S.G.S.

U.S.E.P.A. (1976) *Quality Criteria for Water*. Washington, D. C.

U.S.E.P.A. (1979) *Methods for Chemical Analysis of Water and Wastes*. Environmental Monitoring and Support Laboratory, Cincinnati, Ohio.

Zanoni, A. E. (1972) Ground Water Pollution: Sanitary Landfills-A Critical Review. *Ground Water*. 10: 3-13.

APPENDIX A

Table 9. Date number corresponding to actual date for Tables 10 and 11.

Date #	Actual Date
1	August 3, 1979
2	September 14, 1979
3	October 19, 1979
4	November 9, 1979
5	January 18, 1980
6	February 22, 1980
7	March 21, 1980
8	April 11, 1980
9	May 12, 1980
10	June 2, 1980
11	June 24, 1980
12	July 21, 1980
13	August 12, 1980

Table 10. Upland landfill field measurement results. "*" indicates no sample collected. Dissolved oxygen determined only on last four sample dates.

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Dissolved Oxygen (mg/l)
1	1	31.8	390	8.9	
2		27.0	420	8.6	
3		10.5	360	8.3	
4		9.5	370	8.2	
5		8.0	350	7.6	
6		9.5	450	8.2	
7		10.5	410	7.7	
8		13.0	425	7.6	
9		17.0	360	7.5	
10		24.0	310	7.2	7.4
11		24.0	320	7.8	6.1
12		25.0	300	7.7	6.0
13		26.0	320	7.8	5.9
1	2	26.5	1140	7.5	
2		30.0	750	7.2	
3		9.5	780	7.6	
4		9.5	800	7.3	
5		8.0	650	7.5	
6		10.0	680	7.3	
7		11.0	750	7.4	

Table 10 (Cont'd).

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Dissolved Oxygen (mg/l)
8		11.5	415	7.3	
9		18.0	620	7.1	
10		21.5	1160	6.9	8.8
11		36.0	700	6.8	4.2
12		*	*	*	*
13		*	*	*	*
1	3	35.0	960	7.6	
2		28.0	920	7.6	
3		*	*	*	
4		*	*	*	
5		*	*	*	
6		*	*	*	
7		*	*	*	
8		*	*	*	
9		*	*	*	
10		22.0	1020	7.4	6.2
11		36.0	980	7.3	6.0
12		*	*	*	*
13		*	*	*	*
1	4	36.5	240	7.8	
2		30.0	250	7.7	
3		15.0	240	7.7	
4		12.0	220	7.8	
5		10.0	290	7.8	
6		10.0	280	7.5	
7		11.0	260	7.9	
8		14.0	280	7.6	
9		19.5	350	8.3	
10		25.0	300	7.6	7.0
11		31.0	255	7.6	7.0
12		31.5	250	7.5	6.5
13		36.0	220	7.4	6.6
1	5	41.0	600	8.0	
2		40.0	590	7.4	
3		*	*	*	
4		*	*	*	
5		*	*	*	
6		*	*	*	
7		*	*	*	
8		23.0	460	7.6	
9		*	*	*	
10		27.0	830	7.1	7.2
11		*	*	*	*
12		*	*	*	*
13		*	*	*	*

Table 11. Floodplain landfill field measurement results. "*" indicates no sample collected. Dissolved oxygen determined only on last four sample dates.

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Depth to Water	Dissolved Oxygen mg/l
1	C-1	17.0	1550	7.4	15'3"	
2		17.0	950	7.4	15'0"	
3		15.0	1700	6.8	16'2"	
4		14.5	1500	6.7	16'3"	
5		12.0	1200	7.1	15'0"	
6		11.0	1400	7.2	15'10"	
7		13.0	1550	6.9	15'4"	
8		11.0	1650	6.9	7'8"	
9		12.5	1850	6.8	13'4"	
10		10.0	2150	6.4	12'1"	0.2
11		14.0	1585	6.4	13'8"	0.4
12		15.0	2300	7.1	15'4"	0.4
13		16.0	2100	7.1	15'9"	0.5
1	S-1	16.0	1430	7.3	21'1"	
2		17.0	3000	7.4	23'5"	
3		15.0	1900	6.7	24'8"	
4		15.0	2100	6.7	25'2"	
5		11.5	4400	7.2	24'1"	
6		9.0	3400	7.4	24'10"	
7		14.0	4200	6.4	24'4"	
8		12.0	840	6.9	19'7"	
9		14.0	780	7.0	23'2"	
10		14.0	980	6.8	22'11"	1.0
11		12.0	1010	6.9	23'9"	0.4
12		15.0	990	7.1	25'7"	1.2
13		16.0	1000	7.0	25'9"	0.8
1	S-2	15.0	1480	7.4	12'10"	
2		17.0	3500	7.1	8'1"	
3		17.0	2000	7.3	15'3"	
4		16.0	2300	7.3	15'11"	
5		12.0	4600	7.6	14'8"	
6		14.0	3500	7.6	15'8"	
7		13.0	4350	7.1	15'2"	
8		12.0	1480	6.9	9'11"	
9		12.0	1420	6.8	12'1"	
10		11.0	1550	6.6	11'4"	0.4
11		12.0	1540	6.7	13'3"	0.5
12		13.0	1800	6.9	14'10"	0.5
13		15.0	1700	6.9	15'5"	0.6

Table 11 (cont'd).

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Depth to Water	Dissolved Oxygen mg/l
1	S-3	18.0	1950	8.8	11'2"	
2		*	*	*	*	
3		*	*	*	*	
4		*	*	*	*	
5		*	*	*	*	
6		*	*	*	*	
7		*	*	*	*	
8		*	*	*	*	
9		*	*	*	*	
10		11.5	2920	6.8	10'9"	0.3
11		*	*	*	*	*
12		*	*	*	*	*
13		*	*	*	*	*
1	S-4	17.5	1550	6.9	9'8"	
2		17.0	3000	6.7	11'11"	
3		18.5	1900	6.6	12'5"	
4		18.0	1700	6.7	12'0"	
5		13.0	4400	7.4	11'3"	
6		13.0	3250	7.2	13'4"	
7		13.0	4300	6.4	13'0"	
8		9.5	1910	6.7	6'4"	
9		12.0	1575	7.2	9'8"	
10		13.0	1850	6.8	9'10"	0.4
11		*	*	*	*	*
12		*	*	*	*	*
13		*	*	*	*	*
1	S-5	17.5	4800	6.9	6'0"	
2		19.0	1500	7.4	6'11"	
3		19.0	1500	6.9	8'4"	
4		18.0	1700	6.7	8'1"	
5		10.5	3700	7.6	8'4"	
6		13.0	2500	7.4	9'9"	
7		12.0	3950	7.0	9'3"	
8		10.0	3880	6.8	3'9"	
9		12.5	3600	7.0	6'1"	
10		12.5	4050	6.9	6'1"	0.2
11		16.0	3830	6.8	6'4"	0.3
12		18.0	3900	6.8	7'11"	0.2
13		18.0	4350	6.9	8'4"	0.2
1	S-6	19.0	5500	7.3	5'9"	
2		18.0	1200	6.8	6'6"	
3		19.5	1300	6.8	6'11"	
4		19.5	1000	6.8	6'9"	
5		10.0	4000	7.1	7'1"	
6		12.0	2950	7.0	8'9"	
7		11.0	4020	6.7	8'2"	

Table 11 (cont'd).

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Depth to Water	Dissolved Oxygen mg/l
8		9.5	3900	6.7	3'10"	
9		12.5	3700	6.8	5'10"	
10		13.0	4200	6.8	5'3"	0.3
11		16.0	3800	6.7	6'9"	0.1
12		19.0	4900	6.9	7'1"	0.2
13		19.0	5250	6.8	7'7"	0.1
1	S-9	16.0	2200	7.1	11'10"	
2		15.0	1800	7.0	10'4"	
3		18.5	1700	6.8	14'8"	
4		18.0	1600	6.8	14'0"	
5		12.0	4000	7.2	13'8"	
6		12.0	2950	7.0	13'7"	
7		13.0	4050	6.8	13'1"	
8		10.5	2610	6.8	8'7"	
9		13.0	2450	7.1	11'11"	
10		12.0	3020	6.8	10'11"	0.4
11		13.5	3150	6.9	12'3"	0.4
12		17.0	2600	7.1	13'11"	0.3
13		17.5	2330	7.0	14'1"	0.3
1	S-10	15.0	1370	6.8	9'11"	
2		17.0	4000	7.1	10'6"	
3		18.0	2000	7.1	13'6"	
4		18.0	2000	7.1	13'8"	
5		12.0	4400	7.3	13'7"	
6		13.5	3400	7.3	12'1"	
7		12.5	1530	6.9	11'7"	
8		10.0	1910	6.9	7'5"	
9		12.0	1600	7.1	10'11"	
10		13.0	1850	6.8	9'10"	0.4
11		13.0	2030	6.8	11'8"	0.2
12		16.0	2000	7.0	13'2"	0.3
13		16.0	1780	7.1	13'7"	0.2
1	S-11	17.0	1160	7.3	8'10"	
2		18.0	2200	7.1	13'10"	
3		20.0	1800	7.1	15'4"	
4		18.0	1700	7.3	15'7"	
5		12.5	4200	7.3	14'4"	
6		13.0	3300	7.3	17'3"	
7		13.0	4200	7.2	16'9"	
8		10.0	1880	6.9	8'3"	
9		13.0	1850	7.2	12'4"	
10		14.0	2240	6.8	11'9"	0.2
11		14.0	2360	6.6	16'0"	0.5
12		15.0	1490	7.2	14'10"	0.3
13		16.5	1650	7.1	15'3"	0.3

Table 11 (cont'd).

Date #	Sample	T(°C)	Specific Cond. (umhos)	pH	Depth to Water	Dissolved Oxygen mg/l
1	S-13	15.0	840	8.2	8'9"	
2		14.0	4000	7.4	10'10"	
3		17.5	2000	8.9	13'2"	
4		17.5	1600	8.5	13'4"	
5		10.0	4600	7.4	11'11"	
6		13.0	3250	7.2	14'1"	
7		13.0	4200	7.4	13'7"	
8		8.5	1840	7.1	6'3"	
9		11.0	1000	7.4	9'8"	
10		10.0	1360	7.1	8'8"	0.5
11		11.0	1180	6.9	10'8"	0.1
12		14.0	1300	7.4	11'10"	0.8
13		15.0	1450	7.5	12'3"	0.6
1	S-15	*	*	*	*	
2		*	*	*	*	
3		*	*	*	*	
4		*	*	*	*	
5		10.5	4200	7.2	22'7"	
6		12.0	3500	6.7	23'5"	
7		13.0	2120	7.2	21'11"	
8		12.0	1150	7.3	17'4"	
9		12.5	920	7.0	20'9"	
10		12.5	1040	7.1	20'1"	2.1
11		12.5	1150	7.0	21'6"	0.8
12		13.0	1200	7.2	22'11"	0.8
13		14.5	2250	7.1	23'3"	0.9
1	S-12	19.0	1450	7.9	12'1"	

**Broken off and out of use

APPENDIX B

Table 12. Upland landfill trace metal raw data. "*" indicates no sample collected. "--" indicates below the detection limit. See Appendix A, Table 9 for date number correspondence with actual date.

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
1	1	10	--	--	5	--	2	0.52	0.30	0.05
2		--	--	--	3	--	1	0.51	0.20	0.02
3		3	--	--	--	--	3	0.12	0.16	0.05
4		3	--	--	--	--	2	0.55	0.25	0.24
5		4	--	--	--	--	2	0.23	0.15	0.03
6		4	--	--	2	--	2	0.45	0.17	0.03
7		4	--	--	2	--	1	0.44	0.17	0.03
8		18	--	3	22	10	2	0.12	0.08	0.29
9		8	--	2	19	7	1	0.02	0.14	0.38
10		24	--	--	22	--	1	0.05	0.03	0.11
11		13	--	2	--	--	1	0.31	0.08	0.03
12		10	--	--	--	--	--	0.32	0.08	0.30
13		9	--	--	--	--	--	0.29	0.08	0.30
1	2	6	--	--	7	--	2	0.25	0.30	0.05
2		2	2	--	3	--	2	0.52	0.20	0.04
3		5	--	--	--	--	2	0.12	0.16	0.05
4		3	--	--	2	--	2	0.28	0.28	0.06
5		2	--	--	--	--	2	0.20	0.30	0.07
6		2	2	--	2	--	2	0.18	0.19	0.07
7		2	2	--	2	--	2	0.19	0.19	0.07
8		10	--	2	10	14	1	0.03	0.22	0.32
9		9	--	--	20	6	1	0.02	0.11	0.10
10		7	--	--	10	--	1	0.05	0.17	0.03
11		18	--	--	6	--	1	0.95	0.17	0.16
12		*	*	*	*	*	*	*	*	*
13		*	*	*	*	*	*	*	*	*

Table 12 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
1	3	12	--	--	7	--	2	0.25	0.30	0.05
2		4	--	--	3	--	2	0.06	0.20	0.02
3		*	*	*	*	*	*	*	*	*
4		*	*	*	*	*	*	*	*	*
5		*	*	*	*	*	*	*	*	*
6		*	*	*	*	*	*	*	*	*
7		*	*	*	*	*	*	*	*	*
8		*	*	*	*	*	*	*	*	*
9		*	*	*	*	*	*	*	*	*
10		9	--	--	22	--	1	0.05	0.06	0.01
11		6	--	--	--	--	1	0.05	0.06	0.03
1	4	9	--	--	--	--	2	0.05	0.20	0.01
2		--	--	--	3	--	2	0.07	0.20	0.02
3		1	--	--	2	--	2	0.06	0.11	0.03
4		1	--	--	2	--	2	0.04	0.11	0.03
5		7	--	1	3	--	2	0.04	0.10	0.02
6		6	1	--	3	--	2	0.03	0.09	0.03
7		6	1	--	3	--	2	0.04	0.09	0.03
8		8	--	1	10	15	1	0.03	0.25	0.06
9		9	--	--	25	10	1	0.03	0.20	0.05
10		5	--	--	4	--	1	0.04	0.06	0.01
11		6	--	--	--	--	1	0.05	0.06	0.17
12		6	--	--	--	--	1	0.04	0.06	0.15
13		5	--	--	--	--	1	0.04	0.06	0.15
1	5	5	--	--	3	--	4	0.05	0.20	0.02
2		5	3	--	3	--	4	0.05	0.20	0.02
3		*	*	*	*	*	*	*	*	*
4		*	*	*	*	*	*	*	*	*

Table 12 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
5	5	*	*	*	*	*	*	*	*	*
6		*	*	*	*	*	*	*	*	*
7		*	*	*	*	*	*	*	*	*
8		11	--	1	12	15	1	0.02	0.22	0.15
9		*	*	*	*	*	*	*	*	*
10		5	--	--	--	--	1	0.05	0.11	0.01
11		*	*	*	*	*	*	*	*	*
12		*	*	*	*	*	*	*	*	*
13		*	*	*	*	*	*	*	*	*

Table 13. Floodplain landfill trace metal raw data. "*" indicates no sample collected. "--" indicates below the detection limit. See Appendix A, Table 9 for date number correspondence with actual date.

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
1	C-1	2	4	6	7	--	2	0.44	3.30	0.05
2		1	4	--	3	--	1	0.11	1.15	0.06
3		6	11	--	20	10	1	2.04	4.64	0.05
4		7	5	--	12	2	1	2.05	5.50	0.05
5		2	5	1	6	5	2	0.86	2.27	0.06
6		4	1	--	14	6	1	0.75	3.99	0.01
7		1	--	--	12	6	1	4.72	3.62	0.04
8		9	--	--	12	12	1	1.58	3.07	0.20
9		1	--	--	7	20	1	3.52	1.12	0.01
10		9	4	--	7	13	1	0.11	1.31	0.03
11		12	5	--	8	11	1	3.61	1.00	0.03
12		5	2	--	11	10	1	3.28	8.42	0.03
13		4	2	--	9	9	1	3.19	7.22	0.03
1	S-1	8	4	6	23	6	2	0.19	0.50	0.05
2		4	6	--	3	--	1	0.11	0.50	0.05
3		7	--	1	2	9	3	0.07	0.14	0.25
4		7	6	--	11	7	2	0.08	0.15	0.20
5		7	--	1	7	8	3	0.07	0.21	0.10
6		10	--	1	16	6	1	0.06	0.56	0.08
7		6	8	--	29	10	1	0.02	0.27	0.06
8		6	--	2	47	25	2	0.17	0.28	1.65
9		5	--	4	40	29	2	0.08	0.47	0.24
10		42	--	4	30	37	1	0.04	0.85	0.75
11		16	--	1	10	21	1	0.05	0.06	0.06
12		8	1	--	--	22	1	0.06	0.10	0.03
13		8	1	--	--	22	1	0.06	0.10	0.03

Table 13 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
1	S-2	8	5	2	11	3	2	0.11	0.20	0.04
2		7	7	--	3	4	2	0.44	0.05	0.04
3		10	2	4	8	9	2	0.43	0.20	0.04
4		8	9	--	11	8	2	0.43	0.10	0.04
5		17	2	1	17	5	2	0.64	0.21	0.08
6		11	--	1	15	7	1	0.12	0.27	0.06
7		13	2	2	25	8	1	0.98	0.60	0.03
8		7	--	5	46	19	2	0.54	0.22	0.95
9		8	--	8	30	39	2	1.16	0.31	0.17
10		24	3	1	20	41	2	1.20	0.22	0.15
11		19	4	1	32	40	2	1.20	0.08	0.05
12		5	3	--	19	40	1	1.32	0.38	0.04
13		6	4	--	17	38	1	1.32	0.38	0.04
1	S-3	8	5	16	18	6	4	1.80	0.20	0.12
2		*	*	*	*	*	*	*	*	*
3		*	*	*	*	*	*	*	*	*
4		*	*	*	*	*	*	*	*	*
5		*	*	*	*	*	*	*	*	*
6		*	*	*	*	*	*	*	*	*
7		*	*	*	*	*	*	*	*	*
8		*	*	*	*	*	*	*	*	*
9		*	*	*	*	*	*	*	*	*
10		12	4	1	11	35	2	2.78	13.69	0.31
11		*	*	*	*	*	*	*	*	*
12		*	*	*	*	*	*	*	*	*
13		*	*	*	*	*	*	*	*	*
1	S-4	24	8	2	41	14	4	2.55	46.00	1.10
2		8	8	1	4	8	3	2.01	26.00	1.76
3		9	9	2	32	7	3	1.44	16.95	1.12

Table 13 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
4	S-4	8	8	2	5	10	3	1.57	17.59	1.01
5		11	5	1	15	7	2	1.68	13.46	0.36
6		11	6	--	13	10	1	1.51	19.66	0.32
7		5	5	--	52	15	2	1.03	15.70	0.81
8		11	4	2	65	16	1	2.67	1.22	1.07
9		9	5	--	21	29	1	2.60	11.20	0.53
10		8	5	--	41	34	1	2.96	8.22	0.86
11		*	*	*	*	*	*	*	*	*
12		*	*	*	*	*	*	*	*	*
13		*	*	*	*	*	*	*	*	*
1	S-5	12	8	2	41	14	4	2.55	46.00	1.10
2		7	39	3	16	8	3	1.10	18.00	0.01
3		5	17	5	14	6	3	0.71	6.30	0.25
4		8	13	2	33	2	3	1.00	9.00	0.20
5		9	36	6	18	2	3	0.97	15.66	0.07
6		9	16	3	19	7	1	0.68	12.86	0.08
7		3	23	--	34	12	2	1.08	7.28	0.25
8		9	16	2	47	12	1	1.54	40.76	0.63
9		9	34	1	14	26	1	1.41	18.55	0.25
10		7	22	1	15	29	1	1.42	35.58	0.15
11		19	30	2	35	23	1	1.22	34.27	0.10
12		13	27	--	88	25	1	0.94	24.68	0.08
13		11	25	--	53	22	2	0.93	23.08	0.99
1	S-6	6	38	5	54	5	3	1.31	23.00	0.02
2		7	39	3	16	8	3	1.10	18.00	0.01
3		5	17	5	14	6	5	0.71	6.30	0.03
4		10	38	1	39	12	3	1.58	21.55	0.01
5		16	30	5	18	5	3	1.68	22.43	0.06
6										

Table 13 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
7	S-6	1	27	2	32	7	2	2.32	35.19	0.26
8		8	17	1	57	11	1	2.08	64.46	0.63
9		13	25	2	14	19	1	1.97	75.09	0.34
10		2	27	3	17	19	1	2.05	73.88	0.15
11		17	33	3	43	17	1	1.68	64.10	0.14
12		28	29	4	48	18	2	1.36	56.00	0.13
13		25	28	2	54	17	2	1.26	22.99	0.15
1	S-9	3	1	2	8	6	1	1.30	21.00	0.07
2		2	3	--	3	6	3	4.63	45.00	0.07
3		6	5	--	12	9	3	4.43	13.95	0.06
4		8	2	2	3	12	3	4.21	52.00	0.07
5		7	2	3	7	3	2	4.38	56.36	0.04
6		8	--	3	9	8	1	3.41	27.50	0.02
7		2	5	--	44	14	1	5.90	31.04	0.06
8		12	3	1	62	20	1	6.75	77.60	0.15
9		4	6	1	13	27	1	6.90	48.51	0.10
10		6	--	--	13	30	1	7.04	57.47	0.04
11		9	2	1	16	21	1	6.39	55.74	0.03
12		5	4	--	13	21	2	5.83	47.31	0.03
13		5	4	--	8	20	1	5.76	53.68	0.04
1	S-10	6	3	2	9	6	2	1.35	40.00	0.08
2		13	4	3	15	7	3	1.43	48.00	0.22
3		6	5	2	8	7	2	1.43	15.60	0.08
4		8	5	3	12	2	2	1.32	44.90	0.06
5		6	3	2	6	5	1	1.46	43.14	0.10
6		11	1	1	13	8	1	1.44	46.25	0.10
7		1	8	--	19	10	1	1.99	40.01	0.19
8		10	5	1	57	17	1	2.13	49.62	0.54

Table 13 (cont'd)

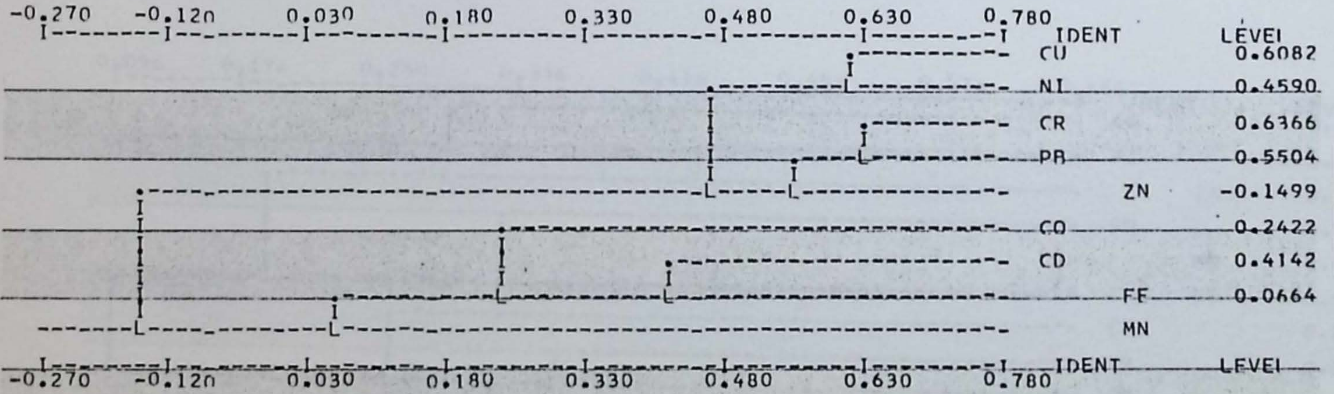
Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
9	S-10	5	9	1	15	42	1	2.32	39.65	0.19
10		6	3	--	5	45	1	2.34	42.42	0.18
11		8	--	1	14	41	1	2.07	44.59	0.04
12		13	3	1	50	41	2	2.00	59.41	0.21
13		13	3	1	20	21	2	2.03	59.06	0.22
1	S-11	19	2	2	9	6	2	1.75	18.00	0.05
2		3	2	--	3	17	2	1.85	22.00	0.05
3		2	2	2	5	5	5	1.76	12.30	0.04
4		2	2	2	4	4	3	1.90	27.50	0.04
5		6	2	2	6	10	2	2.01	34.43	0.06
6		9	--	2	6	11	1	1.51	14.57	0.02
7		1	3	--	28	8	1	1.41	1.54	0.04
8		11	--	2	38	18	1	2.04	27.22	0.15
9		3	--	1	31	26	1	2.25	30.79	0.07
10		8	3	1	11	15	1	2.21	3.38	0.03
11		8	--	1	14	19	1	2.07	44.59	0.04
12		18	2	--	36	17	2	1.89	33.10	0.13
13		18	--	--	14	16	2	1.90	33.50	0.14
1	S-12	13	6	6	8	5	1	3.10	16.00	0.01
	Buried by landfilling equipment and out of use.									
1	S-13	18	3	7	11	3	4	1.15	6.40	0.29
2		12	1	--	3	5	3	1.15	1.50	0.05
3		9	1	--	4	4	2	1.21	4.65	0.05
4		6	1	--	3	4	2	1.10	1.51	0.04
5		6	--	--	14	5	2	0.91	4.82	0.02
6		12	--	1	13	5	1	0.47	1.17	0.01

Table 13 (cont'd)

Date #	Sample	Cu (ug/l)	Co (ug/l)	Cr (ug/l)	Ni (ug/l)	Pb (ug/l)	Cd (ug/l)	Mn (mg/l)	Fe (mg/l)	Zn (mg/l)
7	S-13	4	3	--	18	10	1	1.00	0.23	0.09
8		16	--	1	15	21	1	1.14	0.39	0.71
9		16	--	1	35	18	1	0.99	0.25	0.43
10		6	--	--	10	22	1	1.06	0.55	0.04
11		7	--	--	10	25	1	0.95	3.09	0.01
12		4	--	--	14	22	2	1.11	5.28	0.03
13		4	--	--	11	20	1	1.10	6.01	0.04
1	S-15	*	*	*	*	*	*	*	*	*
2		*	*	*	*	*	*	*	*	*
3		*	*	*	*	*	*	*	*	*
4		*	*	*	*	*	*	*	*	*
5		9	5	3	6	13	1	2.23	2.11	0.04
6		8	1	2	10	7	1	2.07	1.90	0.06
7		1	5	--	11	4	1	1.09	0.30	0.06
8		4	--	1	31	14	1	0.09	0.31	0.05
9		4	--	1	18	19	1	0.15	0.17	0.05
10		5	1	--	7	16	1	0.04	0.14	0.02
11		12	3	1	8	22	1	0.09	0.06	0.01
12		6	--	--	11	19	1	0.22	0.29	0.02
13		6	--	--	7	17	1	0.22	0.28	0.03

APPENDIX C

Figure 9. Correlation coefficients and final cluster analysis for metal concentrations from samples at the upland landfill. A value of one-half the detection limit was assigned to those concentrations that fell below the detection limit for a specific element.



```
-0.270    -0.120     0.030     0.180     0.330     0.480     0.630     0.780
  I---------I---------I---------I---------I---------I---------I---------I   IDENT      LEVEL
                                                           .-----------  CU            0.6082
                                                           I
                                                  .--------L-----------  NI            0.4590
                                                  I
                                                  I         .----------  CR            0.6366
                                                  I         I
                                                  I    .----L----------  PB            0.5504
                                                  I    I
         .----------------------------------------L----L---------------        ZN     -0.1499
         I
         I
         I                         .-----------------------------------  CO            0.2422
         I                         I
         I                         I          .------------------------  CD            0.4142
         I                         I          I
         I             .-----------L----------L------------------------        FE      0.0664
         I             I
  -------L-------------L-----------------------------------------------        MN
  I---------I---------I---------I---------I---------I---------I---------I   IDENT      LEVEL
-0.270    -0.120     0.030     0.180     0.330     0.480     0.630     0.780
```

Figure 10. Correlation coefficients and final cluster analysis for metal concentrations from samples at the floodplain landfill. A value of one-half the detection limit was assigned to those concentrations that fell below the detection limit for a specific element.