HYDROLOGY AND GEOCHEMISTRY OF A SLAG-AFFECTED AQUIFER AND CHEMICAL CHARACTERISTICS OF SLAG-AFFECTED GROUND WATER, NORTHWESTERN INDIANA AND NORTHEASTERN ILLINOIS

U.S. GEOLOGICAL SURVEY
Water-Resources Investigations Report 97-4198

Prepared in cooperation with the
INDIANA DEPARTMENT OF ENVIRONMENTAL MANAGEMENT,
OFFICE OF SOLID AND HAZARDOUS WASTE

U.S Department of the Interior
U.S. Geological Survey

Hydrology and Geochemistry of a Slag-Affected Aquifer and Chemical Characteristics of Slag-Affected Ground Water, Northwestern Indiana and Northeastern Illinois

By E. RANDALL BAYLESS, THEODORE K. GREEMAN,
and COLIN C. HARVEY

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Thomas J. Casadevall, Acting Director

For additional information, write to:
District Chief
U.S. Geological Survey
Water Resources Division
5957 Lakeside Boulevard
Indianapolis, IN 46278-1996

Copies of this report can be purchased from:
U.S. Geological Survey
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CONTENTS

CONTENTS—Continued

FIGURES

1–3. Maps showing:

4–11. Figures showing:

TABLES

CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATIONS

Multiply	By	To obtain
inch (in.)	25.40	millimeter
inch (in.)	25,400	micrometer (µm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer
square mile (mi^2)	2.590	square kilometer
pound per cubic foot (lb/ft^3)	16.02	kilogram per cubic meter
ton	909.0	kilogram (kg)
ounce	28.3	gram (g)
acre	2.470	hectare
cubic yard (cu yard)	.764	cubic meter

Temperature in degrees Celsius (°C) can be converted to degrees Fahrenheit (°F) as follows:

$$°F=1.8x\ °C + 32$$

Sea Level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called "Sea Level Datum of 1929."

Abbreviations for water-quality units used in this report: Chemical concentrations and water temperature are given in metric units. Volumes of sample are given in liters (L) and milliliters (mL). Chemical concentrations are given in milligrams per liter (mg/L), milligrams per kilogram (mg/kg), or micrograms per liter (µg/L). Milligrams per liter is a unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water. One thousand micrograms per liter is equivalent to one milligram per liter. For concentrations less than 7,000 mg/L, the value expressed in milligrams per liter is the same as for concentrations in parts per million (ppm). Mass transfers are expressed in millimoles of mineral per kilogram of H_2O. One mole of a substance has mass equal to the atomic weight of that substance.

Specific conductance of water is expressed in microsiemens per centimeter at 25 degrees Celsius (µS/cm). This unit is equivalent to micromhos per centimeter at 25 degrees Celsius (µmho/cm), formerly used by the U.S. Geological Survey.

Redox potential of water is expressed in millivolts (mv).

Hydrology and Geochemistry of a Slag-Affected Aquifer and Chemical Characteristics of Slag-Affected Ground Water, Northwestern Indiana and Northeastern Illinois

By E. Randall Bayless[1], Theodore K. Greeman[1], *and* Colin C. Harvey[2]

Abstract

Slag is a by-product of steel manufacturing and a ubiquitous fill material in northwestern Indiana. Ground water associated with slag deposits generally is characterized by high pH and elevated concentrations of many inorganic water-quality constituents. The U.S. Geological Survey, in cooperation with the Indiana Department of Environmental Management, conducted a study in northwestern Indiana from June 1995 to September 1996 to improve understanding of the effects of slag deposits on the water quality of a glacial-outwash aquifer.

The Bairstow Landfill, a slag-fill deposit overlying the Calumet aquifer near Hammond, Indiana, was studied to represent conditions in slag-deposit settings that are common in northwestern Indiana. Ground water from 10 observation wells, located in four nests at the site, and surface water from the adjacent Lake George were analyzed for values of field-measured parameters and concentrations of major ions, nutrients, trace elements, and bulk properties. Solid-phase samples of slag and aquifer sediment collected during drilling were examined with X-ray diffraction and geochemical digestion and analysis.

Concentrations of calcium, potassium, sodium, and sulfate were highest in wells screened partly or fully in slag. Potassium concentrations in ground water ranged from 2.9 to 120 milligrams per liter (mg/L), were highest in water from slag deposits, and decreased with depth. The highest concentrations for aluminum, barium, molybdenum, nickel, and selenium were in water from the slag. Silica concentrations were highest in wells screened directly beneath the slag–aquifer interface, and magnesium concentrations were highest in intermediate and deep aquifer wells. Silica concentrations in shallow and intermediate aquifer wells ranged from 27 to 41 mg/L and were about 10 times greater than those in water from slag deposits. The highest concentrations for chromium, lead, and zinc were in ground water from immediately below the slag–aquifer interface.

The solid-phase analyses indicated that calcite, dolomite, and quartz generally were present throughout the slag–aquifer system; barian celestite, cristobalite, manganese-bearing calcite, and minrecordite were present in fewer samples. Trace elements that are liberated from the slag may be incorporated as impurities during precipitation of major minerals, sorbed onto clays and other grain-

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[1] U.S. Geological Survey.

[2] Indiana University, Bloomington, Indiana.

size fractions not analyzed as part of this study, or present in low-abundance minerals that were not detected by the X-ray analysis.

Mass-balance and speciation programs were used to identify geochemical processes that may be occurring as water infiltrates through the slag, flows into the aquifer, and discharges into Lake George. The geochemical models indicate that precipitation of calcite may be occurring where slag-affected water enters the aquifer. Models also indicate that dolomite precipitation and clay-mineral dissolution may be occurring at the slag–aquifer interface; however, dolomite precipitation is generally believed to require geologically long time periods. Silica may be dissolving where slag-affected ground water enters the aquifer and may be precipitating where slag-affected ground water discharges to the lakebed of Lake George.

In addition to the site-specific study, a statistical analysis of regional water quality was done to compare ground water in wells affected and unaffected by slag. When compared to wells in background locations in the Calumet aquifer, wells screened in slag across northwestern Indiana and northeastern Illinois generally had relatively higher pH and specific-conductance values and relatively higher concentrations of alkalinity, dissolved solids, suspended solids, total organic carbon, calcium, potassium, sodium, chloride, aluminum, barium, and possibly magnesium, sulfate, chromium, cobalt, copper, cyanide, manganese, mercury, nickel, and vanadium. When compared to wells in slag and wells in background locations, ground water from immediately beneath or immediately downgradient from slag had relatively high concentrations of arsenic and silica. Water-quality characteristics in ground water at the Bairstow Landfill were similar to water-quality characteristics in slag-contact and slag-affected wells throughout northwestern Indiana.

INTRODUCTION

Slag deposits have been used to reclaim former wetlands and to fill low-lying areas that cover more than 20 mi^2 of the Calumet aquifer in northwestern Indiana. Drainage from slag deposits has been suspected to have some undesirable chemical properties and to be associated with vegetative stress (Kay and others, 1996b; Doss, 1996; Roadcap and Kelly, 1993). A better understanding of the quality of slag-affected ground water would improve the ability of environmental managers to distinguish slag-related effects on water quality from effects of other sources. The U.S. Geological Survey, in cooperation with the Indiana Department of Environmental Management (IDEM), conducted a study from June 1995 to September 1996 to improve understanding of the effects of mixed steel-production slag on the water-quality and mineralogy of the Calumet aquifer (a glacial-outwash aquifer in northwestern Indiana and northeastern Illinois).

Background

Slag originates as a semi-liquid, non-metallic product that is separated from molten iron during the initial reduction of raw iron ore. Blast-furnace slags volumetrically comprise most of the slag produced. Blast-furnace slags generally are 95 percent silicates and aluminosilicates of calcium and magnesium; the remaining 5 percent are composed of iron, manganese, and sulphur compounds, and other iron-ore impurities like phosphorous (Lankford and others, 1985, p. 333–334).

Steel-making slag is another type of slag derived from the production of steel. Lime, used to remove acid oxides from steel, largely is incorporated into the steel-making slag. Steel-making slags typically are comprised of calcium silicates, calcium-iron compounds, and minor quantities of calcium oxide and magnesium oxide.

The physical properties of slag vary. Such properties as degree of crystallization, vesicularity, density, and particle size are determined largely by the method used to cool the molten material (Lankford and others, 1985). Slag is removed from the furnaces at about 1,480°C and may be cooled slowly in the open air, moderately fast with controlled sprays of water, or quickly by water immersion. Slag densities range from 50 to 156 lb/ft^3, depending on the vesicularity (Lankford and others, 1985).

Steel and slag production along the Chicago River began in the late 1830's. In the 1880's, the steel industry moved near the Calumet River in south Chicago. Slag was hauled to Indiana and used as lakeshore protection from erosion wherever the railroads followed the Lake Michigan beach. In 1903, a cement plant was built near Gary Station (later to become Gary, Ind.) to use the slag generated by the south Chicago steel industry for the manufacture of portland cement (Rooney and Carr, 1975, p. 25). In 1906, the steel industry expanded into Indiana, and slag production quickly outpaced the demands of cement manufacturers. During 1945 to 1950, the Gary, Ind., blast furnaces produced 1,000,000 tons of slag annually (Hurley, 1995, p. 20). Slag disposal in swampy, low-lying land at the southern end of Lake Michigan became commonplace to accommodate the oversupply.

In northern Lake County, Ind., more than 20 mi^2 of the Calumet aquifer is covered with slag and a smaller area is covered by "fill" (Kay and others, 1996b, pl. 2). Slag also is mixed with ash, construction debris, dredging spoils, industrial wastes, municipal wastes, sand, and sludges to create "fill." The thickness of most slag and fill deposits in northwestern Indiana is less than 20 ft but may be as much 70 ft adjacent to Lake Michigan (Kay and others, 1996b). As of 1979, about 10.3 mi^2 of manmade land had been added to the State of Indiana by filling along the Lake Michigan shoreline (Indiana Department of Natural Resources, 1979, p. 3).

Several studies that describe slag effects on ground- and surface-water quality have been done in parts of northwestern Indiana. Doss (1996) investigated the effects of drainage from a disposal area where tar decanter, coal-tar bottom sludges, and slag were disposed adjacent to the Miller Woods wetland. Examination of ground and surface water indicated that drainage from the landfill produced elevated values of pH and specific conductance and elevated concentrations of most water-quality constituents, including calcium, magnesium, potassium, sodium, chloride, and sulfate. Chemical analysis of a single sample of solid slag with X-ray fluorescence indicated the presence of calcium, magnesium, potassium, sulfur, phosphorous, aluminum, iron, manganese, silica, and titanium. The Doss slag sample was porous, indicating a large surface area available for chemical interaction.

Fenelon and Watson (1993) examined water quality in 35 wells from six land-use areas in the Calumet aquifer and overlying slag and fill deposits. Water from the monitoring wells screened in slag had the highest values of specific conductance (2,310–5,460 µS/cm) and pH (9.7–11.0) (Fenelon and Watson, 1993, p. 45). The median concentrations of most metals and arsenic in ground water from steel-making land-use areas, however, were less than the concentrations found in other areas such as industrial land-use areas.

Roadcap and Kelly (1994) sampled 21 observation wells surrounding Wolf Lake and Lake Calumet, Ill., approximately 1 and 4 mi, respectively, west of the site examined in this study. Potentiometric data from that study indicated that ground water flowed from the lakes, which were affected by slag filling, into the otherwise unaffected aquifer system. Water quality at background sites was pH 7.7 to 8.8; iron 377 to 1,270 mg/L; silica 3.77 to 8.34 mg/L; and total organic carbon, 6.4 to 12 mg/L (Roadcap and Kelly, 1994, p. C-7). Background water-quality data were used as input to the speciation model EQ3 (Wolery, 1979). Results indicated that background water was supersaturated with respect to calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$); supersaturated with respect to aluminosilicate minerals, goethite (a-$FeO{\cdot}OH$), hematite (Fe_2O_3), and magnesite ($MgCO_3$); undersaturated with respect to amorphous ferric hydroxide ($FeOH_3$) and iron-sulfide minerals; and nearly saturated with siderite.

Water from wells with pH values greater than 11.0 had elevated concentrations of aluminum and depleted concentrations of magnesium, iron, manganese, and silica relative to the background samples (Roadcap and Kelly, 1994). Calculations using EQ3 indicated that the high pH ground water was undersaturated with respect to feldspars, gibbsite ($Al(OH)_3$), and kaolinite ($Al_4(Si_4O_{10})(OH)_8$). Precipitation of brucite ($Mg(OH)_2$), high-magnesian calcite ($Ca,Mg(CO_3)$), iron and manganese carbonate, and oxyhydroxide minerals was thought to explain the relatively low concentrations of magnesium, iron, and manganese in ground water.

Calcium concentrations also were elevated in water from wells with pH values greater than 11.0, relative to background wells (Roadcap and Kelly, 1994). The minerals anorthite ($CaAl_2Si_2O_8$), dicalcium silicate (Ca_2SiO_4), melilite ($(Ca,Na)_2[(Mg,Fe^{+2},AlSi)_3O_7]$), merwinite ($Ca_3Mg(Si_2O_8)$), monticellite ($CaMg(SiO_4)$), rankinite ($Ca_3(SiO_7)$), spinel ($MgAl_2O_4$), and wollastonite ($Ca(SiO_3)$)—all thought to be constituents of slag (Lee, 1974; Loomba and Pandey, 1993)—were below saturation in all samples. Ground water was supersaturated with respect to barite ($BaSO_4$), chromite ($Fe^{+2}Cr_2O_4$), eskolaite (Cr_2O_3), ilmenite ($FeTiO_3$), rhodochrosite (MnO_3), rutile (TiO_2), strontianite ($SrCO_3$), and witherite ($BaCO_3$).

Purpose and Scope

The purpose of this report is to describe the results of a site investigation examining geochemical processes in a glacial aquifer that is receiving drainage from an overlying slag deposit. Ground-water-quality data from this and a previous study of northwestern Indiana and northeastern Illinois also were used in a statistical evaluation of regional data to evaluate whether slag-affected water can be distinguished from background water.

The Bairstow H. Company site (Bairstow Landfill), in Hammond, Ind., was instrumented with 10 observation wells and samples were collected for mineralogic, geochemical, and water-quality analyses. Geochemical models were constructed on the basis of data from the Bairstow Landfill to examine the potential processes responsible for water-quality changes where slag drainage enters a glacial aquifer. The generalized results of the site-specific investigation are expected to apply to numerous settings throughout the region where variable thicknesses of slag overlay glacial outwash and other sand aquifers.

A total of 137 ground-water-quality analyses from this study and a previous investigation of the Calumet aquifer in northwestern Indiana and northeastern Illinois were statistically summarized. The summary was intended to characterize the range and the variability in ground-water quality in northwestern Indiana and to examine distinguishing characteristics of water from wells screened in slag deposits (“slag-contact wells”), wells screened in an aquifer immediately beneath or downgradient from slag (“slag-affected wells”), and wells upgradient or at sufficient distance to be unaffected by slag drainage (“background wells”).

Acknowledgments

The USGS thanks the Robertsdale Foundation, Calumet College, and the Board of Commissioners of Lake County, Ind., for allowing the installation of observation wells at the Bairstow Landfill. The USGS also thanks the Standard Oil Company (Amoco) for coordinating and sharing fluid levels collected on Amoco property. Michal Umbanhowar-Zidek assisted with X-ray diffraction analyses at Indiana University, Bloomington, Ind.

DESCRIPTION OF THE STUDY AREA

The study area is in the Calumet Lacustrine Plain physiographic unit (Schneider, 1966, p. 50). The area was scoured smooth by glacial ice from the Lake Michigan lobe during the Pleistocene epoch. Reoccurring several times over the last million years, glacial ice followed a preferred route from Canada to the Chicago area, following the soft bedrocks that underlie the region. As the last glacier retreated, an ablation till known as the Wheeler Sequence (Brown and Thompson, 1995) was deposited onto older tills, leaving a combined thickness of about 55 ft of till overlying the bedrock. Meltwater from the glacier formed Lake Chicago (ancestral Lake Michigan) that stood at an elevation of about 640 ft (Fullerton, 1980, p1. 2). Sediments deposited from Lake Chicago accumulated to form the relatively thin Lake Border sequence. Southerly lake currents carried large volumes of fine glacial sand into this area. Much of the sand was reworked into numerous, distinct, low dune-beach ridges.

The Calumet aquifer is comprised of those wind- and water-transported fine sand and silt. The Calumet aquifer is a surficial deposit that is 18 to 20 ft thick throughout the study area. A thin sequence of fine-grained lacustrine sediment underlies the sand; parts of the lacustrine sediment may be included in the Lake Border sequence. Thickness of the fine sand aquifer and underlying lacustrine sediment averages 20 ft (Hartke and others, 1975, p. 25). A Wheeler sequence till underlies the lacustrine sediment.

The climate of northwestern Indiana is continental and is characterized by hot, humid summers and cold winters. The study area receives an average of about 37 in. of precipitation and 29 in. of snowfall annually (National Oceanic and Atmospheric Administration, 1992).

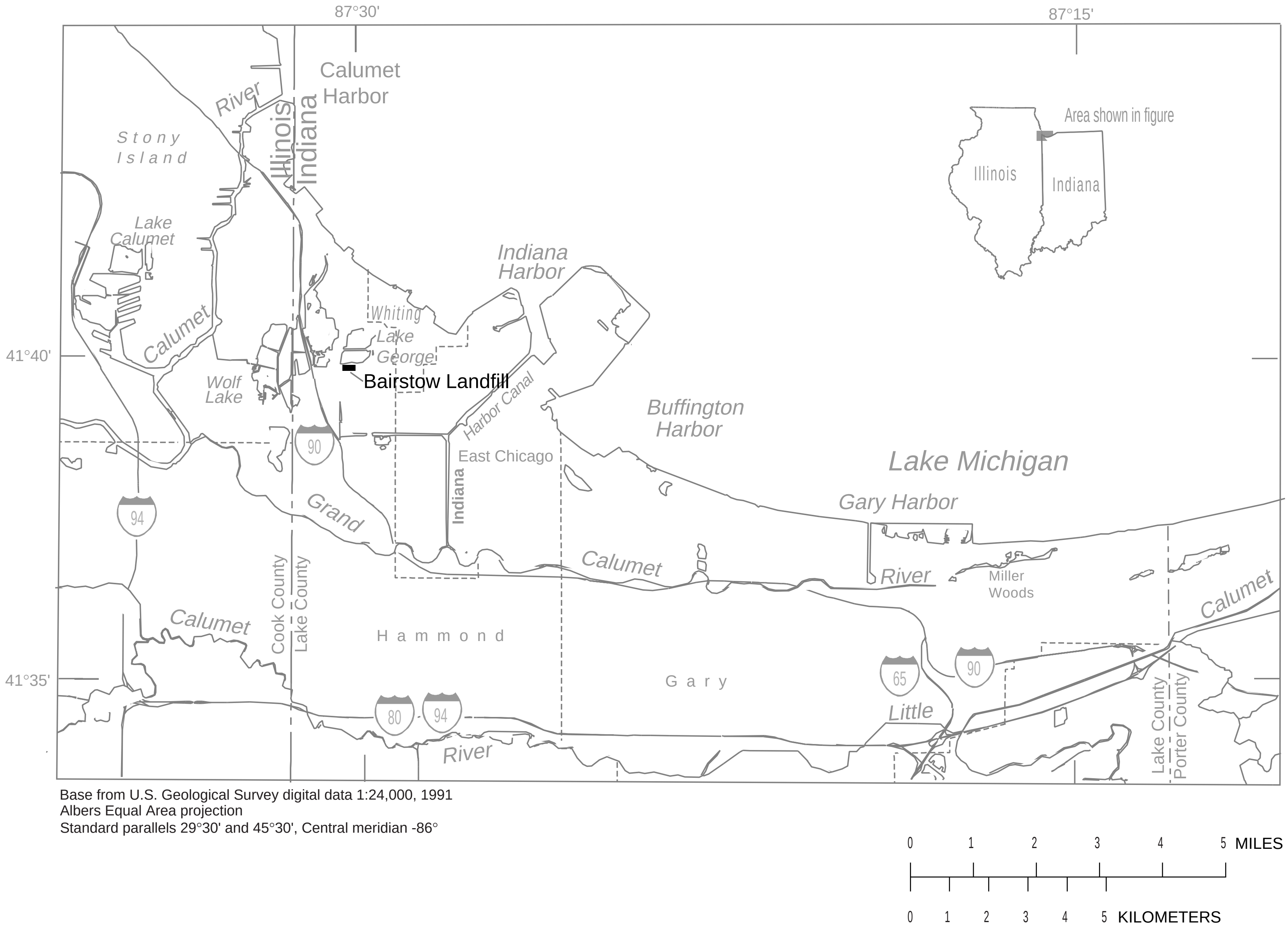
The Bairstow Landfill was selected for a site-specific investigation as part of this study. The Bairstow Landfill is about 1 mi south of Lake Michigan (fig. 1), on the northeast corner of 129th Street and Calumet Avenue in Hammond, Ind. (fig. 2). Lake George is immediately north of the Bairstow Landfill, and a wetlands (Lost Marsh) borders the landfill’s east side (fig. 2). Calumet College and a residential area are north and east of Lake George. Amoco maintains a petroleum-storage facility south of the Bairstow Landfill and 1,000 ft east of the Lost Marsh.

The wastes deposited at the Bairstow Landfill are described as various steel-production slags, fly ash and bottom ash from power plants, drums of oily sludge, and road-construction debris (Ecology and Environment, Inc., 1993). In 1980, a U.S. Environmental Protection Agency (USEPA) inspector noted about 100 drums of grease and oily materials lying on or near the Lake George shoreline. By 1987, the drums and 2.5 million cu yards of contaminated soil had been removed from the site (Ecology and Environment, Inc., 1993, p. 1). Metals concentrations in the laboratory leaching studies of 1993 soil samples were invalidated by quality-control data; PCB was detected in the Lost Marsh sediment. Air photos from 1951, 1965, and 1986 showed the progressive filling of Lake George that produced the Bairstow Landfill.

The surface area of the Bairstow Landfill is 100 acres. The landfill has a maximum relief of about 45 ft (fig. 2). The depth of slag at the site is 5 to 45 ft, including four slag piles of blast-furnace and open-hearth-furnace slag. Roy F. Weston, Inc., (1995) estimated that about 1,500,000 cu yards of slag exist on the site; about 4,000,000 cu yards were originally deposited, but 2,500,000 cu yards were removed later for construction projects.

METHODS OF INVESTIGATION

The investigation consisted of two efforts: (1) a site-specific study to collect data to generate and verify geochemical models for ground-water quality influenced by slag drainage and (2) a statistical analysis of regional data to evaluate whether slag-affected water quality can be distinguished from ambient or background water quality. The methods used for each study are described in this section of the report.



Base from U.S. Geological Survey digital data 1:24,000, 1991
Albers Equal Area projection
Standard parallels 29°30' and 45°30', Central meridian -86°

Figure 1. Location of study area and Bairstow Landfill in northwestern Indiana and northeastern Illinois.

The Bairstow Landfill was selected for site-specific study because (1) well drillers' records for an on-site observation well indicated that the stratigraphy was typical of northwestern Indiana; (2) analytical data for the on-site observation well indicated that water quality was being affected by slag drainage; (3) the hydraulic gradient in the landfill area was known from the USGS records for the northwestern Indiana data network; (4) surface material and material at depth on exposed slopes appeared to be common blast-furnace slag; and (5) active mineral deposition was occurring along the southern shoreline of Lake George, apparently where ground water was discharging from the landfill. The Bairstow Landfill was selected to represent slag–aquifer systems that are common throughout northwestern Indiana; however, the variability of slag-fill chemistry may make the results of this study less applicable to other locations in northwestern Indiana.

Well Installation

A south-to-north flowline was selected for instrumentation and sampling at Bairstow Landfill along the ground-water-flow direction interpreted from water levels of wells in the USGS network (see line A-A′ on fig. 3). The flowline began at the existing well BH-31, ended at Lake George, and was approximately perpendicular to the southern shoreline of Lake George.

The observation-well transect included 10 wells at four sites (figs. 3 and 4). Observation well BH-31 was installed during a previous investigation near the highest accessible elevation at the landfill. The 15-ft screened interval crosses the slag–aquifer interface. Nested observation wells were installed July 16 and 17, 1996, at sites BH-32 and BH-33 at about 510 and 800 ft north of BH-31, respectively. Wells at BH-32 are screened in slag (BH-32SL), in the aquifer just beneath the slag (BH-32S), in the middle of the aquifer (BH-32I), and at the base of the aquifer (BH-32D). Observation wells at BH-33 are installed in the slag (BH-33SL), across the slag–aquifer interface (BH-33S), and in the middle of the aquifer (BH-33I). The naming convention for these wells used "SL," "S," "I," and "D" to indicate "slag," "shallow," "intermediate," and "deep," respectively. The S, I, and D wells also are collectively referred to as "aquifer wells" throughout the report.

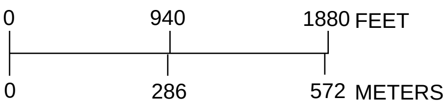
Wells at BH-32 and BH-33 were installed using the hollow-stem auger method. Wells at BH-32 and BH-33 were constructed from 2-in.-inner-diameter polyvinyl chloride (PVC) flush-joint casing and 5-ft-long PVC screens with 0.010-slot size. Wells were installed with a 6.75-in. hollow-stem auger. The annular space was backfilled with sand to 2 ft above the screen, bentonite to within 3 ft of land surface, and concrete to the surface.

Two observation wells were installed in the lakebed immediately north of the southern shore of Lake George on July 12, 1996. BH-34SS and BH-34ND are 1,200 ft north of BH-31 and about 5 and 20 ft north of the mean water line, respectively. BH-34SS is screened 2 to 3 ft below the lakebed and at least partially in slag, and BH-34ND is screened about 4.5 to 6.5 ft below the lakebed. Wells BH-34SS and BH-34ND were hand driven with a 36-kg post driver and constructed from 2-in.-diameter stainless-steel pipe with 1-ft stainless-steel screens having 0.010-sized slots. Lakebed wells are designated "ND" and "SS" to indicate "north-deep" and "south-shallow," respectively.

Water-Sample Collection and Analysis

Ground-water samples were collected August 13 through 15, 1996, from the 10 Bairstow Landfill observation wells. An additional sample was collected from Lake George, 2 ft below water surface, near well BH-34ND. Two additional samples were collected from well BH-32S to examine the reproducibility of sampling methodology and analytical precision.

Aerial photomap by U.S. Geological Survey
from aerial photograph taken September 25, 1990



EXPLANATION

POTENTIOMETRIC CONTOUR--Shows altitude at which water level would have stood in tightly cased wells, (August 13-15, 1996). Dashed where approximately located. Contour interval 1 foot. Datum is sea level

◎ Amoco ground-water observation wells

U.S. Geological Survey observation-well site and identification number

▲ Surface-water location

Figure 3. Ground-water and surface-water levels near Bairstow Landfill, Hammond, Indiana.

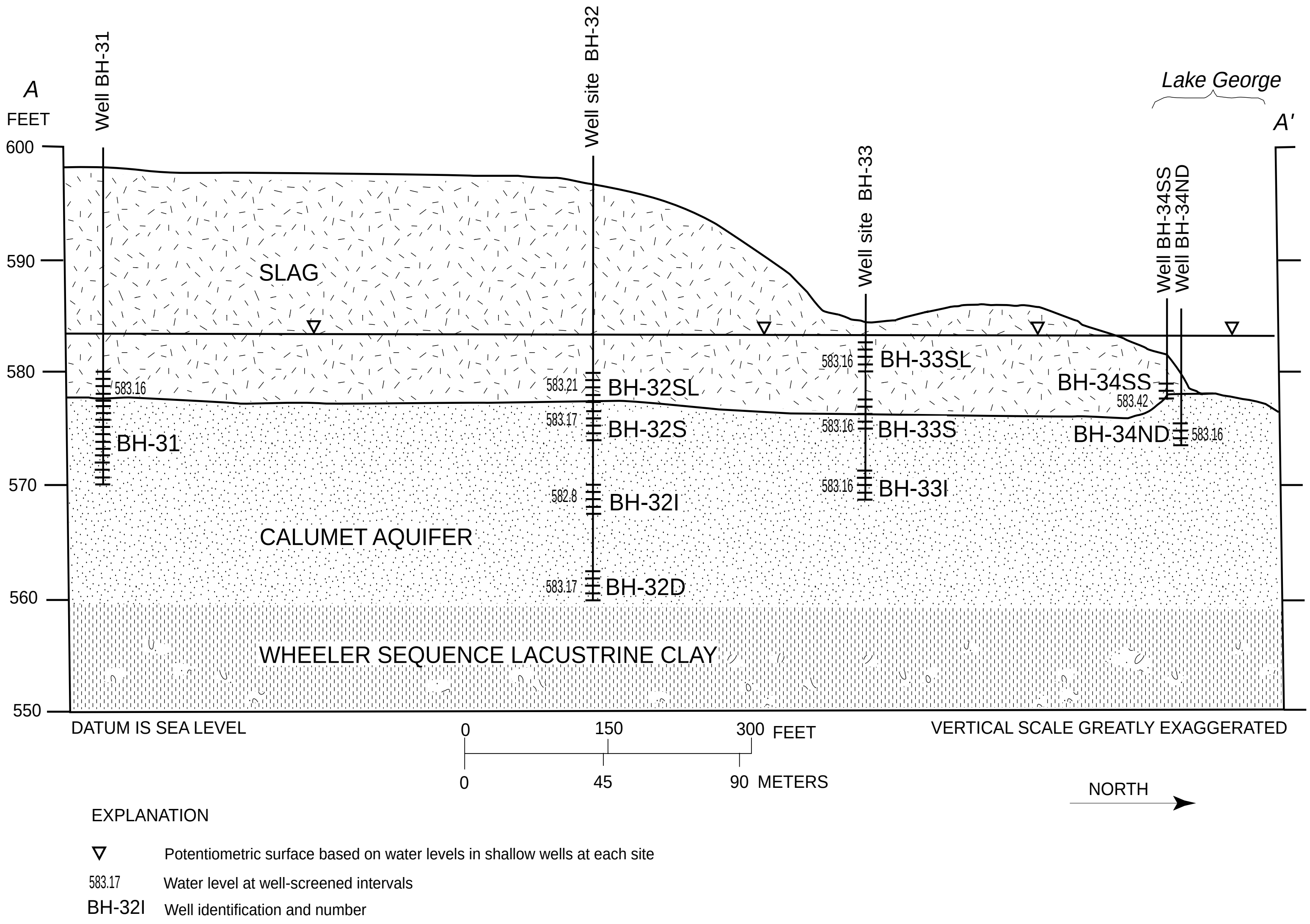


Figure 4. Hydrogeologic section *A-A'* through Bairstow Landfill, Hammond, Ind., showing geology, well positions, and water levels.

Samples for water-quality analysis were collected by use of a peristaltic pump equipped with Tygon tubing. Sampling preparation included removal of three times the standing volume of water in the well; on-site measurements of pH, specific conductance, redox potential, water temperature, and dissolved oxygen were monitored with a multiparameter probe and a stainless-steel flow-through chamber. On-site measurements were recorded and samples collected after on-site measurements had stabilized. Samples were preserved according to the guidelines specified by the USGS National Water-Quality Laboratory and are described in table 1. Measurements of redox potential are reported as a field measurement relative to the calomel electrode. All samples were analyzed by the USGS National Water-Quality Laboratory for concentrations of inorganic compounds similar to those examined by Duwelius and others (1996); the analytical methods are described in Wershaw and others (1987) and Fishman and Friedman (1989). An incremental titration to determine carbonate alkalinity was done at the site.

Decontamination of the pump, hoses, and flow-through chamber was done between each sampling. Decontamination consisted of pumping an Alconox-water mixture through the flow-through chamber, pump, and hoses, followed by tap water and deionized water. The pump and hose exteriors were rinsed between wells with deionized water.

Solid-Phase Sampling and Analysis

Solid-phase samples were collected during installation of all wells, except BH-31. Hand-driven split-spoon samples were collected in liners at BH-34 sites and immediately sealed with beeswax. Split-spoon sampling was not capable of capturing slag or sand at sites BH-32 and BH-33. Samples at sites BH-32 and BH-33 were taken from auger cuttings and from the auger bit as it was retracted from the screened interval. Auger cuttings and bit samples were preserved in zip-lock freezer bags and refrigerated until solid-phase analysis in August and September 1996.

The geochemistry and mineralogy of solid samples were determined at the Indiana University Department of Geological Sciences. Analyses included (1) a determination of solid chemistry from sample digestion and major and minor element quantification using neutron activation or ion chromatography and plasma spectroscopy, and (2) a description of bulk mineralogy based on X-ray diffraction.

Sixteen solid-phase samples were analyzed for geochemistry and mineralogy. The sample distribution included two slag samples, one sample from the slag–sand interface, three sand samples below the sand–slag interface, one sample of the Lake Border lacustrine deposit subdivided into three grain-size fractions (<2 µm, 2–5 µm, and 5–20 µm), and five samples from the lakebed of Lake George (fig. 5). In addition, a sample of sand unaffected by slag drainage was analyzed as a potential representation of native aquifer material. A replicate sample of lakebed sediment was analyzed for quality-assurance purposes.

The solid-phase analyses were intended to provide information to evaluate the extent of mineral precipitation from the interaction of slag drainage with ground water flowing through the site and not previously affected by slag. Mineral cements and overgrowths in the interstices and on the surfaces of aquifer sand grains were expected to be the most likely forms of secondary mineral precipitation.

Cements and overgrowths were removed from sand grains by grinding, followed by size separation. One-hundred-gram samples were mechanically ground for 15 minutes with a high-shear mixer and separated by sieving into >125 µm-, 125–45 µm-, and <45-µm-size fractions. Cements and overgrowths were observed during sample preparation to be best represented in the <45-µm fraction; these samples were examined by X-ray diffraction and geochemical analyses. One-hundred-gram samples typically contained 2 to 4 g of material with grain sizes less than 45 µm. Samples of lakebed material contained substantial quantities of clay-sized material; 100-g samples of lakebed material yielded 2 to 20 g of material with a grain size less than 45 µm.

Table 1. Sample-preservation requirements for ground-water samples collected at Bairstow Landfill, Hammond, Ind., August 1996

[ml, milliliter; µm, micrometer; °C, degrees centigrade; <, less than; HNO_3, nitric acid]

Analytes	Sample container, preparation, and preservation
Nitrite plus nitrate, as nitrogen; nitrite, as nitrogen; ammonia, as nitrogen; ammonia plus organic nitrogen, as nitrogen; phosphorous; and orthophosphate, as phosphorous	One 125-ml brown polyethylene bottle, field rinsed, filtered with a 0.45-µm filter, and maintained at 4°C
Potassium, iron, magnesium, manganese, calcium, silica, and sodium	One 250-ml polyethylene bottle, acid rinsed, filtered with a 0.45-µm filter, acidified with HNO_3 to pH<2 and maintained at 4°C
Residue on evaporation at 180°C and lab alkalinity	One 250-ml polyethylene bottle, field rinsed, unfiltered, nonacidified, and maintained at 4°C
Chloride, sulfate, and fluoride	One 500-ml polyethylene bottle, filtered with a 0.45-µm filter, nonacidified, and maintained at 4°C
Arsenic, aluminum, antimony, barium, beryllium, cadmium, chromium, cobalt, copper, lead, manganese, molybdenum, nickel, selenium, silver, uranium, zinc	One 250-ml polyethylene bottle, acid rinsed, filtered with a 0.45-µm filter, acidified with HNO_3 to pH<2, and maintained at 4°C
Lab pH and lab specific conductance	One 250-ml polyethylene bottle, field rinsed, nonacidified, nonfiltered, and maintained at 4°C

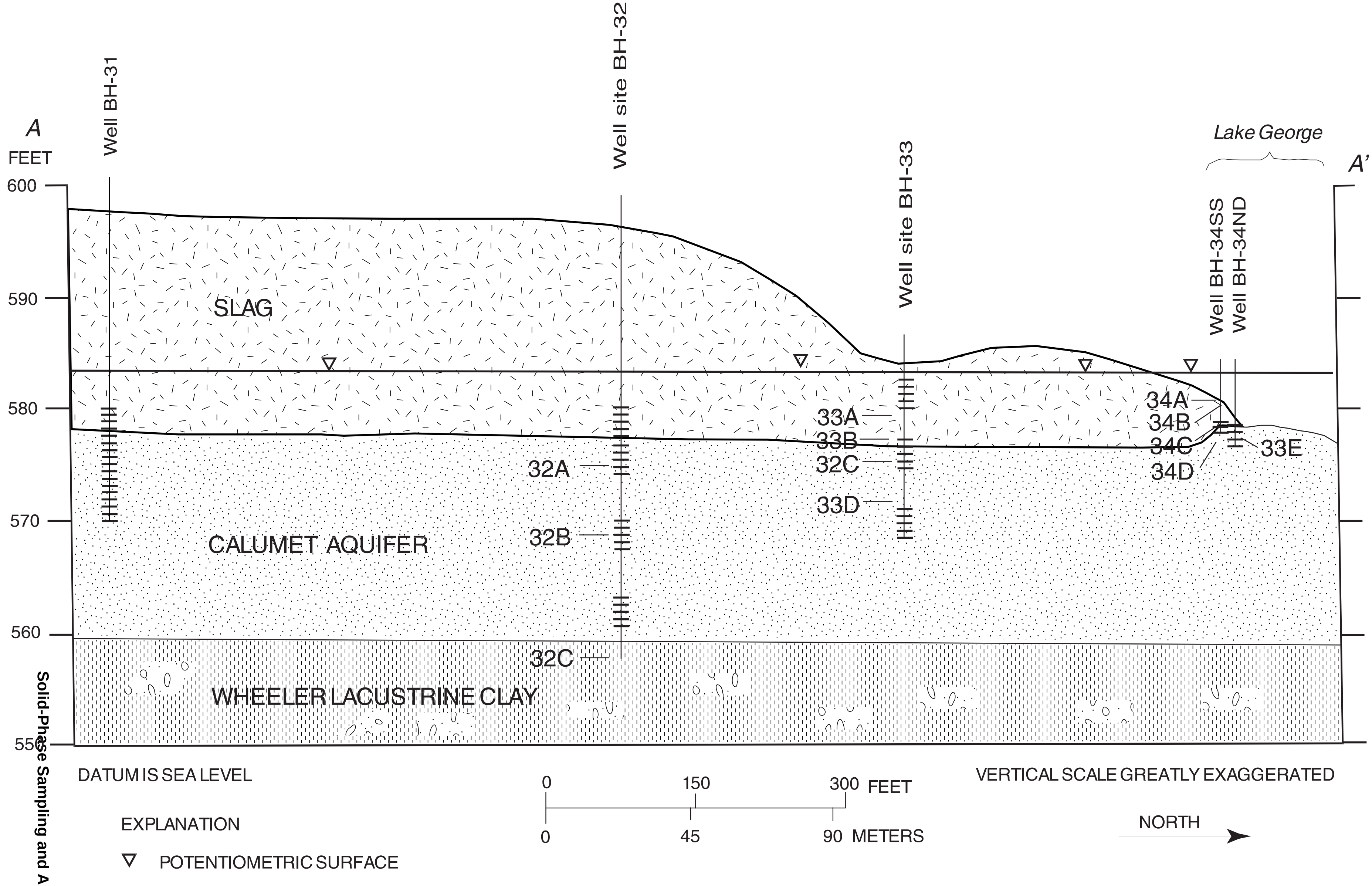


Figure 5. Solid-phase sample sites at Bairstow Landfill, Hammond, Ind.

The <45-µm samples from the Lake Border lacustrine sediment were further separated into <2-µm-, 2- to 5-µm- and 5- to 20-µm-size fractions. All samples were dried at 200°F for 24 hours to facilitate powder-slide preparation for X-ray diffraction.

X-ray diffraction was done at the Indiana University Department of Geological Sciences with a Philips XR 6 3100 X-ray generator with a copper-potassium (alpha) radiation source. Samples were scanned from 5 to 60° 2θ with a 0.05° step size.

Diffractograms, produced by X-ray diffraction analysis of the prepared samples, were compared with standard diffractograms of the International Centre for Diffraction Data, Newtown Square, Pa. Automated software facilitated mineral identification by (1) graphically depicting peak locations and intensities for standards and measured samples, (2) prioritizing most probable mineral occurrences based on the fit of mineral-standard diffractograms to the measured diffractogram, and (3) allowing graphical manipulation of the measured diffractograms (for example, the peaks of most probable mineral phases could be subtracted from the measured diffractogram to focus automated mineral identification on peaks of less-abundant phases). Factors considered when identifying the mineralogy believed represented in X-ray diffractograms included available information about (1) slag geochemistry; (2) mineralogy of the Calumet aquifer and Lake Border sequence lacustrine sediment; (3) general mineral occurrences; (4) precipitation kinetics, ion substitution, and the solid-phase geochemical analyses; (5) calculated saturation indices; and (6) the water-quality information generated by this study.

Solid-phase geochemical analyses were performed on 0.25-g samples. Samples were digested with 10 mL $HClO_4$-HNO_3-HCl-HF at 200°C to fuming and diluted to 10 mL with aqua regia. This method only partially leaches magnetite, chromite, barite, massive sulfides, and oxides of aluminum, zirconium, and manganese and is subject to volatilization losses of antimony, arsenic, chromium, and gold. Replicate analyses of sample 34E indicated that the methods used to prepare and analyze the solid-phase samples produced nearly identical measurements of elemental concentrations.

Mineral-Saturation Calculations

Two types of computer programs commonly are used to generate geochemical models that assist with interpretation of mineralogy and water-quality data: (1) speciation programs and (2) mass-balance programs. Speciation and mass-balance programs can be used to generate plausible precipitation–dissolution scenarios that are thermodynamically favored and to explain field and laboratory observations. Speciation and mass-balance programs were used in this study to determine possible explanations for the geochemical, mineralogical, and water-quality observations at Bairstow Landfill.

The computer program WATEQFP was used to generate geochemical models from water-quality data collected at the study site. WATEQFP is a speciation program based on WATEQ (Truesdell and Jones, 1974), WATEQF (Plummer and others, 1976), and WATEQ4F (Ball and Nordstrom, 1991). The WATEQFP computer program calculates ionic strength, elemental speciation, and mineral-saturation indices, among other things. Of particular interest to this study were the calculated mineral-saturation indices.

A mineral-saturation index is a theoretical calculation that indicates the tendency for a mineral to precipitate from solution or be dissolved from the aquifer matrix. If the saturation index for a particular mineral has a positive value, the mineral is supersaturated, and precipitation from solution is favored. If the saturation index is negative, the mineral is undersaturated and dissolution is thermodynamically favored. The calculated saturation indices do not, however, indicate whether a supersaturated mineral is present as a result of precipitation or if it was originally present in the solid phase. Similarly, the saturation indices do not indicate whether an undersaturated mineral that is not present in the solid phase has been removed by dissolution or whether it was never present.

Mass-Balance Calculations

The computer program NETPATH (Plummer and others, 1994) was used to generate mass-balance models along a ground-water flowpath at Bairstow Landfill. The mass-balance models were used to indicate the relative proportions of precipitating and dissolving minerals that could account for the observed differences in water quality between wells in the Bairstow Landfill.

NETPATH generates models based on minerals specified by the user to be important in the geochemical system. The specified minerals are added to or removed from the ground-water system in the computation to account for the differences in the concentrations of water-quality constituents between two wells. In addition to mineral precipitation and dissolution, NETPATH can calculate proportions of mixed ground-water components that can combine with the mineral precipitation and dissolution to account for changes in constituent concentrations. The number of models generated by NETPATH is based on the number of water-quality constituents, solid phases, and other processes selected by the user to potentially explain the changes in water quality.

Statistical-Summary Calculations

Data from this study (8 sites) and from Duwelius and others (1996; 129 wells sampled in June 1993) were used to characterize the range and variability of constituent concentrations in slag-affected ground water of northwestern Indiana and northeastern Illinois. Ground-water samples collected for both studies were analyzed for the same water-quality constituents and properties. The statistical summary was intended to identify water-quality constituents and properties that may be indicators to distinguish water from slag-contact wells, slag-affected wells, or background wells.

A geographical information system (GIS) was used to superimpose locations of the 137 ground-water-sample sites on digital maps describing (1) locations of known fill deposits (Kay and others, 1996b), (2) ground-water levels (Duwelius and others, 1996), (3) well depths, (4) stratigraphy, and (5) fill thicknesses (Kay and others, 1996a). Results of this study identified 3 subsets of wells that may represent common settings in northwestern Indiana, including 21 slag-contact wells, 25 slag-affected wells, and 11 background wells. Eighty observation wells were removed from the data set because water quality at those sites could be influenced by multiple factors, such as proximity to a domestic-waste landfill, in addition to slag drainage.

The 57 classified sites and their associated water-quality data were then summarized according to the following statistical properties: the total number of measurements in the data set; the median (middle) data value; the minimum and maximum values from the data set; and the first and third quartile (Q1 and Q3). The first quartile and third quartile values are the data at which 25 percent and 75 percent of the data in the distribution are less than or equal to those values in a hierarchial ordering of the data.

HYDROLOGY AND GEOCHEMISTRY OF A SLAG-AFFECTED AQUIFER

The Bairstow Landfill overlays the Calumet aquifer (Roy F. Weston, Inc., 1995). Slag thickness ranges from 0 to 7 m, being thinnest at Lake George and thickening toward the center of the landfill. The Calumet aquifer is fine- to coarse-grained sand with some gravel and is approximately 18 ft thick at the site. The thicknesses of the Lake Border sequence lacustrine sediment and Wheeler sequence till at the site are unknown. The Wheeler sequence is regionally underlain by bedrock shale and limestone.

Hydrology

A potentiometric-surface map (Greeman, 1995) was constructed from quarterly measurements of water levels in a regional network of 76 monitoring wells. That map indicates regional ground-water flow in the Calumet aquifer near the site is generally toward Lake Michigan (Greeman, 1995); ground-water flow near the Bairstow Landfill is usually toward Lake George (fig. 3).

The ground-water flowpaths, based on water levels in the Bairstow wells, indicate that recharge moves down through the slag pile and uppermost sands into the intermediate-depth sands. Ground water also moves up from the deeper sands to converge at the intermediate-depth sands, where ground water gains a strong horizontal component of flow toward Lake George. Ecology and Environment, Inc., (1993) noted an unidentified white leachate material entering Lake George from the northwestern corner of the Bairstow Landfill, suggesting ground-water discharges to Lake George.

Ground-water flow at the site is affected by Lake George, the wetlands bordering the east side of the site, and by induced drawdown on the property of the Amoco petroleum-tank farm. Drawdown of the water table is induced at the Amoco site to inhibit off-site migration of nonaqueous-phase liquids (fig. 3). The possibility exists for ground-water-flow direction at the study site occasionally to be reversed to north-to-south in response to local drawdowns produced by the Amoco barrier wells or elevated water levels in Lake George; as a result of the data collected by this study, however, conclusions cannot be made about occurrences of flow reversals.

Ground-Water and Surface-Water Chemistry

At the Bairstow Landfill, alkalinity, dissolved solids, pH, and specific conductance varied similarly with respect to depth or stratigraphic position and were distinctly highest in the water from the slag wells (BH-31, BH-32SL, BH-33SL, and BH-34SS) (fig. 6; table 2, back of report). Alkalinity values ranged from 626 to 903 mg/L as $CaCO_3$ in the water from the slag wells and 420 to 590 mg/L as $CaCO_3$ in all other wells. Alkalinity was 160 mg/L as $CaCO_3$ in the sample from Lake George. Dissolved-solids concentrations in water from the slag wells ranged from 848 to 1,240 mg/L. The dissolved-solids concentration was 864 mg/L at BH-33S but was less in all other samples, ranging from 506 to 680 mg/L in water from the other wells; it was 433 mg/L in the sample from Lake George. Values of pH ranged from 11.9 to 12.3 in the water from the four slag wells and from 7.4 to 9.9 in all other wells. The pH of the sample from Lake George was 8.5. Values of specific conductance ranged from 844 to 4,640 µS/cm in ground-water; specific conductance was 733 µS/cm in the sample from Lake George. Specific conductance ranged from 3,180 to 4,640 µS/cm in the slag wells and from 844 to 1,300 µS/cm in all other wells. Values of alkalinity, dissolved solids, pH, and specific conductance were lowest in ground water at BH-32I and BH-32D and were consistently higher in aquifer wells at BH-33 than at BH-32.

Concentrations of dissolved oxygen in ground water ranged from 0.1 to 0.4 mg/L, except at BH-34ND where the concentration was 1.6 mg/L. The concentration of dissolved oxygen in the sample from Lake George was 6.5 mg/L. Concentrations of dissolved oxygen were highest in the shallowest well at each site, ranging from 0.2 to 0.4 mg/L.

Redox potentials in ground water ranged from –127 to +53 millivolts (mv). The redox potential in the sample from Lake George was +294 mv. Redox potentials ranged –26 to +53 mv in BH-32I, BH-32D, BH33I, and BH-34ND. Redox potentials ranged from –127 to –64 mv in all other wells.

Ground-water temperatures ranged from 14.5 to 23.5°C. Temperatures at BH-33 were higher than at BH-32. BH-34ND had a higher temperature than BH-34SS, suggesting that BH-34ND may be more strongly influenced by Lake George. The water temperature in the sample from Lake George was 28.5°C.

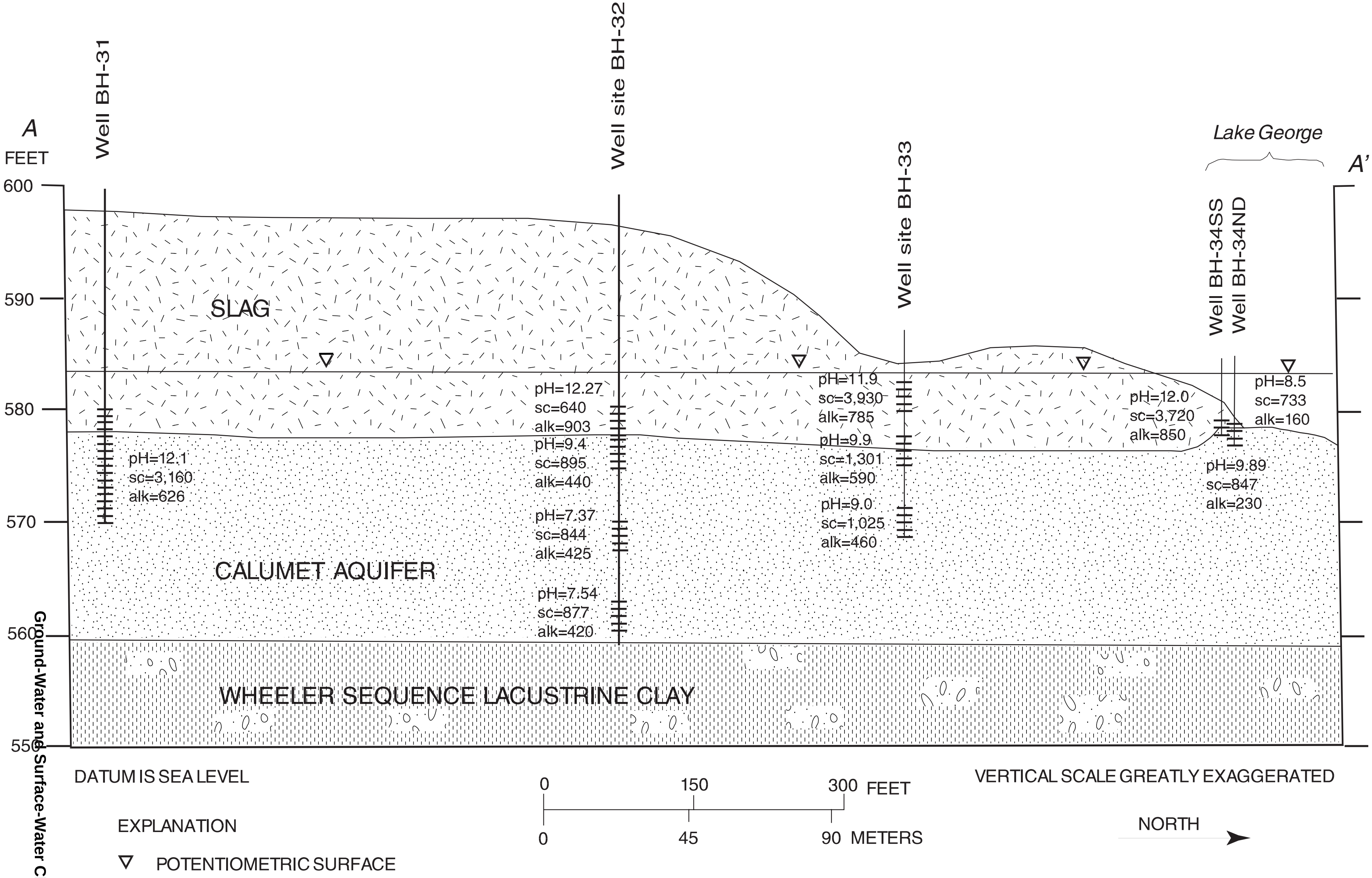


Figure 6. Distribution of pH, specific conductance (sc, in microsiemens per centimeter), and alkalinity (alk, in milligrams per liter as $CaCO_3$) in wells at Bairstow Landfill, Hammond, Ind.

As with the general water-quality properties, the distribution of the major ions and silica at Bairstow Landfill showed distinct patterns related chiefly to depth or stratigraphic position (fig. 7). Major ions (including calcium, magnesium, potassium, sodium, bicarbonate alkalinity, chloride, fluoride, sulfate, and ammonium) generally compose most of the dissolved mass in these water samples. Concentrations of calcium, potassium, and sulfate were highest in wells screened (at least partly) in slag; silica was highest in wells screened below the slag–aquifer interface; and magnesium was highest in intermediate and deep wells. The relative abundances of the carbonate species H_2CO_3–HCO_3^-–CO_3^{2-} are pH dependent, with H_2CO_3 most abundant to pH 6.4, HCO_3^- greatest in the interval pH 6.4 to 10.33, and CO_3^{2-} greatest above pH 10.33 at 25°C (Drever, 1982). Chloride concentrations showed no discernible patterns of distribution.

Concentrations of calcium ranged from 210 to 270 mg/L in water from the slag wells and 11 to 88 mg/L in all other wells (table 2, at back of report). The calcium concentration in the sample from Lake George was 23 mg/L. Between the slag and shallow aquifer wells at sites BH-32 and BH-33, the calcium concentrations decreased by 219 and 259 mg/L, respectively (fig. 7), indicating potential mass transfer lost from solution by mineral precipitation.

Concentrations of potassium in ground water ranged from 81 to 120 mg/L in water from the slag wells and 2.9 to 92 mg/L in water from all other wells (table 2). The potassium concentration in the sample from Lake George was 46 mg/L. At the nested-well sites (BH-32 and BH-33), the potassium concentrations were highest in water from slag wells and decreased with depth. As with many general water-quality parameters, concentrations of potassium in aquifer wells at BH-33 were higher than at BH-32.

Concentrations of sulfate ranged from 91 to 220 mg/L in water from the slag wells, 6.7 to 67 mg/L in water from the aquifer wells at BH-32 and BH-33, and were 110 mg/L in the sample from Lake George and the BH-34 wells.

Concentrations of silica ranged from 3.1 to 3.9 mg/L in water from the slag wells. Silica concentrations in water from shallow and intermediate aquifer wells were greater by about tenfold over the slag wells, ranging from 27 to 41 mg/L. The silica concentration in the sample from Lake George was 2.9 mg/L and was highest in BH-34ND at 74 mg/L (fig. 6).

Concentrations of sodium ranged from 34 to 220 mg/L. Concentrations less than 60 mg/L were limited to water from wells BH-31, BH-32I, and BH-32D.

Concentrations of magnesium were less than the method reporting limit at three of the four slag wells and 0.01 mg/L at BH-32SL. Magnesium concentrations in wells BH32I and BH32D were 27 and 41 mg/L, respectively, and ranged from 0.12 to 4.4 mg/L in other wells screened in the aquifer. The magnesium concentration was 7.2 mg/L in the sample from Lake George.

Trace elements generally compose a small fraction of the total dissolved mass in ground water. Trace-element concentrations measured in this study included aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, molybdenum, nickel, selenium, silver, uranium, and zinc.

Aluminum, barium, molybdenum, nickel, and selenium generally were present in greater concentrations in water from the four slag wells than from the slag–aquifer interface or in deeper parts of the aquifer. Aluminum concentrations in water ranged from 180 to 820 µg/L in the slag wells and ranged from 6 to 160 µg/L in all other wells. Barium concentrations in water ranged from 180 to 410 µg/L in the slag wells and from 24 to 200 µg/L in all other wells. Barium concentrations were elevated slightly in water from wells BH-32I and BH-32D, compared to shallow aquifer wells. Molybdenum concentrations ranged from 58 to 200 µg/L in water from the four slag wells and from 2 to 160 mg/L in all other wells. The molybdenum concentration in water from well BH-34ND (160 µg/L) may be related to elevated concentrations in the sample from Lake George (120 µg/L).

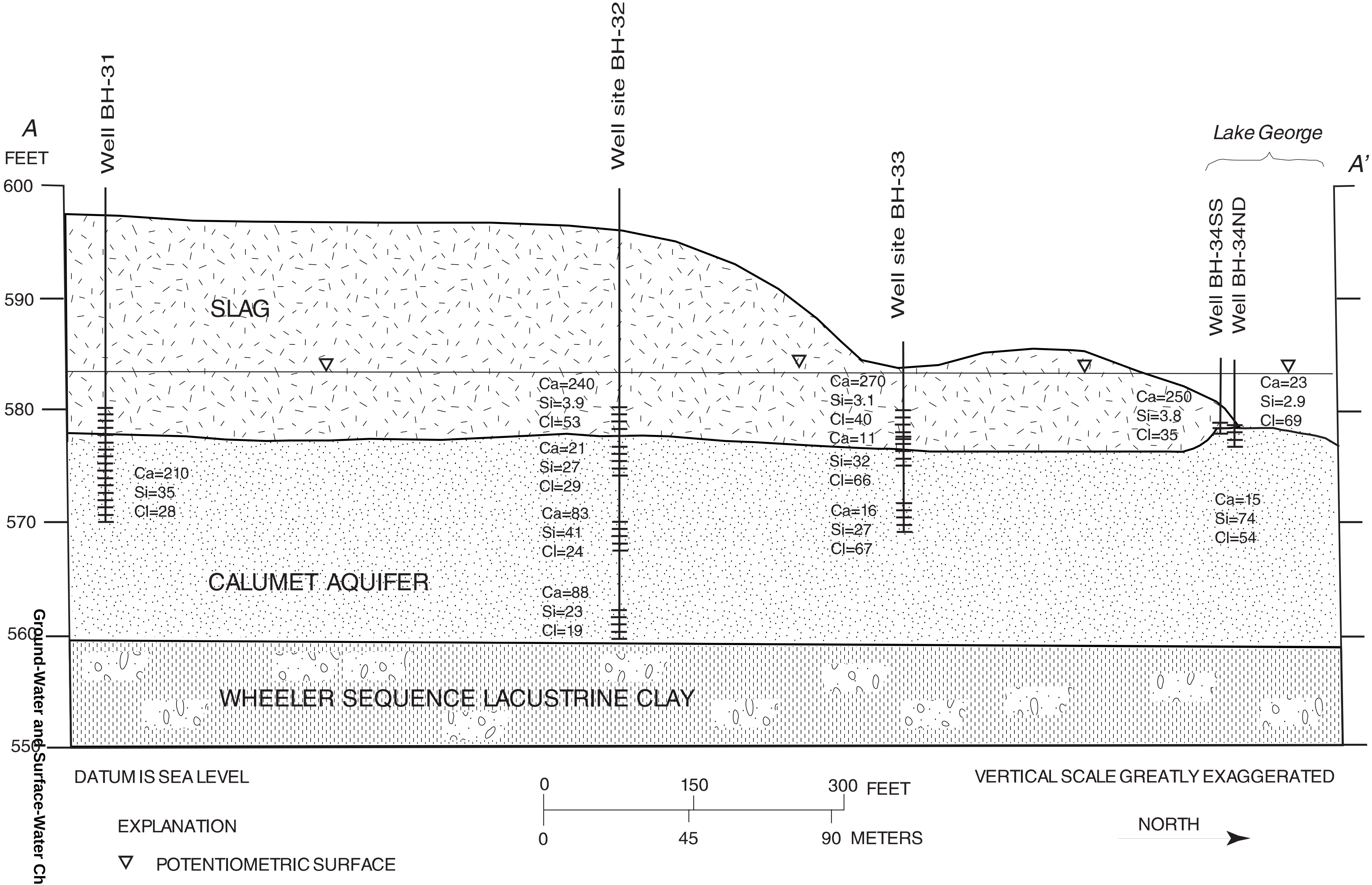


Figure 7. Distribution of calcium (Ca), silica (Si), and chloride (Cl) (in milligrams per liter) in wells at Bairstow Landfill, Hammond, Ind..

Nickel concentrations in water ranged from 9 to 13 µg/L in slag wells and from 2 to 6 µg/L in all other wells. Selenium concentrations in water ranged from 2 to 4 µg/L in slag wells and were below reporting limits (<1 µg/L) in all other wells.

The trace elements chromium, lead, and zinc generally were more abundant in water from wells positioned immediately below the slag–aquifer interface. Chromium concentrations were less than the laboratory reporting limits in water from the four slag wells, the sample from Lake George, and BH-32D. By comparison, chromium concentrations ranged from 2 to 4 µg/L in water from wells BH-32S, BH-32I, BH-33S, and BH33I. Lead concentrations were 6, 15, 12, and 2 µg/L in water from wells BH-32S, BH-33S, BH-33I, and in the sample from Lake George, respectively, but were less than the laboratory reporting limit in all other wells. The lead concentration in the sample from BH-33S, 15 µg/L, was equal to the USEPA maximum contaminant level for lead (U.S. Environmental Protection Agency, 1992). Zinc concentrations ranged from 2 to 3 µg/L in the four slag wells, BH-32I, BH-32D, and the sample from Lake George but were greater in water from wells BH-32S, BH-33S, and BH-33I, respectively.

Iron and manganese were most abundant in water from deeper aquifer wells, and concentrations generally increased with depth. Iron concentrations ranged from 12 to 290 µg/L in water from the four slag wells and ranged from 230 to 2,600 µg/L in aquifer wells at sites BH-32 and BH-33. Iron concentrations in the sample from Lake George and BH-34ND were 74 and 34 µg/L, respectively. Manganese concentrations were less than detection limits in the four slag wells but ranged from 23 to 130 µg/L at sites BH-32 and BH-33. Manganese concentrations in the sample from Lake George and BH-34ND were 5 and 2 µg/L, respectively.

The distribution patterns of arsenic and copper concentrations were difficult to decipher. The trace elements antimony, cadmium, cobalt, silver, and uranium did not exceed analytical reporting limits with enough frequency to permit determination of a distribution pattern.

The concentrations of several major ions and trace elements in water relate to pH values and bicarbonate concentrations and their effect on mineral-water interactions. For example, the water samples with the largest concentrations of calcium, potassium, sodium, sulfate, aluminum, barium, molybdenum, nickel, and selenium also have pH values ranging from 11.9 to 12.3 and are from slag wells. By comparison, water from wells with larger concentrations of silica and smaller concentrations of calcium also have smaller pH values. Water with the largest concentrations of chromium, lead, and zinc also generally have pH values that range from 9.0 to 9.9. Bicarbonate concentrations, as represented by alkalinity, are substantially greater in water from slag wells. Larger alkalinity values favor greater partial pressures of carbon dioxide and the dissolution of potassium and sodium aluminosilicate minerals (Stumm and Morgan, 1981, p. 545), possibly explaining the elevated potassium and sodium concentrations in slag-affected water.

The concentrations of nitrite as nitrogen and nitrate plus nitrite as nitrogen were highest in slag. Nitrite concentrations ranged from 0.40 to 2.90 mg/L in the four slag wells and from 0.01 to 0.10 mg/L in all other wells. The nitrite concentration in the sample from Lake George was 0.04 mg/L. Nitrate as nitrogen was present in such insignificant quantities that concentrations of nitrate plus nitrite as nitrogen were not significantly different from the concentrations of nitrite as nitrogen.

Concentrations of ammonia plus organic nitrogen, organic nitrogen, orthophosphate, and total phosphorous were highest in wells positioned immediately below the slag–aquifer interface. Ammonia concentrations ranged from 1.40 to 2.20 mg/L in water from the four slag wells, 2.60 to 13.0 mg/L in all other wells, and was 0.27 mg/L in the sample from Lake George. Ammonia concentrations in BH32S, BH-33S, and BH-33I ranged from 7.20 to 13.0 mg/L. Organic nitrogen ranged from 13.0 to 20.0 mg/L in BH32S, BH-33S, and BH-33I and ranged 2.60 to 3.90 mg/L in all other wells and 0.90 mg/L in the sample from Lake George. Orthophosphate concentrations ranged from 0.11 to 0.47 mg/L in the shallow and intermediate-depth aquifer wells at BH-32 and BH-33; concentrations ranged from <0.01 to 0.07 mg/L in all other samples. The highest orthophosphorous concentration (0.57 mg/L) was measured in well BH-32S. Total phosphorous ranged from 0.39 to 0.63 mg/L in wells BH-32S, BH-33S, and BH-33I and ranged <0.01 to 0.08 mg/L in all other wells.

Solid-Phase Geochemistry

Concentrations of cadmium, calcium, chromium, copper, iron, lead, manganese, niobium, phosphorus, tungsten, vanadium, and zinc concentrations generally were greater in slag samples than in samples of the Calumet aquifer sand and the Lake Border sequence lacustrine sediment. Slag samples were depleted in aluminum, potassium, scandium, and sodium relative to samples of the Calumet aquifer sand and the Lake Border sequence lacustrine sediment. By comparison, analyses of samples from 1 to 2 ft below the bases of three slag piles at Bairstow Landfill found concentrations of barium, calcium, chromium, copper, iron, manganese, nickel, selenium, and vanadium that were elevated relative to a background sample collected from a park about 1.2 mi north of the Bairstow Landfill (Roy F. Weston, Inc., 1995). In the 1995 Weston sample, collected from the slag pile nearest the study transect, the elements aluminum, beryllium, potassium, cobalt, silver, and arsenic also were detected.

The elements barium, copper, lead, nickel, sodium, strontium, titanium, vanadium, and zinc commonly occurred in concentrations ranging from 0.1 to 1.0 percent of the total mass of most samples. Concentrations in the range 0.1 to 1.0 percent were sometimes large enough to cause mineral precipitation and ion substitution that were apparent in X-ray diffractograms; minrecordite ($CaZn[CO_3]_2$) is an example. Two noticeably high lead, strontium, and zinc concentrations are associated with lakebed samples 34A and 34C (and possibly sample 34B). Copper concentrations were elevated in all lakebed sediments. Copper and lead concentrations at the site were lowest in the Lake Border lacustrine sediment. Barium and titanium concentrations were highest in the Lake Border lacustrine sediment.

Elements occurring in concentrations exceeding 1 percent of the total mass in most samples include aluminum, calcium, iron, manganese, and potassium. These elements are the primary constituents of many rock-forming minerals. Concentrations of calcium, iron, and manganese were notably elevated in slag; potassium and sodium were relatively depleted. Trends similar to the slag analyses were observed in samples from the slag–aquifer interface, but concentrations generally were lower than slag. Calcium concentrations at the site were lowest in the Lake Border lacustrine sediment. Aluminum and potassium concentrations were highest in the Lake Border lacustrine sediment.

Analysis of the various grain-size fractions from sample BH-32C showed only slight variations in trace-element chemistry. Scandium, aluminum, cobalt, iron, nickel, potassium, sodium, strontium, thorium, uranium, vanadium, yttrium, zinc, and zirconium concentrations showed slightly increasing concentrations with decreasing grain size.

The concentrations of many trace elements in solid-phase samples were less than 100 ppm (0.1 percent of the total mass), including antimony, arsenic, beryllium, cadmium, cobalt, bismuth, gold, lanthanum, molybdenum, niobium, scandium, silver, thorium, tin, titanium, tungsten, uranium, yttrium, and zirconium (table 3, back of report). With the exception of the 2- to 5-µm Lake Border lacustrine sediment sample and the lakebed sediments, antimony levels were less than 5 mg/kg. Two notably high tin concentrations were associated with lakebed samples 34A and 34C. Scandium concentrations were slightly elevated in all Lake Border lacustrine sediment samples.

By comparison, two sediment samples collected during 1995 from the southeastern corner of Lake George contained concentrations of antimony, arsenic, chromium, cobalt, iron, lead, manganese, nickel, vanadium, and zinc more than three times background levels (Roy F. Weston, Inc., 1995). Sediment from the southwestern corner of Lake George had no elevated constituent concentrations relative to background samples (Roy F. Weston, Inc., 1995). One sediment sample from the Lost Marsh had concentrations of arsenic, cadmium, cobalt, copper, iron, lead, nickel, selenium, thallium, vanadium, and zinc at concentrations more than three times greater than background (Roy F. Weston, Inc., 1995).

All elemental concentrations in aquifer samples from Bairstow Landfill were elevated relative to the sample of native sand unaffected by slag drainage. The sample of dune sand, which may have origins that are similar to the lacustrine Calumet aquifer sand, probably has undergone considerable reworking by sedimentary processes and has been exposed to surficial conditions that may have altered its geochemistry.

Figure 8 shows the X-ray diffractograms for five solid-phase samples, stacked according to depth, at BH-32 and BH-33. The intensities of the various X-ray peaks were assumed to indicate changing proportions of the minerals present. The stacked diffractograms indicate that quartz (SiO_2) is present at most depths but is most abundant in slag and least abundant in the Lake Border lacustrine sediment. The primary peak for calcite ($CaCO_3$), $2\theta = 29.37°$, indicates that quartz is present at all depths but is most abundant at the slag–aquifer interface. Dolomite ($CaMg(CO_3)_2$), primary peak at $2\theta = 30.94°$, is similarly present at all depths and is most abundant in slag, least abundant at the slag–aquifer interface, and is present in substantial quantities in the sand immediately below the slag–aquifer interface.

Cristobalite (low temperature SiO_2), barian celestite ($Ba_{0.25}Sr_{0.75}SO_4$), and minrecordite ($CaZn(CO_3)_2$) are less abundant in all solid-phase samples than quartz, calcite, and dolomite. The primary diffractogram peak of low-temperature cristobalite ($2\theta = 21.94°$) indicates that small amounts of this mineral are in the slag samples and that it is less abundant in deeper samples. The primary and secondary diffractogram peaks for barian celestite overlap with the primary quartz and calcite peaks, making barian celestite difficult to distinguish. The identification of barian celestite was based on a prominent lower order peak at slightly more than $2\theta = 44.5°$ but, because other lower-order peaks for celestite are not in the diffractogram for sample 32A, this identification is considered tentative.

The primary peak for minrecordite ($CaZn(CO_3)_2$) overlaps the primary dolomite peak and cannot be identified in the solid-phase samples. The secondary and tertiary peaks, however, are clearly identified at about $2\theta = 50.75°$ and $51.25°$. Minrecordite may be present at all depths, but it is most abundant near the slag–sand interface.

Clay minerals can be an important geochemical consideration because of their cation-exchange capabilities. Clay minerals are not present in significant quantity in any solid-phase sample from the study site. Primary diffractogram peaks for clays generally occur at $2\theta < 20°$; peak intensities in this region were subdued. It is possible that X-ray examination of a finer grain-size fraction would allow clay minerals to be identified. Clay minerals are considered to be allochthonous minerals and their presence nonindicative of secondary mineralization processes.

Figure 9 shows stacked diffractograms for a set of four solid-phase samples taken over approximately a 3-ft lakebed interval at BH-34SS. This sequence may be useful in describing geochemical changes occurring as ground water discharges into Lake George. Similar to the core samples at BH-32 and BH-33, quartz, dolomite, and calcite are the most abundant minerals in the lakebed samples. The stacked X-ray diffractograms may indicate that quartz abundance increases with depth, but calcite and dolomite abundances decrease with depth.

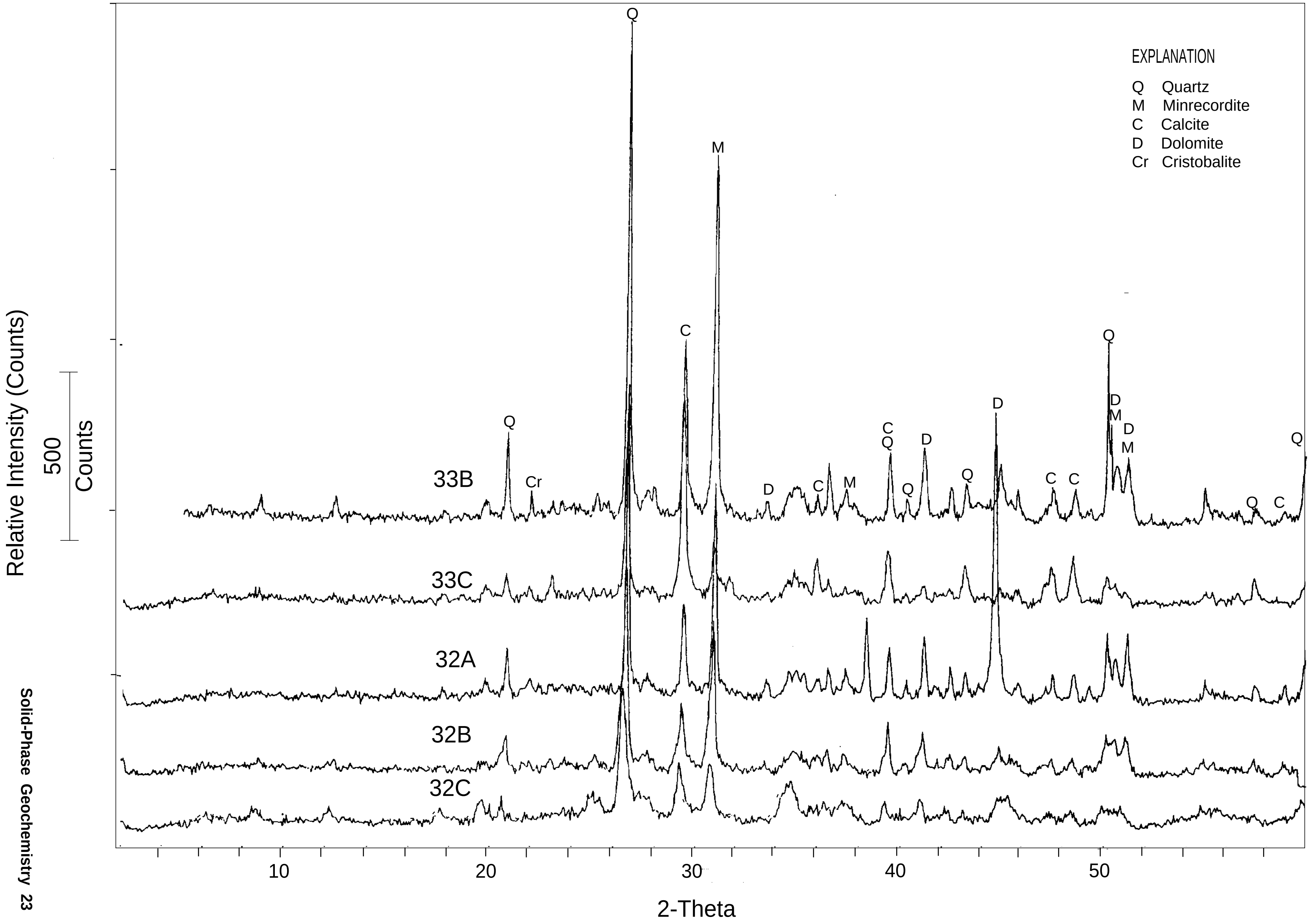


Figure 8. Stacked X-ray diffractograms for samples from sites BH-32 and BH-33 at Bairstow Landfill.

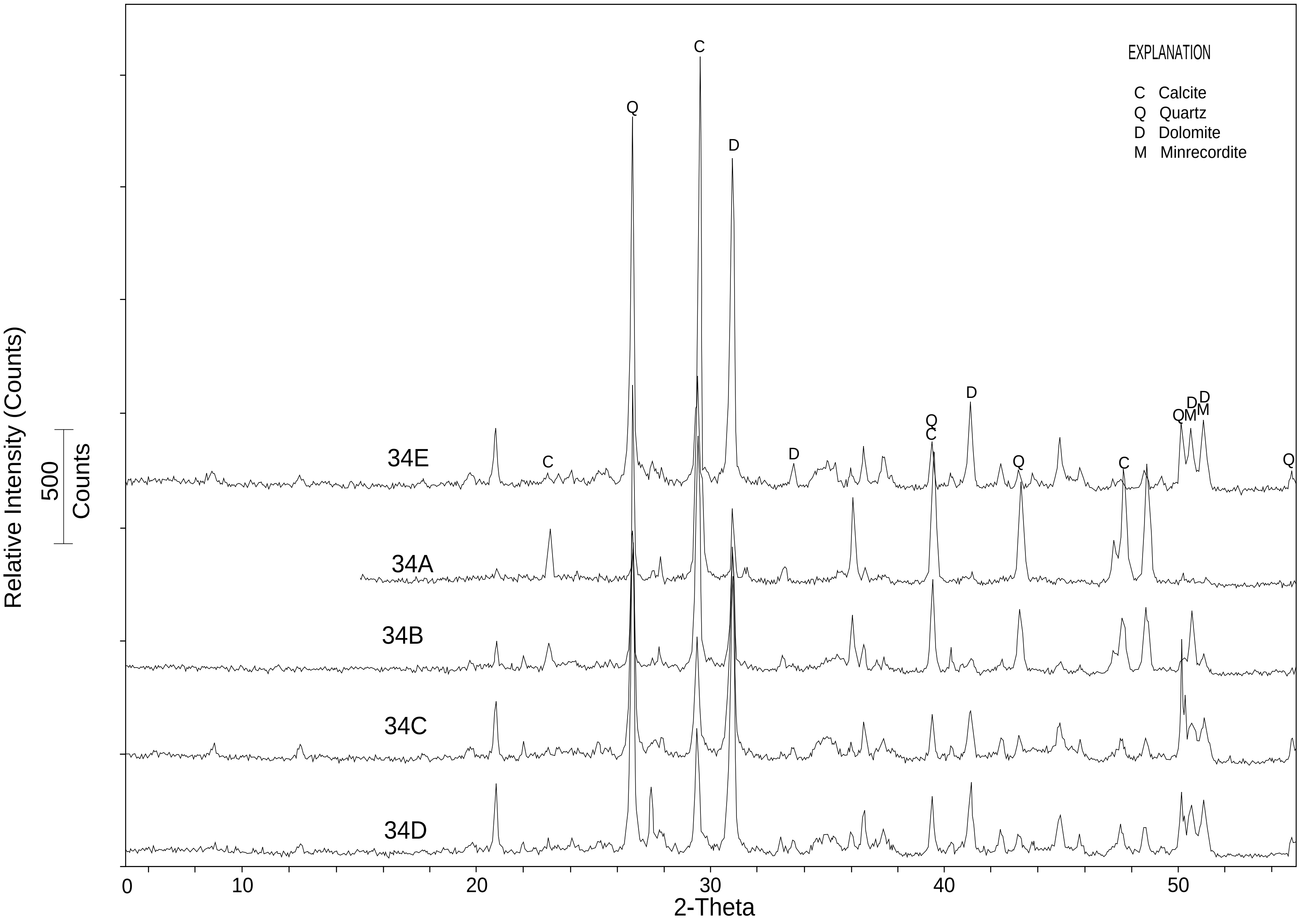


Figure 9. Stacked X-ray diffractograms for samples from site BH-34 at Bairstow Landfill.

Cristobalite may be present in the two middle-depth samples but only as a minor constituent. Minrecordite abundance increases with depth and is almost nonexistent in the shallowest lakebed sample.

A rock sample encrusted with a whitish precipitate from the southern shoreline of Lake George also was examined by X-ray diffraction. The mineralogy of the precipitate consisted predominantly of calcite, with minor quantities of quartz and dolomite and lesser quantities of barian celestite, manganese-bearing calcite, and minrecordite.

Mineral-Saturation Simulations

Constituents used in the WATEQFP calculations are identified in table 2 (back of report) and the calculated saturation indices are shown in table 4 (back of report). Saturation indices for 183 minerals were calculated; the minerals automatically were selected from the WATEQFP data base if the chemical components necessary to build the mineral had positive concentrations in the water-quality sample. For some water-quality-data sets, the saturation indices of some minerals were not calculated because concentrations of the necessary chemical constituents were less than reporting limits. Minerals with no calculated saturation index greater than 10^{-5} are omitted from table 4.

Calculated saturation indices show that many minerals are undersaturated in all water samples; these minerals are not thermodynamically favored to precipitate in the Bairstow Landfill ground-water system and may dissolve if present. Of 108 minerals shown in table 4 (back of report), 49 were undersaturated in all samples. Eleven of 108 minerals were supersaturated in all samples, and 46 minerals were supersaturated in some samples (table 4). Of the 46 minerals that occasionally are supersaturated, most indicate a tendency to precipitate in the Calumet aquifer and at site BH-34ND. Minerals were less frequently supersaturated in the four slag wells and in the sample from Lake George.

The saturation states of minerals commonly identified in X-ray diffractograms for solid phase samples from Bairstow Landfill (including calcite, dolomite, quartz, and less commonly cristobalite) indicate a dependency on solution pH (fig. 10). The saturation indices of calcite and dolomite increase with pH, but the saturation indices of quartz and cristobalite become negative at high pH.

Calcite is supersaturated at all well points. Low pH precipitation that has been observed in northwestern Indiana (Willoughby, 1995, p. 14 and p. 30) may react with near-surface slag and may be responsible for the tremendous quantities of dissolved calcium and carbonate present in slag. As ground water percolates into the aquifer, calcium and carbonate concentrations are lowered by dilution and (or) calcite precipitation; as a result, the calcite saturation indices are lowered.

Calcite is undersaturated at the Lake George sample point; the constituents required to form calcite are depleted in lake water by dilution and (or) mineral precipitation at the ground-water/surface-water interface. Active mineral precipitation was observed along the lakeshore of Lake George, and X-ray diffraction analysis indicates that calcite is the primary phase.

Dolomite is saturated at all sample points, except at well BH-32I. The solution pH was lowest for all wells at BH-32I; dolomite-forming constituents were not particularly low at this well site. Dolomite saturation indices were not calculated for monitoring wells with magnesium concentrations less than the method reporting limit, including wells BH-31, BH-33SL, and BH-34SS. Dolomite, however, was observed in X-ray diffraction analyses of slag samples, indicating that dolomite is present but may be insoluble in slag. Dolomite precipitation is generally believed to require geologically long time periods.

Cristobalite and quartz were supersaturated at all sample points, except the four slag wells. Silica is generally soluble at pH values in excess of 9.9 (Krauskopf, 1979), and the four slag wells had pH values greater than 11.9. The pH of all other

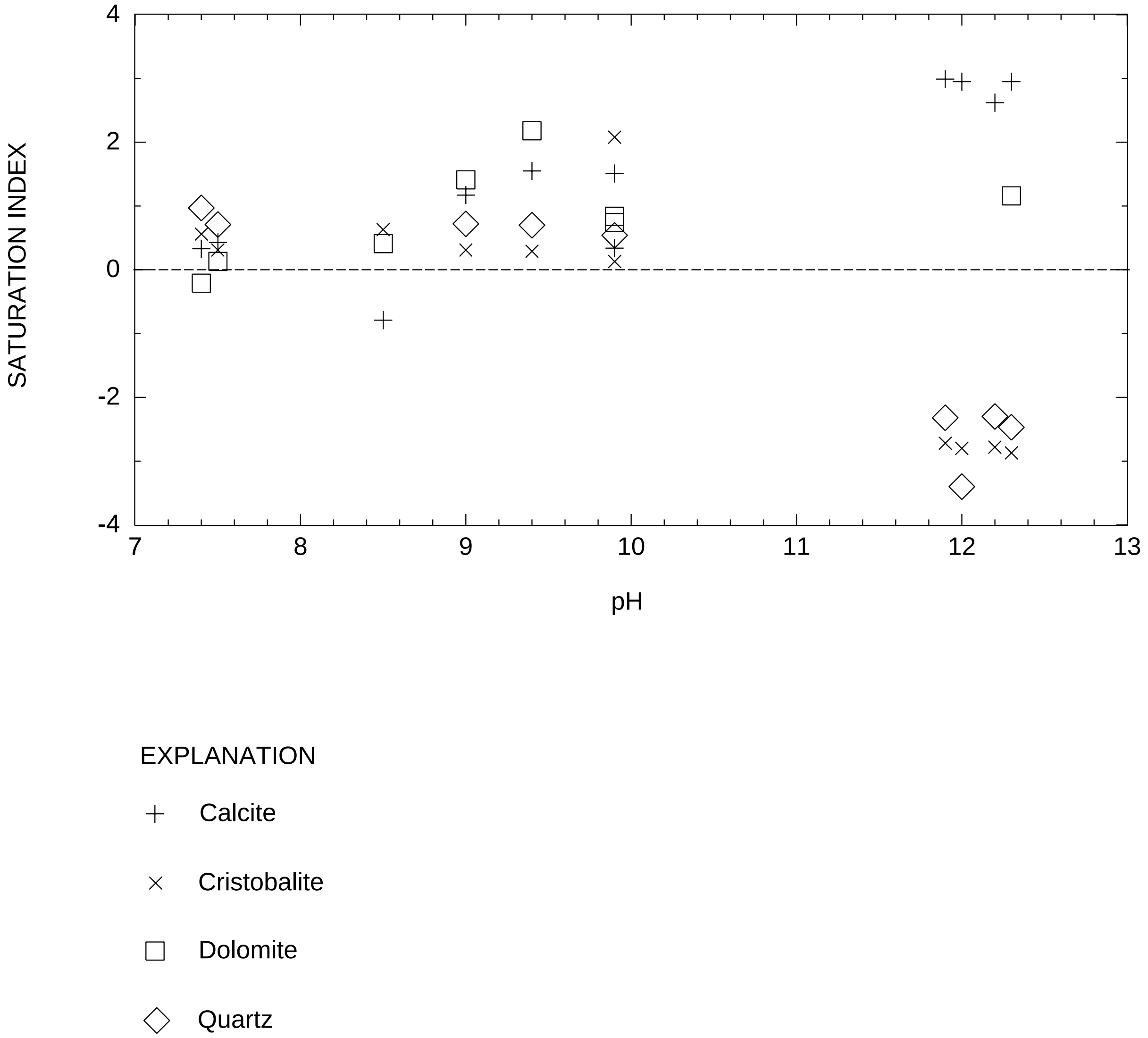


Figure 10. Plot of saturation index by pH and total dissolved solids for all wells at Bairstow Landfill.

sample points was less than 9.9. The pH data and calculated mineral-saturation indices imply that silica forms would be dissolving in slag and precipitating in the shallow aquifer.

Several trace carbonate minerals, notably rhodochrosite ($MnCO_3$) and siderite ($FeCO_3$), also were undersaturated in the slag but were slightly undersaturated to slightly supersaturated in aquifer wells below the slag. These minerals were not present in quantities sufficient to be detected by the X-ray diffraction analyses but may be precipitation controls on manganese and iron concentrations in water.

The ability of WATEQFP to calculate accurately the tendency for a mineral to precipitate or dissolve is constrained by several factors, including the precision of water-quality analyses, the accuracy of the thermodynamic data base, and the kinetics of mineral formation. The water-quality input data must (1) include concentrations for chemical constituents that are significant components of most minerals in the system and (2) be analytically accurate.

The accuracy of water-quality analyses commonly is assessed by calculating charge balances. The charge balances for the Bairstow samples ranged from +4.97 to –15.4 percent and had median value –4.31 percent. The most extreme imbalances were measured in the four slag wells, with magnitudes of +4.97, –15.9, –10.4 and –15.4 percent. Doss (1996) similarly identified relatively large errors in slag-affected ground-water-quality analyses. The charge balances for wells screened in the Calumet aquifer and in the sample from Lake George ranged from –1.41 to –6.50 and had a median value of –3.66.

Generally, charge balances in water analyses that are greater than 5 percent or are less than –5 percent are considered to be missing an important constituent or to be the result of inaccurate laboratory analyses or inappropriate field methods. Replicate analyses done as part of this study indicate that the field and laboratory methods produced constituent concentrations and water-quality characteristics that were nearly identical; however, the possibility of a consistent error in methods is not eliminated. Potential sources of error in the results of analyses of samples from Bairstow Landfill analyses may include, but are not limited to, methods used to determine carbonate alkalinity, changes in water quality between sampling and analysis, and complexing of cations by organic constituents and colloids (Hem, 1989). The extreme and unique water-quality conditions at the Bairstow Landfill, particularly at the four slag wells, may make these samples unstable at land surface and require non-standard methods of preservation and analysis.

In addition to the errors introduced by sampling, preservation, and analysis, saturation indices calculated by WATEQPF may be affected by inaccuracies in the thermodynamic data base (Ball and Nordstrom, 1991). Thermodynamic data generally are more reliable for common rock-forming minerals with simple mineral structure but may contain significant errors for more complex and rare minerals.

Mass-Balance Simulations

The observation-well transect at the Bairstow Landfill intentionally was aligned parallel to the direction of ground-water flow so that mass balances of dissolved constituents could be calculated. The main flow components simulated by the calculations included (1) horizontal flow conditions along the south-to-north ground-water flowpath toward Lake George, (2) flow of recharge down through the slag pile and uppermost sands into the intermediate-depth sands, and (3) ground-water flow up from the deeper sands to converge with shallow ground-water flow at the intermediate-depth sands. When reviewing results of the mass-balance calculations, the alternate, occasional possibility of flow reversals also needs to be considered relative to the apparent precision of the results.

The solid phases included in the mass-balance models were limited to the minerals detected in significant quantities in the X-ray diffraction analyses—calcite, dolomite, and quartz (expressed as SiO_2 in NETPATH). The minerals minrecordite, cristobalite, and barian celestite were not part of the NETPATH data base and, because they were only detected infrequently and in relatively small abundances in X-ray diffractograms, they were not added to the NETPATH data base. Other phases that exist in the NETPATH data base were not considered in the Bairstow Landfill models because (1) they were not identified in the X-ray diffraction analyses, and (2) the relatively minute abundances of other minerals probably disqualifies them from having a major impact on the geochemical system.

Calcium, magnesium, and silica concentrations were used as constraints for the mass-balance models. Chloride was used as a fourth constraint of mixing proportions calculated for models where ground water from two wells combined to form the water quality observed at a third well. In all models, the situation’s number of constraints equalled the degrees of freedom to obtain a unique solution.

The non-mixing mass-balance models indicated that slag drainage into the Calumet aquifer may precipitate calcite near the slag–aquifer interface (fig. 11). The simulated masses of calcite precipitated from solution, 4.92 and 5.69 millimoles per kilogram of water, as ground water moves from BH-31 to BH-32S and from BH-32SL to BH-33S. These mass transfers were large relative to others calculated for this model. The mass-balance models also indicated that calcite may be precipitating as ground water moves from the deeper aquifer (BH-32D) to the intermediate-depth aquifer (BH-33I) and as ground water flows horizontally, such as from BH-32I to BH-33I. The possibility that changes in concentrations of calcium and carbonate can be attributed to dilution of slag-contact ground water with less strongly affected ground water is addressed by the mixing mass-balance models. The calcite-rich precipitate observed along the Lake George shoreline confirms that some calcite precipitation is occurring; the mass transfer between BH-34SS and Lake George is 5.74 millimoles per kilogram of water.

The non-mixing mass-balance calculations indicate that dolomite is neither precipitating nor dissolving in significant quantities; mass transfers range from –1.56 to +1.13 millimoles per kilogram of water. The models indicate that dolomite dissolution may be occurring where ground water is flowing from slag into the shallow aquifer or into Lake George. The mass-balance models also indicate that dolomite may be precipitating where ground water flows from the shallow aquifer (BH-32S) and deeper aquifer (BH-32D) to the intermediate-depth aquifer (BH-33I); however, laboratory studies have shown that dolomite formation occurs by calcite replacement where magnesium/calcium is greater than 100 and that the process is relatively slow (Ehlers and Blatt, 1980).

The mass-balance model indicates that silica, present as quartz and cristobalite in slag and as quartz in the aquifer, may be dissolving from the aquifer matrix where ground water flows from the slag into the shallow aquifer. Dissolution of silicate minerals where slag drainage enters the aquifer ranges from 0.40 to 0.46 millimoles per kilogram of water, approximately 10 times less than the transfer rate for calcite. The transfer rate actually may be lower than the model calculated values because silica concentrations in BH-32D indicate that a significant silica concentration may naturally be present in the Calumet aquifer; therefore the silica concentration in the aquifer is not entirely the result of quartz and cristobalite dissolution. The low concentrations of dissolved silica in the slag-contact wells, despite the abundance of quartz and cristobalite in X-ray diffractograms and pH values in excess of 12, imply that quartz dissolution may be kinetically inhibited.

Mass-balance calculations indicate that silica may precipitate where ground water discharges from the aquifer into Lake George; the mass transfer for ground water flowing from BH-34ND to the Lake George sample point is –1.18 millimoles of silica per kilogram of water. A more likely process to account for the change in silica concentrations

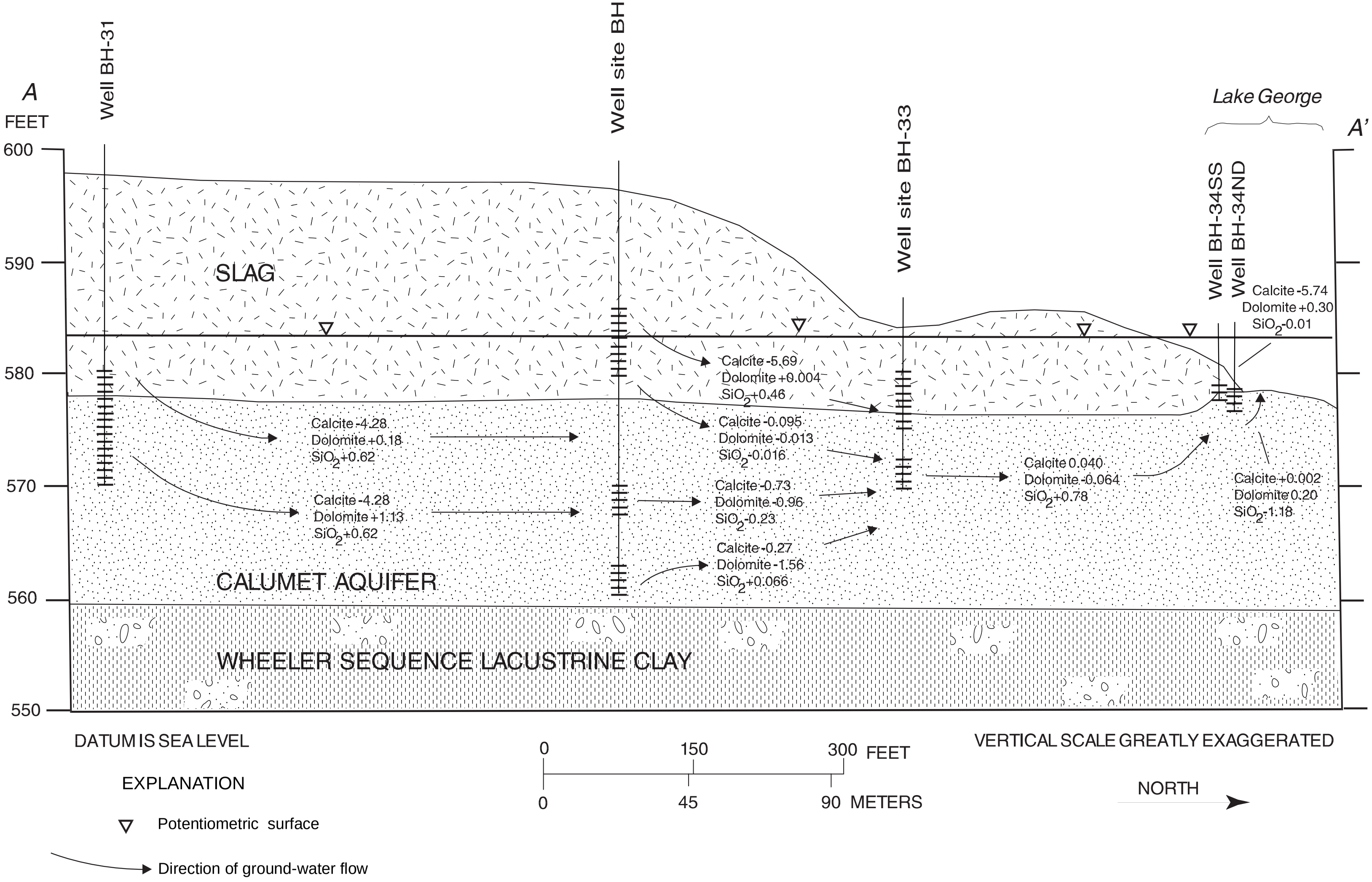


Figure 11. Mass exchanges between aqueous and solid phases along a ground-water flowpath at Bairstow Landfill, Hammond, Ind. Negative values indicate mineral precipitation.

across the ground-water/surface-water interface is dilution and dispersion of silica in the more dilute water of Lake George. Some combination of the two processes also may explain the observations.

Mass-balance models also were developed to evaluate the importance of ground-water mixing relative to concomitant mineral precipitation and dissolution to explain the observed changes in ground-water quality along the Bairstow Landfill flowpath. For example, the water quality at BH-32S alternately may be produced by mixing ground water from BH-31 and BH-32SL or by precipitating or dissolving calcite, dolomite, and silica.

Concentrations of chloride were used as a model constraint to determine the mixing proportions. Chloride is recognized as a conservative ion, and changes in concentration are likely attributable to ground-water mixing. Besides chloride, all other constraints (calcium, magnesium, and silica) and phases (calcite, dolomite, and SiO_2) remained the same as those used in the previously described non-mixing models.

The results of mixing mass-balance models are expressed as equations that relate mixing proportions of the two initial well chemistries to produce the water quality at the final downgradient well. Ground-water flowpaths determined from water-level data were used to select four potential mixing situations at the study site. Results of these models are presented in table 5.

Mixing model 1 explores the possibility that ground-water quality at well BH-32S is a result of combining water from BH-31 and BH-32SL (table 5). The calculations indicate that approximately 97 parts BH-31 ground water and 3 parts BH-32SL water plus calcite precipitation of 4.94 millimoles per kilogram of water and dissolution of dolomite and SiO_2 at 0.180 and 0.407 millimoles per kilogram of water, respectively, would yield the ground water observed at well BH-32S. These mass transfers are similar to the previously calculated two-component flow system from BH-31 to BH-32S.

Mixing model 2 investigated the possibility that ground-water quality at well BH-32I is a result of mixing ground water from BH-31, BH-32S, and BH-32D; these flowpaths are indicated by water-level data. Model results indicate the water quality at BH-32I can be simulated by mixing about 11 parts of water from BH-31, 38 parts of water from BH-32S, and 51 parts of water from BH-32D, with a net dissolution of 0.186 and 0.304 millimoles per kilograms of water of dolomite and SiO_2, respectively.

Mixing model 3 attempted to explain the ground-water quality at well BH-33S by mixing water from wells BH-32S, BH-33SL, and BH-33I. A model could not be calculated based on the water-quality information measured in these wells. The high chloride concentration measured in water from well BH-33S was a limiting factor in calculating a satisfactory model. None of the initial ground-water compositions contained sufficient chloride to define the high chloride concentrations in BH-33S ground water.

Mixing model 4 derived the water quality from well BH-33I by mixing water from wells BH-32S, BH-32I, and BH-33S. Ground-water levels did not indicate flow from BH-32I to BH-33I, but horizontal flow of water with properties similar to BH-32I toward BH-33I may commonly occur. Results indicated that water quality at well BH-33I could be simulated by mixing about 1 part water from BH-32I with 99 parts water from BH-33S, dissolving 0.033 and 0.085 millimoles per kilogram of water of calcite and SiO_2, respectively, and precipitating 0.151 millimoles dolomite per kilogram of water. Calculations indicate that ground water at BH-33I did not require a contribution from well BH-32S. Water-level data did not indicate a significant component of flow from BH-33S to BH-33I, but the mixing model indicates that this may be the most common situation. Measurements of water levels were taken only on one occasion and may not represent all details of the prevalent flow regime.

Table 5. Geochemical mixing models and mass-transfer estimates along ground-water flowpaths at Bairstow Landfill, Hammond, Ind.

[mmol/kg, millimoles per kilogram of water; SiO_2, quartz or cristobalite; --, not used or calculated]

Mixing model	Initial Well 1	Initial Well 2	Initial Well 3	Final Well	Simulated volume of water from initial well 1 in final well (percent)	Simulated volume of water from initial well 2 in final well (percent)	Simulated volume of water from initial well 3 in final well (percent)	Simulated mass transfer of calcite (mmol/kg)	Simulated mass transfer of dolomite (mmol/kg)	Simulated mass transfer of SiO_2 (mmol/kg)
1	BH-31	BH-32SL	--	BH-32S	96.6	3.36	--	-4.94[1]	0.180	0.407
2	BH-31	BH-32S	BH-32D	BH-32I	11.1	38.0	50.8	.000	.186	.304
3	BH-32S	BH-33SL	BH-33I	BH-33S	no models	--	--	--	--	--
4	BH-32S	BH-32I	BH-33S	BH-33I	0.000	1.07	98.9	-.033	.151	-.085

[1] Negative mass transfer indicates mineral precipitation from the dissolved solids in the mixed waters. Positive mass transfer indicates mineral dissolution contributing dissolved solids to the mixed waters.

No mixing combination could account for the chloride concentration in BH-33S because the value in that well exceeded the concentration in both wells immediately upgradient. Similarly, an appropriate mixing model could not be found by combining ground water at BH-34ND and BH-34SS to derive the water quality in the sample from Lake George because the chloride in the lake is higher than in both ground-water samples. Lake George likely has a source of chloride, such as from road-deicer-affected runoff, in addition to ground-water discharge.

Several situations can lead to inaccuracies in mass-balance models, including:

1) selection of water-quality data from observation wells that are not along the same ground-water flowpath,
2) analytical mass-balance errors,
3) hydrodynamic dispersion in the aquifer,
4) departure from steady-state conditions,
5) significant and unaccounted recharge (dilution) or evaporation during the period that the ground water flows from the initial to final observation well, and
6) unknown or unquantified ion exchange and mineral impurities.

CHEMICAL CHARACTERISTICS OF SLAG-AFFECTED GROUND WATER

The statistical summary of the water-quality data identified properties and constituents that have been influenced by ground-water interaction with slag. Table 6 (back of report) presents calculated values of the total number of measurements in the data set, the median (middle) data value, the minimum and maximum values from the data set, and the first and third quartile (Q1 and Q3) that are the data values at the 25th and 75th positions of a hierarchical ordering of the data.

The statistical summary indicates that water-quality properties that generally are elevated in water from slag-contact wells include alkalinity, pH, and specific conductance. Alkalinity generally is elevated in water from slag-contact wells, relative to other ground-water settings, but not all slag-contact wells have high alkalinity values. Alkalinity values range from 24.9 to 884 mg/L in water from slag-contact wells, with a median of 279 mg/L. Almost all of the highest alkalinity values are from slag-contact wells; however, the highest alkalinity value in the data set was measured at a hazardous-waste-disposal site. The most consistently observed effect of slag on ground water is elevated pH. The pH values observed in water from 21 slag-contact wells ranged from 6.60 to 12.3, with median of 8.70. Water from slag-affected wells also has higher pH values, ranging from 6.9 to 9.90 than values in background water that ranged from 6.0 to 7.70. Specific conductance values, a bulk measure of dissolved solids, also are highest in water from slag-contact wells.

Numerous chemical elements can be elevated in slag-contact wells. Major ions with higher concentrations in water from slag-contact wells include calcium, potassium, sodium, chloride, and occasionally magnesium and sulfate. The median calcium concentration is more than three times greater in water from slag-contact wells than from background wells. The median potassium concentration in water from slag-contact wells (25.1 mg/L) was more than 20 times greater than from background wells. Potassium concentrations in water from slag-contact wells range from 3.12 to 170 mg/L. More than 75 percent of water samples from background wells contained less potassium than the lowest measured concentration in all slag-contact samples. The median sodium concentration in water from slag-contact wells is eight times greater than in background ground water, and the median chloride concentration in slag-contact wells (69 mg/L) is more than five times that of water from background wells. Magnesium concentrations in slag-contact ground water vary the most, ranging from 0.010 to 211 mg/L, but the median value is about 5.0 mg/L less than the median concentration of water from slag-affected wells (22.8 mg/L). The median sulfate concentration in slag-contact wells is 190 mg/L, compared to 72 and 27.3 mg/L in slag-affected and background wells, respectively, but the highest measured value occurred in a slag-affected well.

Trace elements having higher concentrations in water from slag-contact wells include aluminum and barium and less frequently chromium, cobalt, copper, cyanide, manganese, mercury, nickel, and vanadium. The median aluminum concentration in slag-contact ground water is 49.7 µg/L compared to 23.5 and 23.5 µg/L for water from slag-affected and background wells, respectively. The median barium concentration in slag-contact ground water is 128 µg/L compared to 58.0 and 31.1 µg/L for slag-affected and background ground water, respectively. Chromium, cobalt, copper, cyanide, manganese, nickel, and vanadium in slag-contact ground water have median concentrations that are about the same as slag-affected and background ground water. The range of chromium, cobalt, copper, cyanide, manganese, nickel, and vanadium concentrations in slag-contact ground water, however, is more variable than in slag-affected or background ground water; usually the maximum concentration is several times higher in water from slag-contact wells. For example, the median concentration of cyanide, a trace species, for all ground water is 10.0 µg/L (the minimum reporting limit) but the maximum concentrations for slag-contact, slag-affected, and background ground water are 212, 31.4, and 10.0 µg/L, respectively. Some trace elements or species like cyanide are not present in solid slag and may indicate possible disposal with industrial waste.

Total organic carbon (TOC) and suspended-solids concentrations also are high in slag-contact ground water relative to slag-affected and background ground water. Concentrations of TOC in slag-contact samples range from 3.70 to 209 mg/L, with median value 15.1 mg/L, compared to concentrations of TOC in background ground water that range from <2.00 to 26.9 mg/L, with a median of 4.80 mg/L. Most ground water contains organic compounds in the form of humic and fulvic acids; however, high TOC concentrations in slag-contact ground water may indicate disposal of slag and human-affected organic compounds.

Suspended-solids concentrations in slag-contact ground water ranged from <3.00 to 5,310 mg/L, with a median value of 54.0 mg/L, compared to the range of <3.00 to 15.5 mg/L and the median of 3.0 mg/L in background ground water. High suspended-solids concentrations also may represent particulate oxides and hydroxides of aluminum, iron, and manganese (Hem, 1985, p. 60).

Although many ions are elevated in the slag-contact ground water, several chemical elements are more highly concentrated in slag-affected ground water. For example, arsenic concentrations in slag-affected ground water ranged from <1.70 to 55.0 µg/L, while maximum values in slag-contact and background samples were 22.6 µg/L and 12.8 µg/L, respectively. The largest silica concentrations also were from slag-affected ground water.

Chemical elements not commonly detected in slag-contact wells include antimony, beryllium, cadmium, mercury, silver, and thallium. Lead is detected only randomly in any wells and may be present from atmospheric deposition. Zinc, also randomly detected, may represent industrial wastes deposited with slag. Zinc detections in background wells correspond to wells cased with galvanized steel.

CONCLUSIONS

Several conclusions can be drawn from this study of slag effects on aquifer geochemistry and ground-water quality:

(1) Concentrations of calcium, potassium, sodium, and sulfate were highest in water from wells screened partially or fully in slag. Potassium concentrations in ground water ranged from 2.9 to 120 mg/L, were highest in water from slag deposits, and decreased with depth. Silica concentrations were highest in wells screened directly beneath the slag–aquifer interface, and magnesium concentrations were highest in intermediate and deep aquifer wells. Silica concentrations in shallow and intermediate aquifer wells ranged from 27 to 41 mg/L and were about 10 times greater than those in water from slag deposits. The highest concentrations for aluminum, barium, molybdenum, nickel, and selenium were in the slag. The highest concentrations for chromium, lead, and zinc were in ground water from immediately below the slag–aquifer interface. Nitrite in ground-water from slag and ammonium in ground water from below the slag were the most abundant nutrients.

(2) Concentrations of several major ions and trace elements in water relate to pH values and bicarbonate concentrations and their effect on mineral-water interactions. For example, the samples with the largest concentrations in water of calcium, potassium, sodium, sulfate, aluminum, barium, molybdenum, nickel, and selenium, also have pH values ranging from 11.9 to 12.3 and are from slag wells. By comparison, water from wells with larger concentrations of silica and smaller concentrations of calcium also have smaller pH values. Water with the largest concentrations of chromium, lead, and zinc also generally have pH values that range from 9.0 to 9.9. Bicarbonate concentrations, as represented by alkalinity, are substantially greater in water from slag wells. Larger alkalinity values favor greater partial pressures of carbon dioxide and the dissolution of potassium and sodium aluminosilicate minerals (Stumm and Morgan, 1981, p. 545), possibly explaining the elevated potassium and sodium concentrations in slag-affected water.

(3) The concentrations of several chemical constituents and values of water-quality properties consistently are elevated in slag-contact ground water relative to regional background water quality; however, spuriously high values also can occur in slag-affected ground water. Consistently elevated water-quality properties in slag-contact wells included alkalinity, dissolved solids, suspended solids, total organic carbon, pH, and specific conductance. Major ions that consistently are elevated in slag-contact wells include calcium, potassium, sodium, chloride, and occasionally magnesium and sulfate. Trace elements having high concentrations in slag-contact wells include aluminum and barium and occasionally chromium, cobalt, copper, cyanide, manganese, mercury, nickel, and vanadium. Arsenic and silica concentrations generally are higher in slag-affected ground water than in slag-contact ground water.

(4) X-ray diffraction analyses of solid-phase samples indicate that the minerals calcite, dolomite, and quartz are relatively abundant and that barian celestite, cristobalite, manganese-bearing calcite, and minrecordite may occur infrequently.

(5) Of the 183 minerals examined with WATEQFP, 57 are supersaturated in at least one ground-water sample from Bairstow Landfill. Eleven minerals consistently were supersaturated, including calcite and dolomite. Quartz was not supersaturated in slag-contact wells, despite its abundance in X-ray diffractograms; both factors may indicate that quartz dissolution is a kinetically retarded process. Saturation indices for other minerals observed in X-ray diffractograms (barian celestite, cristobalite, and minrecordite) could not be calculated because the WATEQFP data base did not include the necessary thermodynamic information.

(6) Mass-balance models examined mass exchanges occurring along a ground-water flowpath at Bairstow Landfill. Results indicate that calcite precipitation is the dominant geochemical reaction where slag drainage enters the aquifer.

Carbonate-mineral precipitation also may be occurring where slag drainage discharges to Lake George. Mass-balance models indicate that silica is dissolving where high-pH slag drainage enters the aquifer and continues to be an active process for some distance downgradient. Silica may be precipitating, probably as an amorphous form, where deeper ground water discharges to Lake George or concentrations of silica may be lowered in the lake as a result of dilution with lake water. Dolomite is dissolved in small quantities in all mixing mass-balance models. The minerals minrecordite, barian celestine, and cristobalite were not included in the mass-balance models because their presence in solid-phase analyses was infrequent.

(7) Many chemical elements present in solid-phase analyses are not primary elements in the minerals identified in X-ray diffractograms. These elements may not have been identified if they occur as amorphous compounds, as impurities in abundant rock-forming minerals or clay minerals, or if they predominantly occur in grain-size fractions other than those examined in this study.

(8) The water quality of Lake George reflects the influence of ground-water discharges from Bairstow Landfill, as well as non-slag-related chemical inflows to the lake from sources other than the landfill. Active precipitation of carbonates in the aquifer and at the ground-water/surface-water interface may remove some trace elements from solution and incorporate them as mineralogic impurities; the identification of barian celestite and manganese-bearing calcite in a sample of lakeshore precipitate may be evidence of this process.

(9) Concentrations of most elements in the solid-phase geochemical analysis of samples from Bairstow Landfill exceeded the values in a sample of “background” sand from the Indiana Dunes National Lakeshore. The elevated concentrations in sand from the Calumet aquifer below the Bairstow Landfill may indicate that the slag deposit has affected the entire aquifer beneath the landfill or the dune sand has been leached of accessory minerals by surficial processes.

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SUPPLEMENTAL DATA

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996

[USGS, U.S. Geological Survey; ^{o}C, degree Celsius; mg/L, milligram per liter; µS/cm, microsiemens per centimeter at 25 degrees Celsius; $CaCO_3$, computed as calcium carbonate; --, no data; five-digit number in parentheses is WATSTORE (USGS) and usually the STORET (U.S.Environmental Protection Agency) parameter code; µg/L, microgram per liter; <, less than]

Well name	USGS site identification number	Date sampled	Time sampled	Temperature, on site[1] (^{o}C) (00010)	pH, on site[1] (standard units) (00400)	Redox potential, on site[1, 2] (millivolts) (00095)	Dissolved oxygen, on site[1] (mg/L) (00300)	Specific conductance, on site[1] (µS/cm) (00095)	Alkalinity, on site[1] (mg/L as $CaCO_3$) (00419)
BH-31	413947087302501	08-15-96	1010	14.5	12.2	-94	0.4	3,180	626
BH-32D	413949087301901	08-13-96	1255	15.0	7.5	+38	.1	877	420
BH-32I	413949087301902	08-13-96	1117	14.5	7.4	+53	.2	844	425
BH-32S	413949087301903	08-12-96	1645	14.5	9.3	-82	.1	891	447
		08-13-96	940	14.5	9.4	-65	.1	895	440
		08-13-96	1010	14.5	9.4	--	.1	895	--
BH-32SL	413949087301904	08-12-96	1516	14.5	12.3	-127	.3	4,640	903
BH-33I	413951087301901	08-14-96	1510	16.0	9.0	-26	<.1	1,030	460
BH-33S	413951087301902	08-14-96	1345	16.5	9.9	-100	<.1	1,300	590
BH-33SL	413951087301903	08-14-96	1215	20.0	11.9	-77	.2	3,930	785
BH-34ND	413954087301901	08-15-96	1220	23.5	9.9	+24	1.6	847	230
BH-34SS	413954087301902	08-13-96	1522	22.0	12.0	-64	.3	3,720	850
Lake George	413954087301903	08-13-96	1620	28.5	8.5	+294	6.5	733	160

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996—Continued

Well name	Date sampled	Time sampled	Calcium, dissolved[1] (mg/L as Ca) (00915)	Magnesium, dissolved[1] (mg/L as Mg) (00925)	Potassium, dissolved[1] (mg/L as K) (00935)	Sodium, dissolved[1] (mg/L as Na) (00930)	Chloride, dissolved[1] (mg/L as Cl) (00940)	Fluoride, dissolved[1] (mg/L as F) (00950)	Sulfate, dissolved[1] (mg/L as SO_4) (00945)	Silica, dissolved[1] (mg/L as SiO_2) (00955)
BH-31	08-15-96	1010	210	<0.03	81	55	28	0.4	91	3.5
BH-32D	08-13-96	1255	88	41	2.9	34	19	.3	67	23
BH-32I	08-13-96	1117	83	27	3.7	50	24	.3	7.6	41
BH-32S	08-12-96	1645	20	3.6	19	170	29	1.1	6.7	28
BH-32S	08-13-96	940	21	4.4	20	170	29	1.0	6.5	26
BH-32S	08-13-96	1010	21	3.7	18	170	29	1.0	6.9	27
BH-32SL	08-12-96	1516	240	.01	120	170	53	.6	140	3.9
BH-33I	08-14-96	1510	16	3.9	24	190	67	.9	41	27
BH-33S	08-14-96	1345	11	.12	92	220	66	.8	25	32
BH-33SL	08-14-96	1215	270	<.03	100	130	40	.9	220	3.1
BH-34ND	08-15-96	1220	15	2.4	81	98	54	.7	110	74
BH-34SS	08-13-96	1522	250	<.03	79	87	35	.6	110	3.8
Lake George	08-13-96	1620	23	7.2	46	86	69	.8	110	2.9

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996—Continued

Well name	Date sampled	Time sampled	Nitrogen, ammonia, total[1] (mg/L as N) (00610)	Nitrogen, nitrite, total[1] (mg/L as N) (00615)	Nitrogen, ammonia+ organic, total (mg/L as N) (00625)	Nitrogen, nitrite +nitrate, total (mg/L as N) (00630)	Phosphorus, total (mg/L as P) (00665)	Phosphorus, ortho, total[1] (mg/L as P) (70507)	Aluminum, dissolved[1] (µg/L as Al) (01106)	Antimony, dissolved (µg/L as Sb) (01095)
BH-31	08-15-96	1010	1.7	2.9	2.6	3.1	<0.01	<0.01	180	<2
BH-32D	08-13-96	1255	2.6	.01	2.9	<.05	.08	.07	6	<1
BH-32I	08-13-96	1117	2.6	.01	2.7	<.05	.08	.11	8	<1
BH-32S	08-12-96	1645	7.1	.04	13	.07	.63	.52	80	<1
BH-32S	08-13-96	940	7.2	.04	13	.08	.50	.51	110	<1
BH-32S	08-13-96	1010	7.2	.03	13	.06	.54	.57	110	<1
BH-32SL	08-12-96	1516	2.0	1.1	3.4	1.1	.03	.01	820	<2
BH-33I	08-14-96	1510	12	.10	16	.19	.46	.47	30	<1
BH-33S	08-14-96	1345	13	.02	20	.07	.39	.36	160	<1
BH-33SL	08-14-96	1215	1.4	.74	2.6	.77	<.01	<.01	170	<2
BH-34ND	08-15-96	1220	3.4	.02	3.9	.10	.04	.04	20	<1
BH-34SS	08-13-96	1522	2.2	.40	3.4	.46	<.01	<.01	320	<2
Lake George	08-13-96	1620	.27	.04	.9	.10	<.01	<.01	50	11

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996—Continued

Well name	Date sampled	Time sampled	Arsenic, dissolved (µg/L as As) (01000)	Barium, dissolved[1] (µg/L as Ba) (01005)	Beryllium, dissolved (µg/L as Be) (01010)	Cadmium, dissolved[1] (µg/L as Cd) (01025)	Chromium, dissolved[1] (µg/L as Cr) (01030)	Cobalt, dissolved[1] (µg/L as Co) (01035)	Copper, dissolved[1] (µg/L as Cu) (01040)	Iron, dissolved[1] (µg/L as Fe) (01046)
BH-31	08-15-96	1010	<1	180	<2	<2	<2	<2	<2	290
BH-32D	08-13-96	1255	55	200	<1	<1	<1	<1	<1	2,600
BH-32I	08-13-96	1117	7	180	<1	<1	4	1	<1	2,500
BH-32S	08-12-96	1645	15	40	<1	<1	5	<1	<1	240
BH-32S	08-13-96	940	19	44	<1	<1	4	<1	3	360
BH-32S	08-13-96	1010	18	45	<1	<1	2	<1	2	230
BH-32SL	08-12-96	1516	<1	410	<2	<2	<2	<2	<2	12
BH-33I	08-14-96	1510	19	41	<1	<1	2	<1	3	510
BH-33S	08-14-96	1345	43	24	<1	<1	3	<1	1	870
BH-33SL	08-14-96	1215	<1	220	<2	<2	<2	<2	3	91
BH-34ND	08-15-96	1220	8	24	<1	<1	<1	<1	<1	34
BH-34SS	08-13-96	1522	1	320	<2	<2	<2	<2	<2	160
Lake George	08-13-96	1620	3	41	<1	<1	<1	<1	2	74

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996—Continued

Well name	Date sampled	Time sampled	Lead, dissolved[1] (µg/L as Pb) (01049)	Manganese, dissolved[1] (µg/L as Mn) (01056)	Molybdenum, dissolved[1] (µg/L as Mo) (01060)	Nickel, dissolved[1] (µg/L as Ni) (01065)	Selenium, dissolved[1] (µg/L as Se) (01145)	Silver, dissolved[1] (µg/L as Ag) (01075)	Uranium, dissolved[1] (µg/L as U) (22703)	Zinc, dissolved[1] (µg/L as Zn) (01090)
BH-31	08-15-96	1010	<2	<2	58	9	4	<2	<2	2
BH-32D	08-13-96	1255	<1	97	2	3	<1	<1	<1	3
BH-32I	08-13-96	1117	<1	130	9	4	<1	<1	<1	3
BH-32S	08-12-96	1645	5	25	35	3	<1	--	<1	8
BH-32S	08-13-96	940	10	45	34	4	<1	<1	<1	14
BH-32S	08-13-96	1010	3	48	35	5	<1	<1	<1	14
BH-32SL	08-12-96	1516	<2	<2	170	9	2	<2	<2	2
BH-33I	08-14-96	1510	12	48	120	4	2	<1	<1	11
BH-33S	08-14-96	1345	15	23	80	6	<1	<1	<1	28
BH-33SL	08-14-96	1215	<2	<2	160	13	4	<2	<2	3
BH-34ND	08-15-96	1220	<1	2	160	2	2	<1	<1	5
BH-34SS	08-13-96	1522	<2	<2	200	13	3	--	<2	2
Lake George	08-13-96	1620	2	5	120	2	<1	<1	<1	3

Table 2. Results of analyses of ground water and surface water from Bairstow Landfill, Hammond, Ind., August 1996—Continued

Well name	Date sampled	Time sampled	Hardness, total (mg/L as $CaCO_3$) (00900)	Solids, residue at 180°C, total[1] (mg/L) (70300)	Solids, sum of constituents, dissolved (mg/L) (70301)	Analytical charge imbalance (percent)
BH-31	08-15-96	1010	--	848	--	+4.97
BH-32D	08-13-96	1255	390	527	519	-2.88
BH-32I	08-13-96	1117	320	506	500	-1.82
BH-32S	08-12-96	1645	65	634	555	--
BH-32S	08-13-96	940	71	588	565	-1.41
BH-32S	08-13-96	1010	68	624	560	--
BH-32SL	08-12-96	1516	600	1,240	1,270	-15.9
BH-33I	08-14-96	1510	56	680	629	-5.75
BH-33S	08-14-96	1345	28	864	785	-3.66
BH-33SL	08-14-96	1215	--	1,230	--	-10.4
BH-34ND	08-15-96	1220	47	581	551	-6.50
BH-34SS	08-13-96	1522	--	1,060	--	-15.4
Lake George	08-13-96	1620	87	433	423	-4.31

[1] Denotes water-quality property used in WATEQFP mineral-saturation calculations.

[2] Redox-potential value reported relative to the calomel electrode.

Table 3. Geochemical analyses of solid-phase samples from Bairstow Landfill, Hammond, Ind.

[percent, percent of total sample mass; ppm, parts per million; <, less than; µm, micrometers]

Sample identifier	Lithology	Aluminum (percent)	Antimony (ppm)	Arsenic (ppm)	Barium (ppm)	Beryllium (ppm)	Bismuth (ppm)	Cadmium (ppm)
32A	Calumet aquifer	4.22	<5	5	267	2	<5	<0.4
32B	Calumet aquifer	4.66	<5	<5	291	2	<5	<.4
33D	Calumet aquifer	5.25	<5	6	302	2	<5	<.4
33C	Slag-Calumet aquifer interface	4.20	<5	8	317	2	<5	.7
33A	Slag	2.47	<5	<5	324	3	<5	1.9
33B	Slag	2.56	<5	<5	265	2	<5	.9
32C <2µm	Lake Border sequence lacustrine sediment	9.99	<5	<5	451	3	<5	<.4
32C 2–5µm	Lake Border sequence lacustrine sediment	8.22	7	<5	513	2	<5	<.4
32C 5–20µm	Lake Border sequence lacustrine sediment	7.19	<5	<5	463	2	<5	<.4
34A	Lakebed sediment	1.46	13	5	135	1	<5	3.3
34B	Lakebed sediment	2.53	7	<5	261	<1	<5	<.4
34C	Lakebed sediment	2.16	17	12	269	2	<5	3.8
34D	Lakebed sediment	3.81	<5	<5	260	1	<5	<.4
34E	Lakebed sediment	4.87	<5	<5	274	2	<5	<.4
34E replicate	Lakebed sediment	4.64	<5	<5	264	1	<5	<.4
Dune sand	Dune sand	1.94	<5	<5	325	<1	<5	<.4

Table 3. Geochemical analyses of solid-phase samples from Bairstow Landfill, Hammond, Ind.—Continued

Sample identifier	Lithology	Calcium (percent)	Chromium (ppm)	Cobalt (ppm)	Copper (ppm)	Gold (ppm)	Iron (percent)	Lanthium (ppm)
32A	Calumet aquifer	11.05	1,268	19	65	<4	4.87	22
32B	Calumet aquifer	10.39	929	16	74	<4	4.41	23
33D	Calumet aquifer	10.71	1,403	23	81	<4	4.97	26
33C	Slag-Calumet aquifer interface	12.99	892	13	139	<4	5.72	22
33A	Slag	16.98	2,789	10	117	<4	11.4	22
33B	Slag	14.28	2,106	5	105	<4	10.0	19
32C <2µm	Lake Border sequence lacustrine sediment	4.49	111	17	40	<4	4.70	38
32C 2–5µm	Lake Border sequence lacustrine sediment	6.81	123	10	54	<4	3.78	38
32C 5–20µm	Lake Border sequence lacustrine sediment	6.22	78	9	85	<4	3.18	29
34A	Lakebed sediment	21.62	347	5	255	<4	4.21	14
34B	Lakebed sediment	11.65	176	7	26	<4	1.42	9
34C	Lakebed sediment	16.38	877	12	214	<4	5.89	18
34D	Lakebed sediment	11.42	1,374	26	278	<4	4.09	19
34E	Lakebed sediment	10.46	305	13	258	<4	3.36	23
34E replicate	Lakebed sediment	9.99	299	13	252	<4	3.23	21
Dune sand	Dune sand	.75	468	2	11	<4	1.31	8

Table 3. Geochemical analyses of solid-phase samples from Bairstow Landfill, Hammond, Ind.—Continued

Sample identifier	Lithology	Lead (ppm)	Manganese (ppm)	Molybdenum (ppm)	Nickel (ppm)	Niobium (ppm)	Phosphorous (percent)
32A	Calumet aquifer	58	5,121	13	467	6	0.133
32B	Calumet aquifer	57	4,319	12	335	6	.084
33D	Calumet aquifer	55	3,643	15	531	6	.053
33C	Slag-Calumet aquifer interface	133	6,919	12	208	8	.096
33A	Slag	116	25,109	20	295	18	.294
33B	Slag	117	23,754	11	165	18	.212
32C <2µm	Lake Border sequence lacustrine sediment	26	492	3	56	4	.051
32C 2–5µm	Lake Border sequence lacustrine sediment	35	808	3	51	9	.048
32C 5–20µm	Lake Border sequence lacustrine sediment	33	465	3	42	7	.031
34A	Lakebed sediment	402	4,990	12	87	4	.059
34B	Lakebed sediment	16	597	3	77	2	.023
34C	Lakebed sediment	587	10,042	31	250	6	.083
34D	Lakebed sediment	147	2,147	19	562	4	.035
34E	Lakebed sediment	68	771	8	128	4	.033
34E replicate	Lakebed sediment	67	731	7	125	4	.032
Dune sand	Dune sand	8	193	<2	12	2	.0107

Table 3. Geochemical analyses of solid-phase samples from Bairstow Landfill, Hammond, Ind.—Continued

Sample identifier	Lithology	Potassium (percent)	Scandium (ppm)	Silver (ppm)	Sodium (percent)	Strontium (ppm)	Thorium (ppm)	Tin (ppm)
32A	Calumet aquifer	1.90	9	0.6	0.29	107	6	6
32B	Calumet aquifer	2.03	9	<.5	.42	119	6	5
33D	Calumet aquifer	2.17	10	.5	.38	111	7	12
33C	Slag-Calumet aquifer interface	1.63	9	1.2	.39	176	6	26
33A	Slag	.21	6	1.3	.10	151	6	13
33B	Slag	.33	6	.8	.14	180	5	11
32C <2µm	Lake Border sequence lacustrine sediment	3.63	18	1.0	.72	210	14	4
32C 2–5µm	Lake Border sequence lacustrine sediment	3.30	14	1.1	.49	168	11	19
32C 5–20µm	Lake Border sequence lacustrine sediment	3.01	12	<.5	.47	136	6	14
34A	Lakebed sediment	.41	3	.5	.21	663	2	34
34B	Lakebed sediment	1.33	4	<.5	.63	123	2	<2
34C	Lakebed sediment	.63	5	.8	.20	356	4	42
34D	Lakebed sediment	1.92	8	<.5	.38	108	5	15
34E	Lakebed sediment	2.51	9	.6	.45	204	7	14
34E replicate	Lakebed sediment	2.43	9	<.5	.44	197	6	13
Dune sand	Dune sand	1.26	3	<5	.50	89	<2	<2

Table 3. Geochemical analyses of solid-phase samples from Bairstow Landfill, Hammond, Ind.—Continued

Sample identifier	Lithology	Titanium (percent)	Tungsten (ppm)	Uranium (ppm)	Vanadium (ppm)	Yttrium (ppm)	Zinc (ppm)	Zirconium (ppm)
32A	Calumet aquifer	0.24	7	<10	158	18	152	47
32B	Calumet aquifer	.26	16	10	142	18	148	52
33D	Calumet aquifer	.26	6	<10	119	21	170	53
33C	Slag-Calumet aquifer interface	.24	32	<10	166	21	266	44
33A	Slag	.25	17	<10	336	25	526	38
33B	Slag	.30	13	<10	304	21	486	48
32C <2µm	Lake Border sequence lacustrine sediment	.34	8	19	163	26	101	83
32C 2–5µm	Lake Border sequence lacustrine sediment	.45	4	18	129	23	95	64
32C 5–20µm	Lake Border sequence lacustrine sediment	.36	<4	<10	103	17	78	55
34A	Lakebed sediment	.10	8	<10	79	11	889	23
34B	Lakebed sediment	.11	<4	15	30	9	52	22
34C	Lakebed sediment	.15	12	<10	129	16	1,046	31
34D	Lakebed sediment	.21	<4	<10	84	17	231	48
34E	Lakebed sediment	.24	<4	<10	84	18	149	55
34E replicate	Lakebed sediment	.23	<4	<10	80	17	147	51
Dune sand	Dune sand	.11	<4	<10	20	7	11	30

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.

[**Bold type**, positive saturation indices; Shaded rows, minerals present in X-ray diffraction analyses; --, not caluclated because constituent concentrations were less than laboratory reporting limits]

Mineral name	Well Name										
	BH-31	BH-32SL	BH-32S	BH-32I	BH-32D	BH33-SL	BH-33S	BH-33I	BH-34SS	BH-34ND	Lake George
Adularia $KAlSi_3O_8$	-6.13	-5.62	**2.31**	**1.23**	**0.21**	-6.13	**2.55**	**1.83**	-6.25	**2.00**	**1.45**
Albite $Na(Si_3Al)O_8$	-8.78	-7.97	**.76**	-.10	-1.18	-8.44	**.48**	**.27**	-8.61	-.28	-3.48
Allophane (a) $[Al(OH)_3]_{(1-x)}[SiO_2]_x$	-3.17	-2.37	-.76	-.05	-.31	-3.72	-1.38	-1.09	-3.75	-3.22	-1.69
Allophane (P) $[Al(OH)_3]_{(1-x)}[SiO_2]_x$	-1.52	-.70	**.43**	**.83**	**.58**	-2.12	-.10	**.05**	-2.13	-1.93	-.63
$Al(OH)_3$	-5.13	-4.60	-2.54	-1.70	-1.94	-5.11	-2.97	-2.77	-5.01	-4.15	-2.54
Analcime $NaAlSi_2O_6 \cdot H_2O$	-7.76	-6.80	-1.30	-2.44	-3.26	-7.46	-1.41	-1.80	-7.54	-2.35	-4.37
Anhydrite $CaSO_4$	-1.79	-2.12	-3.72	-2.93	-2.01	-1.79	-3.73	-2.97	-2.08	-2.72	-2.24
Anorthite $CaAl_2Si_2O_8$	-6.30	-5.67	-1.40	-2.38	-3.14	-6.62	-1.98	-2.53	-6.23	-3.04	-4.08
Aragonite $CaCO_3$	**2.46**	**2.80**	**1.40**	**.18**	**.28**	**2.84**	**1.36**	**1.02**	**2.80**	**1.36**	**.50**
Artinite $Mg(CO_3)(OH)_2 \cdot 3(H_2O)$	**5.40**	**5.47**	-2.12	-6.55	-6.22	**5.20**	-1.58	-3.23	**5.48**	-.80	-3.69

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	BH-32SL	BH-32S	BH-32I	BH-32D	BH33-SL	BH-33S	BH-33I	BH-34SS	BH-34ND	Lake George
$Ba_3(AsO_4)_2$	--	--	**13.8**	**14.8**	**10.8**	--	**12.1**	**10.9**	**11.4**	--	**8.36**
Barite $BaSO_4$	**.58**	**.74**	-.99	-.31	**.59**	**.59**	-.91	-.26	-.41	-.28	-.02
Beidellite $(Ca, Na)_{0.3}Al_2$ $(OH)_2$-$(Al, Si)_4$ $O_{10}\cdot(H_2O)_4$	-14.4	-13.3	**2.21**	**4.60**	**3.13**	-14.2	**.72**	**1.62**	-14.2	-1.29	-2.20
Bixbyite γ-AlO·OH	--	--	-13.6	-18.0	-18.2	--	-13.1	-13.7	--	-9.08	-4.52
Boehmite γ-AlO(OH)	-2.96	-2.43	-.37	**.47**	**.23**	-2.91	-.79	-.597	-2.81	-1.94	-.31
Brucite $Mg(OH)_2$	--	-.37	-2.83	-5.92	-5.53	--	-3.48	-3.53	--	-1.56	-3.37
Bunsenite NiO	-2.18	-2.34	-5.56	-4.15	-7.14	-1.74	-5.21	-5.75	-1.55	-4.59	-5.24
Calcite $CaCO_3$	**2.62**	**2.95**	**1.55**	**.33**	**.43**	**2.99**	**1.51**	**1.17**	**2.95**	**1.50**	**.64**
Cerrusite $PbCO_3$	--	--	-1.83	--	--	--	-2.14	-1.56	--	-1.54	-2.23
$CaSeO_3$	-4.73	-5.59	--	--	--	-5.16	--	-6.00	-5.28	-6.13	--
Chalcedony SiO_2	-2.84	-2.93	**.24**	**.51**	**.25**	-2.77	**.08**	**.26**	-2.84	**.31**	-.82

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	BH-32SL	BH-32S	BH-32I	BH-32D	BH33-SL	BH-33S	BH-33I	BH-34SS	BH-34ND	Lake George
Chlorite 14A $Mg_5Al(Si_3Al)O_{10}(OH)_8$	--	**7.51**	**8.83**	-4.11	-3.43	--	**4.14**	**4.84**	--	**11.6**	**2.14**
Chlorite 7A $Mg_5Al(Si_3Al)O_{10}(OH)_8$	--	**4.04**	**5.35**	-7.59	-6.90	--	**.69**	**1.38**	--	**8.27**	-1.19
Chrysotile $Mg_3[Si_2O_5](OH)_4$	--	**4.81**	**3.76**	-4.96	-4.32	--	**1.40**	**1.62**	--	**7.23**	-.73
Clinoenstatite $(Mg,Fe^{+2})[SiO_3]$	--	-1.29	-.58	-3.40	-3.27	--	-1.40	-1.27	--	**.70**	-2.27
Clpyromorphite $Pb_5(PO_4)_3Cl$	--	--	-4.33	--	--	--	-6.68	-1.88	--	-4.88	--
Cristobalite SiO_2	-2.78	-2.87	**.29**	**.56**	**.31**	-2.72	**.13**	**.31**	-2.80	**.34**	-.79
Copper Cu (metal)	--	--	**1.67**	--	--	-3.15	**1.33**	**.99**	--	--	-9.34
$Cu(OH)_2$	--	--	-2.74	--	--	-2.50	-3.12	-2.73	--	--	-2.38
Cupric Ferrite $CuFe_2O_4$	--	--	**13.6**	--	--	**8.44**	**13.8**	**13.8**	--	--	**13.2**
Cuprite Cu_2O	--	--	**3.15**	--	--	-1.46	**2.41**	**2.47**	--	--	-7.59

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
Cuprous Ferrite $CuFeO_2$	--	--	**16.6**	--	--	**11.5**	**16.5**	**16.4**	--	--	**10.7**
Diaspore α-AlO(OH)	-1.16	-.63	**1.42**	**2.27**	**2.02**	-1.16	**.98**	**1.18**	-1.07	-.22	**1.36**
Diopside $Ca(Mg,Fe)[Si_2O_6]$	--	**4.52**	**2.21**	-3.54	-3.44	--	**1.76**	**.77**	--	**4.71**	-1.53
Dioptase $Cu_6(SiO_{18})\cdot 6H_2O$	--	--	-3.87	--	--	-6.66	-4.42	-3.84	--	--	-4.63
Dolomite (d) $CaMg(CO_3)_2$	--	**1.16**	**2.10**	-.21	**.15**	--	**.84**	**1.41**	--	**2.08**	**.63**
Dolomite (c) $CaMg(CO_3)_2$	--	**1.76**	**2.69**	**.37**	**.74**	--	**1.43**	**2.00**	--	**2.64**	**1.16**
Epsomite $MgSO_4\cdot 7H_2O$	--	-8.36	-6.23	-5.32	-4.24	--	-7.48	-5.46	--	-5.41	-4.79
FCO_3 Apatite $Ca_{9.49}Na_{.36}Mg_{.14}(PO_4)_{4.8}(CO_3)_{1.2}F_{2.48}$	--	**21.9**	**22.7**	**12.2**	**12.7**	--	**19.4**	**20.0**	--	**15.9**	--
$Fe_3(OH)_8$	-2.75	-6.79	**6.24**	**.05**	**.35**	-4.94	**6.98**	**6.00**	-5.36	-.96	-1.70
$Fe(OH)_{2.7}Cl_{0.3}$	**3.34**	**1.88**	**6.43**	**4.96**	**4.95**	**2.80**	**6.61**	**6.68**	**2.74**	**4.71**	**5.99**

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
Fluorapatite $Ca_5(PO_4)_3F$	--	**4.36**	**5.53**	**0.09**	**0.38**	--	**3.97**	**4.27**	--	**2.32**	--
Fluorite CaF_2	-1.56	-1.71	-1.60	-1.89	-1.92	-1.31	-2.41	-1.76	-1.69	-2.28	-1.76
Forsterite $MgSiO_4$	--	-1.82	-3.58	-9.49	-8.96	--	-5.03	-4.96	--	-.98	-5.75
Gibbsite (c) $Al(OH)_3$	-2.35	-1.81	**.24**	**1.08**	**.84**	-2.37	-.20	.00	-2.29	-1.45	**.11**
Goethite α-FeO·OH	**5.91**	**4.40**	**8.15**	**6.11**	**6.15**	**5.23**	**8.38**	**8.17**	**5.22**	**6.50**	**7.33**
Greenalite $(Fe,Mg)_3Si_2O_5(OH)_4$	-10.9	-14.1	**8.82**	**3.03**	**3.13**	-13.1	**9.39**	**8.02**	-14.4	-2.57	-11.9
Gypsum $CaSO_4 \cdot 2H_2O$	-1.54	-1.87	-3.47	-2.68	-1.76	-1.56	-3.49	-2.72	-1.85	-2.49	-2.04
Halloysite $Al_2O_3.2SiO_2 \cdot 4H_2O$	-13.8	-12.9	-2.51	-.27	-1.28	-13.7	-3.70	-2.93	-13.6	-5.67	-4.77
Hematite $Fe(OH)_3$	**13.0**	**9.99**	**17.4**	**13.4**	**13.5**	**12.0**	**18.1**	**17.6**	**12.2**	**14.8**	**16.9**
Huntite $Mg_3 Ca(CO_3)_4$	--	-4.99	**.61**	-3.91	-3.01	--	-3.10	-.71	--	**.55**	-2.10

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
Hydrocerrusite $Pb_3(CO_3)_2(OH)_2$	--	--	-5.83	-2.25	--	--	-6.21	-5.43	--	-3.79	-7.12
Hydroxyapatite $Ca_5(PO_4)_3$ (OH,F,Cl)	--	**6.63**	**4.65**	--	-1.83	--	**3.78**	**3.10**	--	**2.46**	--
Illite $K_{1-1.5}Al_4[Si_{7-6.5}Al_{1-1.5}O_{20}]$ $(OH)_4$	--	-10.1	**2.89**	**3.37**	**2.00**	--	**1.84**	**2.05**	--	**.23**	-1.59
Kaolinite $Al_4[Si_4O_{10}]$ $(OH)_8$	-8.68	-7.79	**2.67**	**4.90**	**3.89**	-8.59	**1.45**	**2.23**	-8.59	-.59	**.24**
Laumontite $Ca[Al_2Si_4O_{12}]$ $\cdot 4H_2O$	-7.34	-6.90	**3.71**	**3.28**	**1.98**	-7.79	**2.72**	**2.56**	-7.64	**1.78**	-1.76
Leonhardite $CaAl_2Si_4O_{12}$ $\cdot 3.5H_2O$	-6.57	-5.67	**15.5**	**14.6**	**12.0**	-7.61	**13.5**	**13.2**	-7.37	**11.4**	**4.22**
Maghemite $\gamma\text{-}Fe^{3+}{}_2O_3$	**3.44**	**.43**	**7.91**	**3.83**	**3.92**	**2.08**	**8.37**	**7.97**	**2.06**	**4.61**	**6.27**
Magnesite $MgCO_3$	--	-1.73	**.59**	-.50	-.23	--	-.63	**.27**	--	**.55**	-.06
Magnetite $Fe^{2+}Fe^{3+}{}_2O_4$	**12.3**	**8.34**	**21.3**	**15.1**	**15.5**	**10.9**	**22.3**	**21.3**	**10.7**	**15.3**	**15.2**

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	BH-32SL	BH-32S	BH-32I	BH-32D	BH33-SL	BH-33S	BH-33I	BH-34SS	BH-34ND	Lake George
Malachite $Cu_2(OH)_2CO_3$	--	--	-4.62	--	--	-8.45	-5.95	-4.20	--	--	-3.80
Manganite $MnO \cdot OH$	--	--	-6.08	-8.27	-8.39	--	-5.98	-6.25	--	-4.55	-2.68
$MnHPO_4$	--	--	-.52	**.27**	**.10**	--	-1.40	-.14	--	-3.29	--
Montmorillonite BF $Al_2Si_4O_{10}(OH)_2 \cdot xH_2O$	--	-9.90	**5.00**	**4.97**	**3.64**	--	**4.05**	**4.60**	--	**3.63**	**1.28**
Montmorillonite AB $Al_2Si_4O_{10}(OH)_2 \cdot xH_2O$	--	-9.42	**4.93**	**4.06**	**2.84**	--	**3.93**	**4.41**	--	**3.59**	**.91**
Montmorillonite CA $Al_2Si_4O_{10}(OH)_2 \cdot xH_2O$	-14.4	-13.5	**1.89**	**4.33**	**2.84**	-14.2	**.36**	**1.29**	-14.3	-1.56	-2.43
Muscovite $K_2Al_4[Si_6Al_2O_{20}](OH,F)_4$	-5.28	-3.70	**8.35**	**8.96**	**7.44**	-5.30	**7.70**	**7.38**	-5.25	**4.70**	**4.40**
Nahcolite $NaHCO_3$	-6.78	-5.65	-3.69	-4.23	-4.41	-5.57	-3.62	-3.61	-5.92	-4.45	-4.53
Nantokite $CuCl$	--	--	-4.78	--	--	-9.58	-5.34	-4.39	--	--	-9.12

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
Natron $Na_2CO_3\cdot 10H_2O$	-6.61	-4.94	-5.82	-8.89	-9.15	-5.48	-5.20	-6.13	-5.95	-6.51	-8.14
Nesquehonite $MgCO3\cdot 3H_2O$	--	-4.13	-1.79	-2.89	-2.63	--	-3.03	-2.12	--	-1.85	-2.47
$Ni(OH)_2$	**.92**	**.75**	-2.46	-1.04	-4.10	**.58**	-2.39	-2.86	**.50**	-2.74	-4.05
Ni_2SiO_4	-.14	-.55	-3.81	-.72	-6.96	**.67**	-3.31	-4.20	**.93**	-2.03	-4.59
$Pb(OH)_2$ (c)		--	-1.62	--	--	--	-1.36	-1.74		-.06	-1.97
$PbSiO_3$		--	-4.10	--	--	--	-3.99	-4.20		-2.47	-5.52
Phillipsite $(0.5Ca,Na,K)_3$ $[Al_3Si_5O_{16}]$ $\cdot 6H_2O$	-7.63	-6.97	**1.36**	**.39**	-.62	-7.05	**1.49**	**.99**	-7.06	**1.34**	-1.64
Portlandite $Ca(OH)_2$	-2.05	-2.31	-8.51	-11.7	-11.47	-2.56	-7.91	-9.22	-2.21	-7.05	-9.01
Prehnite $Ca_2Al[AlSi_3O_{10}]$ $(OH)_2$	**.70**	**.98**	**2.22**	-1.69	-2.49	-.41	**1.95**	**.30**	**.11**	**1.53**	-2.91
Pyrochroite $Mn(OH)_2$	--	--	-4.21	-6.46	-6.41	--	-4.00	-4.66	--	-4.72	-5.95
Pyrolusite MnO_2	--	--	-15.6	-17.7	-17.9	--	-15.2	-15.2	--	-10.5	-4.75

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
Pyrophyllite $Al_4[Si_8O_{20}](OH)_4SiO_2$	**12.6**	-11.9	**4.89**	**7.67**	**6.22**	-11.5	**3.67**	**4.73**	-11.4	**3.12**	**2.40**
Quartz SiO_2	-2.38	-2.47	**.70**	**.97**	**.71**	-2.32	**.54**	**.72**	-2.40	**.74**	-.41
Rhodochrosite (d) $MnCO_3$	--	--	-.61	-.87	-.93	--	-.89	-.61	--	-1.96	-1.74
Rhodochrosite (c) $MnCO_3$	--	--	**.08**	-.17	-.23	--	-.18	**.09**	--	-1.22	-.99
Sepiolite (d) $Mg_4(OH)_2Si_6O_{15}\cdot H_2O+4H_2O$	--	-4.10	**.48**	-4.87	-4.91	--	-1.48	-1.00	--	**2.32**	-5.20
Sepiolite (c) $Mg_4(OH)_2Si_6O_{15}\cdot H_2O+4H_2O$	--	-1.49	**3.10**	-2.26	-2.28	--	**1.18**	**1.65**	--	**5.18**	-2.21
Siderite (d) $FeCO_3$	-9.29	-9.73	**.62**	**.50**	**.60**	-8.83	**.42**	**.79**	-9.46	-4.12	-5.07
Siderite (c) $FeCO_3$	-8.92	-9.35	**.99**	**.88**	**.97**	-8.42	**.81**	**1.17**	-9.04	-3.69	-4.61
Silica gel	-3.38	-3.47	-.30	-.02	-.28	-3.30	-.45	-.27	-3.38	-.22	-1.35
SiO_2 (a)	-3.72	-3.81	-.63	-.36	-.62	-3.62	-.78	-.60	-3.69	-.53	-1.65
Smithsonite $ZnCO_3$	-9.06	-8.62	-3.44	-3.26	-3.21	-7.24	-3.66	-3.21	-7.72	-3.99	-3.04

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	BH-32SL	BH-32S	BH-32I	BH-32D	BH33-SL	BH-33S	BH-33I	BH-34SS	BH-34ND	Lake George
Strengite $Fe^{3+}(PO_4)\cdot 2H_2O$	--	-16.8	-3.40	-2.39	-2.55	--	-4.22	-2.52	--	-7.18	--
Talc $Mg_6[Si_8O_{20}]$ $(OH)_4$	--	**2.49**	**7.80**	-.38	-.25	--	**5.15**	**5.72**	--	**11.5**	**1.35**
Tenorite CuO	--	--	-1.72	--	--	-1.48	-2.10	-1.71	--	--	-1.36
Tremolite $Ca_2Mg_5Si_8O_{22}$ $(OH)_4$	--	**16.5**	**17.2**	-2.48	-2.16	--	**13.6**	**12.2**	--	**25.6**	**2.77**
Vivianite $Fe_3(PO_4)_2\cdot 8H_2O$	--	-37.3	-1.30	-1.52	-1.34	--	-2.58	-.48	--	-17.3	--
Wairakite $CaAl_2Si_4O_{12}$ $\cdot 2H_2O$	-11.9	-11.5	-.89	-1.32	-2.61	-12.2	-1.81	-1.99	-11.9	-2.51	-5.89
Willemite $ZnSiO_4$	-5.85	-6.26	-2.32	-5.67	-5.59	-4.33	-1.80	-2.65	-4.74	-1.00	-2.77
Witherite $BaCO_3$	-.64	.17	-1.36	-2.69	-2.60	-.20	-1.28	-1.74	-.93	-1.60	-2.62
Zincite (c) ZnO	-3.28	-3.44	-3.05	-4.86	-4.69	-2.51	-2.70	-3.22	-2.67	-2.36	-2.65
$Zn(OH)_2$ (e)	-3.05	-3.21	-2.83	-4.64	-4.50	-2.60	-2.59	-3.08	-2.86	-2.64	-3.19
$Zn(OH)_2$ (c)	-3.75	-3.91	-3.53	-5.34	-5.20	-3.30	-3.29	-3.78	-3.56	-3.34	-3.89

Table 4. Mineral-saturation indices calculated for water samples from Bairstow Landfill, Hammond, Ind.—Continued

Mineral name	Well Name										
	BH-31	**BH-32SL**	**BH-32S**	**BH-32I**	**BH-32D**	**BH33-SL**	**BH-33S**	**BH-33I**	**BH-34SS**	**BH-34ND**	**Lake George**
$Zn(OH)_2$ (g)	-3.26	-3.42	-3.04	-4.85	-4.71	-2.81	-2.80	-3.29	-3.07	-2.85	-3.40
$Zn(OH)_2$ (b)	-3.30	-3.46	-3.08	-4.89	-4.75	-2.85	-2.84	-3.33	-3.11	-2.89	-3.44
$Zn(OH)_2$ (a)	-4.00	-4.16	-3.78	-5.59	-5.45	-3.55	-3.54	-4.03	-3.81	-3.59	-4.14
$ZnCO_3{\cdot}H_2O$	-8.68	-8.25	-3.06	-2.88	-2.84	-6.92	-3.31	-2.85	-7.43	-3.71	-2.81
ZnO (a)	-2.86	-3.02	-2.64	-4.45	-4.31	-2.41	-2.40	-2.89	-2.67	-2.45	-3.00
$ZnSiO_3$	-1.49	-1.74	**1.81**	**.27**	**.18**	-.64	**2.01**	**1.67**	-.86	**2.59**	1.19

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.

^{o}C, degrees Celsius; mv, millivolts; mg/L, milligrams per liter; µS/cm, microsiemens per centimeter at 25 degrees Celsius; $CaCO_3$, computed as calcium carbonate; µg/L, micrograms per liter, <, less than]

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Temperature (^{o}C)	Slag-contact	21	11.9	14.1	15.0	16.4	22.0
	Slag-affected	25	11.6	12.8	14.5	16.0	23.5
	Background	11	10.3	11.2	12.0	14.0	14.8
pH (standard units)	Slag-contact	21	6.60	7.50	8.70	11.5	12.3
	Slag-affected	19	6.90	7.30	7.50	8.00	9.90
	Background	11	6.00	6.80	7.30	7.40	7.70
Redox potential (mv)	Slag-contact	21	-446	-195	-64.0	84.5	134
	Slag-affected	25	-183	-134	-100	-1.00	136
	Background	11	-198	-163	-78.0	5.00	52.0
Dissolved oxygen (mg/L)	Slag-contact	19	.10	.30	.60	1.40	4.20
	Slag-affected	22	<.10	.10	.20	1.45	6.30
	Background	11	<.10	.10	.20	.30	6.10
Specific conductance (µS/cm)	Slag-contact	21	192	930	1,880	3,820	5,720
	Slag-affected	26	186	469	847	1,300	3,480
	Background	11	164	239	445	900	1,190
Alkalinity (mg/L as $CaCO_3$)	Slag-contact	20	24.9	153	279	661	884
	Slag-affected	25	74.6	177	207	399	532
	Background	11	41.8	95.5	163	216	279

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.—Continued

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Calcium (mg/L)	Slag-contact	21	10.4	121	189	245	445
	Slag-affected	25	9.18	53.7	83.0	132	514
	Background	11	10.7	27.0	54.2	60.0	301
Magnesium (mg/L)	Slag-contact	21	.01	.09	17.6	60.0	211
	Slag-affected	25	.12	6.71	22.8	35.6	68.4
	Background	11	5.03	8.63	14.4	26.4	50.6
Potassium (mg/L)	Slag-contact	21	3.12	12.3	25.1	68.6	170
	Slag-affected	25	.88	4.26	6.72	18.1	102
	Background	11	.51	.59	1.18	2.65	26.2
Sodium (mg/L)	Slag-contact	21	22.7	59.1	125	220	1,450
	Slag-affected	25	2.97	18.3	84.4	180	465
	Background	11	1.11	2.18	11.3	74.1	129
Chloride (mg/L)	Slag-contact	20	12.0	29.0	69.0	628	2,600
	Slag-affected	25	5.20	20.5	54.0	142	674
	Background	11	1.40	6.20	12.2	134	139
Sulfate (mg/L)	Slag-contact	20	8.0	74.5	190	382	1,160
	Slag-affected	25	5.0	13.8	72.0	175	1,570
	Background	11	5.0	5.00	27.3	67.0	660

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.—Continued

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Silica (mg/L)	Slag-contact	20	3.10	10.0	17.0	28.7	35.3
	Slag-affected	25	9.60	17.5	25.3	32.6	74.0
	Background	11	10.5	12.2	12.6	13.4	29.7
Ammonia (mg/L)	Slag-contact	20	.10	1.02	2.40	8.80	44.3
	Slag-affected	24	.10	.10	1.30	3.38	90.3
	Background	11	.10	.30	.32	.90	2.60
Aluminum (µg/L)	Slag-contact	21	20.9	23.5	49.7	251	820
	Slag-affected	25	6.00	20.9	23.5	36.7	160
	Background	11	20.9	20.9	23.5	74.9	383
Antimony (µg/L)	Slag-contact	21	<2.00	<15.7	<15.7	<17.9	<17.9
	Slag-affected	25	<1.00	<8.35	<15.7	<17.9	<17.9
	Background	11	<15.7	<15.7	<17.9	<17.9	<17.9
Arsenic (µg/L)	Slag-contact	21	<1.00	<1.70	2.70	5.00	22.6
	Slag-affected	25	<1.70	<1.70	3.60	18.8	55.0
	Background	11	<1.70	<1.70	1.90	3.30	12.8
Barium (µg/L)	Slag-contact	21	30.2	56.3	128	284	690
	Slag-affected	25	10.7	25.0	58.0	134	410
	Background	11	14.7	16.5	31.1	83.0	229

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.—Continued

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Beryllium (µg/L)	Slag-contact	21	<.50	<.50	<1.20	<1.20	2.00
	Slag-affected	25	<.50	<.50	<1.00	<1.20	<1.20
	Background	11	<.50	<.50	<.50	<1.20	<1.20
Cadmium (µg/L)	Slag-contact	21	<1.50	<1.50	<1.50	<1.70	2.00
	Slag-affected	25	<1.00	<1.25	<1.50	<1.70	<1.70
	Background	11	<1.50	<1.50	<1.70	<1.70	<1.70
Chromium (µg/L)	Slag-contact	21	<2.00	<5.10	<5.10	<5.80	9.20
	Slag-affected	25	<1.00	4.55	<5.10	<5.80	<5.80
	Background	11	<5.10	<5.10	<5.80	<5.80	<5.80
Cobalt (µg/L)	Slag-contact	21	<2.00	<2.50	<2.50	<3.70	19.7
	Slag-affected	25	<1.00	1.75	<2.50	<3.70	6.10
	Background	11	<2.50	<2.50	<3.70	<3.70	<3.70
Copper (µg/L)	Slag-contact	21	<2.00	<4.00	<4.00	<4.20	1,660
	Slag-affected	25	<1.00	3.50	<4.20	<4.20	11.5
	Background	11	<4.00	<4.00	<4.20	<4.20	7.50
Iron (µg/L)	Slag-contact	21	12.0	37.0	160	1,190	58,400
	Slag-affected	25	18.0	214	1,430	3,690	52,200
	Background	11	24.0	72.0	1,980	5,260	29,600

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.—Continued

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Lead (µg/L)	Slag-contact	21	<1.10	<1.10	1.90	2.10	10.8
	Slag-affected	25	<1.00	<1.10	<1.10	2.30	15.0
	Background	11	<1.10	<1.10	1.60	3.20	4.60
Manganese (µg/L)	Slag-contact	21	<2.00	<2.00	44.0	309	2,460
	Slag-affected	25	<1.80	19.8	99.0	473	1,540
	Background	11	<2.00	82.0	197	383	1,230
Mercury (µg/L)	Slag-contact	18	<.10	<.10	.17	.32	1.1
	Slag-affected	19	<.10	<.10	.11	.15	.18
	Background	11	<.10	<.10	.11	.14	.16
Nickel (µg/L)	Slag-contact	21	<4.70	<4.70	<6.10	11.0	148
	Slag-affected	25	2.00	<4.70	<6.10	<6.10	38.0
	Background	11	<4.70	<4.70	<6.10	<6.10	27.5
Selenium (µg/L)	Slag-contact	21	<2.00	<2.30	3.00	3.85	14.3
	Slag-affected	25	<1.00	2.15	<2.30	<2.30	17.2
	Background	11	<2.30	<2.30	<2.30	2.90	3.70
Silver (µg/L)	Slag-contact	21	<2.00	<3.80	<5.00	<5.00	5.00
	Slag-affected	25	<1.00	2.40	<3.80	<5.00	5.00
	Background	11	<3.80	<3.80	<3.80	<5.00	5.00

Table 6. Statistical summary of ground-water-quality data from Bairstow Landfill and from the Grand Calumet aquifer, Hammond, Ind.—Continued

Water-quality property	Well type	Number of samples	Minimum value	1st quartile	Median	3rd quartile	Maximum value
Thallium (µg/L)	Slag-contact	16	<1.10	<1.10	1.85	3.60	7.00
	Slag-affected	19	<1.10	1.30	1.90	2.30	2.80
	Background	10	<1.10	1.35	1.75	2.92	3.80
Vanadium (µg/L)	Slag-contact	18	<2.60	<2.60	3.35	8.15	52.7
	Slag-affected	19	<2.60	<2.60	<3.30	<3.30	9.10
	Background	11	<2.60	<2.60	<3.30	<3.30	5.00
Zinc (µg/L)	Slag-contact	21	2.00	<3.40	<3.70	4.40	531
	Slag-affected	25	3.00	<3.40	<3.70	5.70	28.0
	Background	11	<3.40	<3.70	4.10	411	1,090
Dissolved solids (mg/L)	Slag-contact	18	270	401	742	1,890	5,030
	Slag-affected	26	172	372	624	1,060	2,150
	Background	11	95.0	158	262	622	5,080
Suspended solids (mg/L)	Slag-contact	17	3.00	29.0	54.0	231	5,310
	Slag-affected	20	3.00	4.10	14.0	43.1	214
	Background	11	3.00	3.00	3.00	8.50	15.5
Total organic carbon (mg/L)	Slag-contact	17	3.7	6.80	15.1	48.5	209
	Slag-affected	20	2.00	4.17	5.60	13.4	24.7
	Background	11	2.00	2.90	4.80	11.6	26.9
Cyanide (µg/L)	Slag-contact	18	<10.0	<10.0	<10.0	16.6	212
	Slag-affected	19	<10.0	<10.0	<10.0	<10.0	31.4
	Background	11	<10.0	<10.0	<10.0	<10.0	<10.0