Geology, Hydrology, and Water Quality of the Glacial Drift Aquifer in the Vicinity of the Nelson Landfill near Yorkville, Illinois

By Robert T. Kay

In cooperation with the U.S. Environmental Protection Agency and
the United City of Yorkville, Illinois

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Conversion Factors, Datums, and Water-Quality Units

Inch/Pound to SI

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
foot (ft)	.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	.4047	hectare (ha)
square foot (ft^2)	.09290	square meter (m^2)
section (640 acres or 1 square mile)	259.0	square hectometer (hm^2)
	Volume	
gallon (gal)	3.785	liter (L)
	Flow rate	
foot per day (ft/d)	.3048	meter per day (m/d)
gallon per minute (gal/min)	3.785	liter per minute
gallon per day (gal/d)	3.785	liter per day
cubic foot per day (ft^3/d)	.02832	cubic meter per day (m^3/d)
	Pressure	
pound per square inch (lb/in^2)	6.895	kilopascal (kPa)
	Hydraulic conductivity	
foot per day (ft/d)	.3048	meter per day (m/d)
	Hydraulic gradient	
foot per foot (ft/ft)	.3048	meter per meter (m/m)
	Transmissivity*	
foot squared per day (ft^2/d)	.09290	meter squared per day (m^2/d)

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

$$°F=(1.8\times°C)+32$$

Vertical coordinate information for the site is referenced to an internal datum, which is a point 91.48 feet below the top of the inner casing in well NL3.

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

*Transmissivity: The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft3/d)/ft2]ft. In this report, the mathematically reduced form, foot squared per day (ft2/d), is used for convenience.

Specific conductance is given in microSiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

Oxidation-reduction potential is given in millivolts (mv).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

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Abstract

The geologic and hydrologic characteristics of the sand-and-gravel deposits that compose the glacial drift aquifer in the vicinity of the Nelson Landfill site in Yorkville, Illinois indicate that the aquifer could be developed as a source of public water supply. The geology of these deposits within the Newark Bedrock Valley is complex, however, and a detailed investigation of their water bearing and transmitting properties will be required to successfully locate high-capacity wells.

Volatile organic compounds, pesticides, and cyanide were not detected in ground water during this investigation. Metals and nitrogen compounds were not detected at concentrations above their Maximum Contaminant Level. Iron, manganese, and aluminum were detected at concentrations above their Secondary Maximum Contaminant Level and various constituents were detected at concentrations above background levels downgradient of the landfill. Nitrate and ammonia, presumably derived from agricultural practices, also were detected in samples from locations hydraulically upgradient of the landfill.

Oxidation-reduction conditions in the aquifer become more reducing with depth. This change is reflected by a change in the type of nitrogen compound detected and the concentration of dissolved oxygen and iron in the glacial drift aquifer. Concentrations of some of the major ions and metals may be affected by dissolution of carbonate minerals in the aquifer and perhaps road salts.

INTRODUCTION

The United City of Yorkville (Yorkville), as well as the remainder of Kendall County, in northeastern Illinois is experiencing a large increase in population. This increase in population is expected to continue (John Kos, Illinois Department of Transportation, written commun., 2004), with a consequent increase in demands for water. Water needs for Yorkville, and many of the other municipalities in Kendall County, are being supplied predominately or exclusively by ground water from the Cambrian-Ordovician aquifer system, a series of deep bedrock aquifers. Withdrawals from the Cambrian-Ordovician aquifer system in northeastern Illinois have exceeded recharge, resulting in water-level declines of hundreds of feet (Burch, 2002) and a degradation in water quality (Balding, 1991). These problems, as well as concentrations of naturally occurring radium in water from the Cambrian-Ordovician aquifer system greater than U.S. Environmental Protection Agency (USEPA) health-based standards, have resulted in a shift to alternate water supplies, particularly Lake Michigan and the various glacial drift aquifers, in parts of northeastern Illinois. Yorkville and other municipalities in Kendall County anticipate expanding the development of water supplies from the glacial drift aquifers in the County to meet the growing demand for water.

Sand-and-gravel deposits within the ancestral Newark Bedrock Valley in the northern part of Yorkville (and elsewhere in Kendall County) (fig. 1) have been identified as having a great potential for development of municipal-water supplies. Development of water supplies from the Newark Bedrock Valley is hindered by a lack of detailed information on the hydrogeology of the sand-and-gravel deposits within the valley that compose the aquifer. The impact of potential point sources (such as the Nelson Landfill)[1] (fig. 1) and non-point sources (such as fertilizer application and road salts) of contamination on water quality also are concerns for the development of drinking-water supplies from the aquifer (Kay and others, 2005). In addition, both Yorkville and the USEPA are considering redevelopment of the Nelson Landfill for reuse as a Brownfield (Anton Graff, United City of Yorkville, oral commun., 2004). To supplement the information needed to address these concerns and issues, the U.S. Geological Survey (USGS), in cooperation with United City of Yorkville and the USEPA conducted an investigation of the geology, hydrology, and water quality of the glacial drift aquifer in the New-

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[1] So named by U.S. Environmental Protection Agency (Ecology and Environment, Inc., 1986)

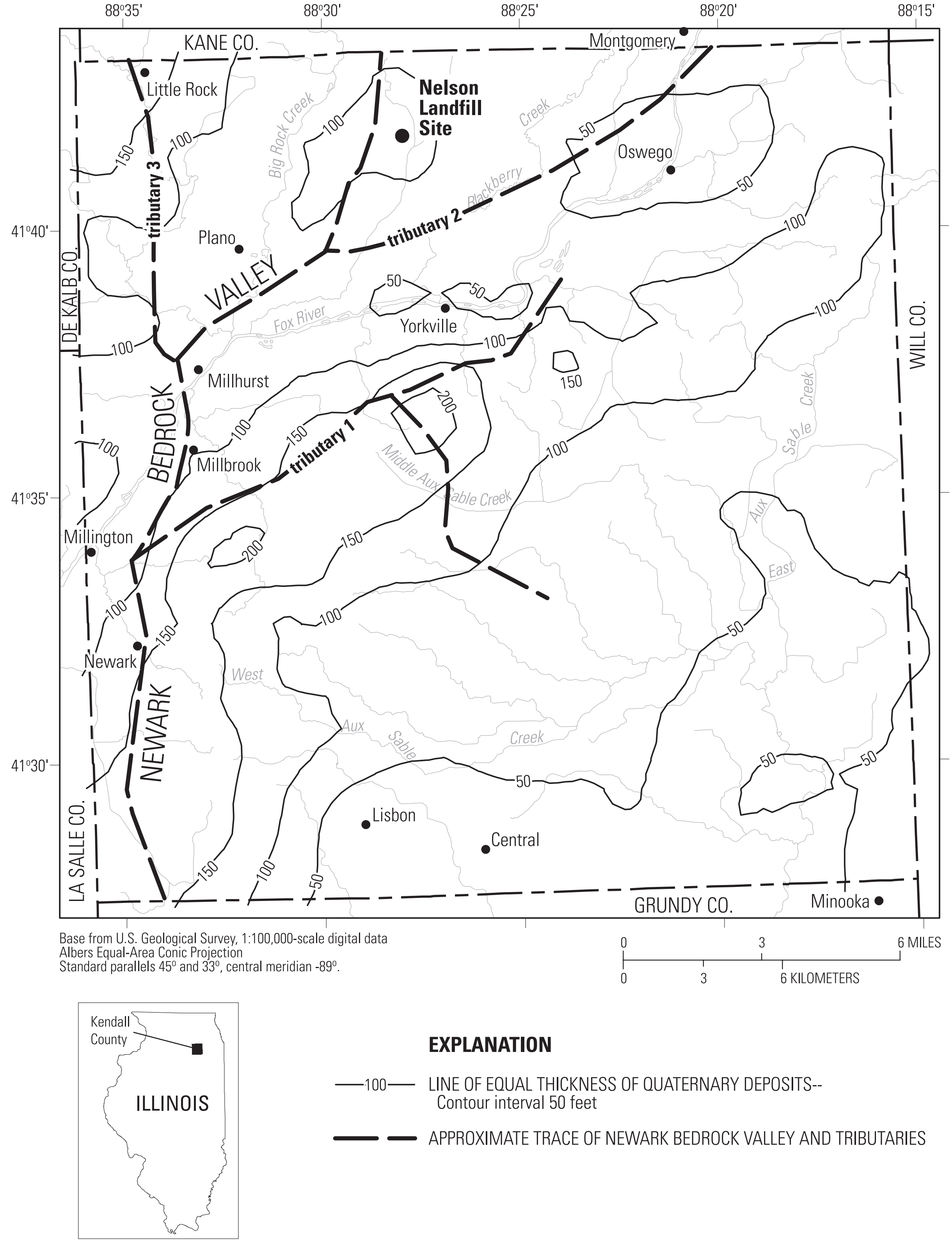


Figure 1. Thickness of glacial drift deposits and location of the Nelson Landfill site and the Newark Bedrock Valley in Kendall County, Illinois.

ark Bedrock Valley in the vicinity of the Nelson Landfill from February through July 2005.

The Nelson Landfill, hereafter referred to as the site, is a closed landfill located on East Beecher road (fig. 2), 3 mi north of the center of Yorkville. The site is located in section 8 of Township 37 north and Range 7 east. Approximately 20 acres of the site have been used as a landfill (fig. 3). Surface-water drainage at the site is to a tributary of Rob Roy Creek (hereafter referred to as the creek) (fig. 2), located approximately 500 ft east of the site, then south approximately 6 river miles to the Fox River. Currently (2006), land use surrounding the site is agricultural, but residences are located within 1,600 ft to the north and south. The agricultural areas surrounding the site are projected to be converted to residential land use within the next 10 years (Anton Graff, United City of Yorkville, oral commun., 2004). For the purposes of this report, the site consists of the area as defined in figure 3; the landfill is the area defined by the limits of the waste material; and the study area is defined by well NL1 to the north, the creek to the east, wells NL4 and GS4D to the south, and well NL2 to the west.

Purpose and Scope

This report describes the results of an investigation of the hydrogeology and water quality in the glacial drift aquifer in the study area surrounding the Nelson Landfill site. The results of water-level measurements, slug testing, and water-quality sampling from vertical aquifer profile locations and finished monitoring wells are presented. Additionally, the report describes the geology and hydrology of the glacial drift aquifer underlying the study area, identifies the directions and estimates the rate and volume of ground-water flow, describes ground-water quality upgradient and downgradient of the landfill, and identifies potential factors that affect the hydrology and water quality in the glacial drift deposits underlying the study area.

Site History

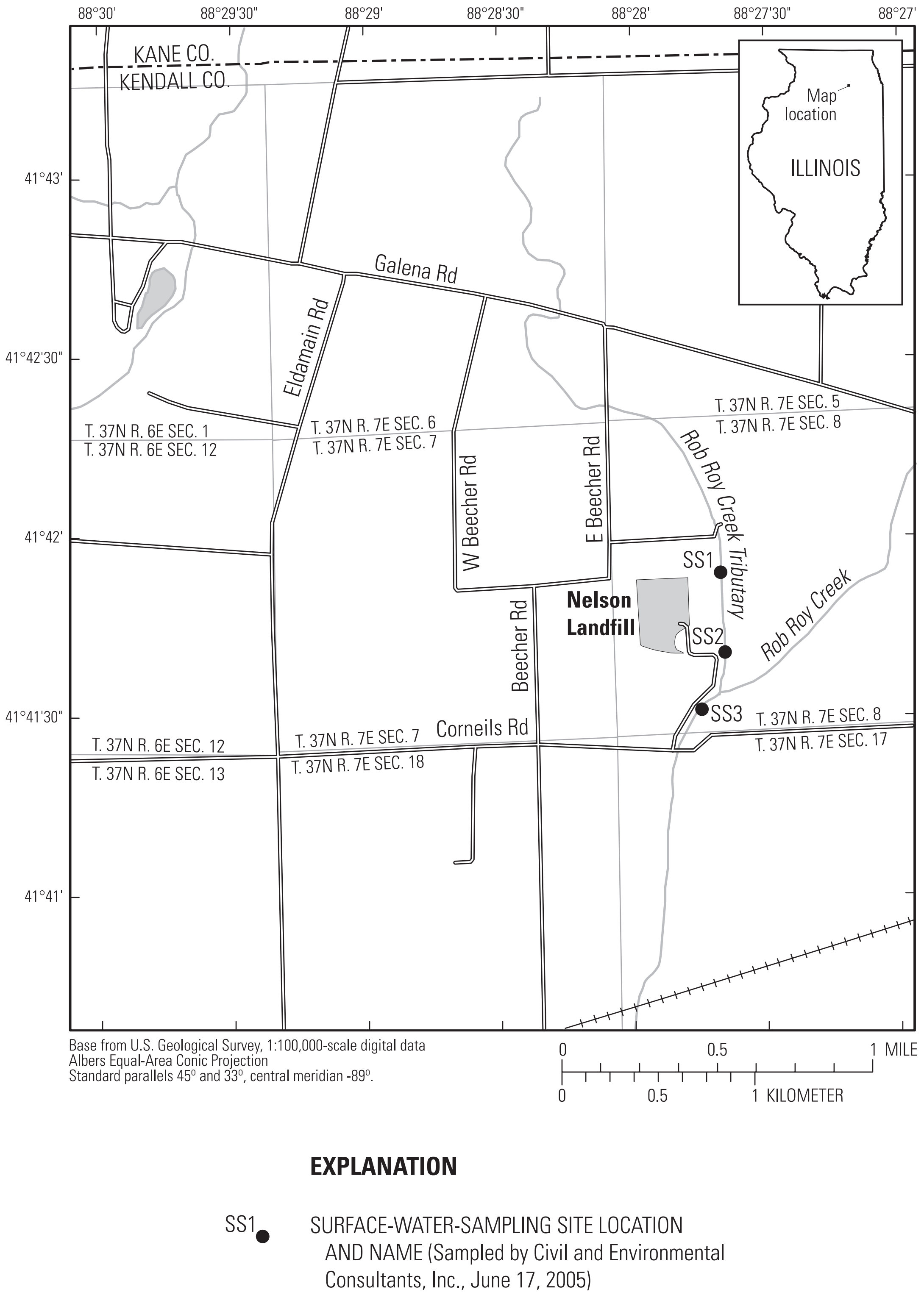
Sand and gravel were mined at the site prior to the beginning of landfill operations in 1967. The depth of excavation is unknown as is the volume of waste material in the landfill. Landfilled materials consist mostly of general refuse and construction debris. An Illinois Environmental Protection Agency (IEPA) inspection on March 17, 1971 indicated that approximately 8,000 gal of septic tank sludge, 500 gal of sludge from a packing plant, 500-750 gal of waste from an oil company, 500-1,000 gal of industrial waste consisting mainly of water and ink, and 1,000 gal of plating wastes were disposed of in the landfill every week (Ecology and Environment, Inc., 1987). Other wastes also were accepted, potentially including paint filter wastes, iron/steel foundry wastes, plating/polishing wastes, and laboratory/hospital wastes. Early reports by State inspectors note the disposal of oily wastes and sludge. The site was no longer authorized to accept septic or industrial waste in the early 1970s and was closed and covered in 1977 (Ecology and Environment, Inc., 1987).

In 1974, at IEPA request, synthetic liners and leachate collection systems were installed in disposal trenches, but it is unknown if the landfill area is completely lined. No complete leachate collection system is believed to be present. Polyethylene pipes were noted to have been installed in the trenches for leachate collection in January 1975, but the IEPA inspector concluded the pipes likely were being crushed by the weight of the fill and the liner was ineffective because of improper sealing and overlap (Illinois Environmental Protection Agency, written commun., 1975).

A sample of ponded leachate was collected at the landfill by the IEPA in May 1981 (Illinois Environmental Protection Agency, 1983). The sample contained 34 mg/L of ammonia, 1.6 mg/L of boron, 2,350 mg/L of chemical oxygen demand, 38 mg/L of oil and grease, 1.14 mg/L of phenolic compounds, 1,800 μg/L of toluene, and unidentified ketones and aliphatic alcohols at concentrations up to 7 mg/L. These results indicate that constituents in landfill leachate could present a threat to human health and the environment if released to ground water.

Seven monitoring wells (denoted by an NL prefix on figure 3 and table 1) were installed in the glacial drift deposits around the site as part of a series of environmental investigations begun in January 1985 (Ecology and Environment, Inc., 1986, 1987, 1988, 1991). These monitoring wells, and five residential-supply wells, were sampled by Ecology and Environment, Inc., in January 1985. Well-construction logs and interviews with homeowners indicate that the residential-supply wells north of the landfill draw water from glacial drift deposits, but that most or all of the residential-supply wells south of the landfill draw water from the underlying bedrock deposits.

Samples collected from monitoring well NL4 during January 1985 contained about 16 μg/L of 1,1-dichloroethene (Ecology and Environment, Inc., 1986). The pesticide aldrin was detected in a sample from monitoring well NL6A at a concentration of 0.2 μg/L. Vinyl chloride was detected in samples from monitoring wells NL6 and NL6A at concentrations of 2.1 and 3.0 μg/L, respectively. Concentrations of vinyl chloride exceeded the current USEPA Maximum Contaminant Level (MCL) for drinking water, which is 2 μg/L (U.S. Environmental Protection Agency, 2004). Calcium, magnesium, iron, barium, and manganese were detected in samples from monitoring and residential-supply wells located hydraulically downgradient of the landfill at concentra-



Base from U.S. Geological Survey, 1:100,000-scale digital data
Albers Equal-Area Conic Projection
Standard parallels 45° and 33°, central meridian -89°.

EXPLANATION

SS1 ● SURFACE-WATER-SAMPLING SITE LOCATION AND NAME (Sampled by Civil and Environmental Consultants, Inc., June 17, 2005)

Figure 2. Location of the Nelson Landfill site, Yorkville, Illinois.

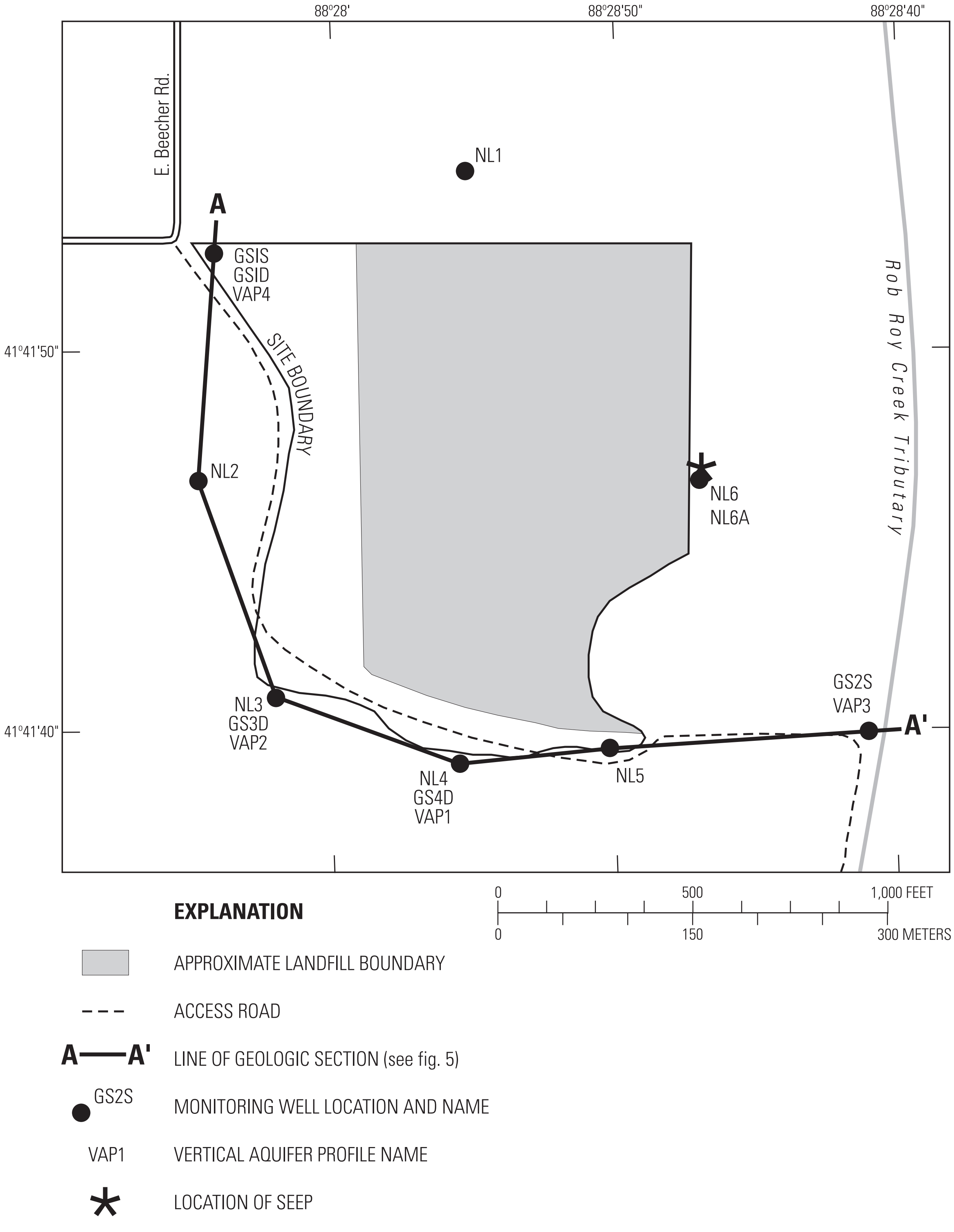


Figure 3. Ground-water sampling locations and line of geologic section in the vicinity of the Nelson Landfill site.

Table 1. Monitoring well data in the vicinity of the Nelson Landfill site, Yorkville, Illinois.

[Datum is 91.48 feet below top of inner casing in well NL3; bold denotes approximate value; nm, not measured]

Well name (locations shown in figure 3)	Elevation of measuring point (feet above site datum)	Elevation of land surface (feet above site datum)	Well depth (feet below land surface)	Water-level elevation (feet above site datum)					Horizontal hydraulic conductivity (feet per day)	Condition of well	Height of water column in well on 7/13/2005 (feet)
				2/28/2005	7/13/2005	7/27/2005	7/29/2005	8/15/2005			
NL1	92.15	**91**	29	77.32	76.53	76.30	nm	nm	nm	Partial obstruction	14.3
NL2	98.31	**96**	35	76.70	76.07	75.86	nm	nm	nm	Bent casing	14.9
NL3	91.48	89.4	29	74.55	73.87	73.68	73.68	73.48	0.34	In use	13.3
NL4	88.33	87.4	28	74.52	73.77	73.60	nm	73.38	65	In use	14.2
NL5	85.07	**83**	30	nm	nm	nm	nm	nm	nm	Destroyed	nm
NL6	nm	nm	28	nm	nm	nm	nm	nm	nm	Destroyed	nm
NL6A	nm	nm	13	nm	nm	nm	nm	nm	nm	Destroyed	nm
GS1S	108.82	106.0	34	nm	77.78	77.59	77.59	77.41	28	In use	5.6
GS1D	107.22	106.0	70	nm	74.55	74.36	74.36	74.17	160	In use	38.4
GS2S	81.81	79.9	32	nm	73.68	73.51	nm	73.21	640	In use	25.6
GS3D	90.70	88.6	75	nm	73.81	73.65	73.62	73.50	110	In use	60.1
GS4D	89.23	86.8	75	nm	73.80	73.57	nm	73.38	150	In use	61.8

tions higher than were detected in samples from monitoring wells located hydraulically upgradient of the landfill, but at concentrations below MCLs. Water levels measured in January 1985 indicated that ground-water flow was from north to south with a slight easterly component of flow.

Analysis of additional samples collected in August 1987 from five of the monitoring wells and five residential-supply wells identified the presence of 31 µg/L of chloroethane at well NL3, 3 µg/L of 1,1-dichloroethane at wells NL3 and NL4, and 2 µg/L of 1,2-dichloroethene at well NL4 (Ecology and Environment, Inc., 1991). Mercury was detected in the sample from well NL3 at a concentration of 4.1 µg/L, which is above its MCL of 2 µg/L. Arsenic was detected in the sample from well NL6 at a concentration of 2.8 µg/L µg/L. Cyanide was detected in the samples from wells NL4, NL5, and NL6 at concentrations of 22, 27, and 44 µg/L, respectively. Semivolatile organic compounds, pesticides, and polychlorinated biphenols (PCBs) detected at low concentrations (below MCLs) in some of the samples were attributed to field and laboratory contamination, not in-situ water quality. No constituent whose presence was attributed to the site was detected in a sample from any residential-supply well. Water levels measured at the time of sampling in 1987 also indicated ground-water flow in the study area was from north to south with a slight easterly component of flow.

Because of the low concentrations of chemicals in ground water at the monitoring wells and the absence of contaminants in samples from the nearby residential wells, the site was not investigated further. However, the detection of contaminants, in some instances at concentrations at or above MCLs, in addition to the desire by the United City of Yorkville to redevelop the site and to use the aquifer in the Newark Valley for public-water supply have resulted in a renewed interest in investigating hydrogeologic and water-quality conditions at and near the site.

STUDY METHODS

The investigation was conducted in three phases. Phase one consisted of redevelopment, water-level measurement, slug testing, and water-quality sampling of the accessible previously installed monitoring wells and water-quality sampling of a surface seep. These data were analyzed to guide the placement of additional monitoring wells at the site. Phase two consisted of vertical aquifer profiling for geology, hydraulic properties, and water quality at four locations in the study area. Phase three consisted of installation, development, water-level measurement, slug testing, and water-quality sampling of newly installed monitoring wells and a second round of water-quality sampling at the previously installed monitoring wells. All field activities, including lithologic and water-quality sampling, slug testing, well construction, well development, and equipment decontamination were performed in accordance with a USEPA-approved Work Plan, a Quality-Assurance Project Plan (QAPP), and a Field Sampling Plan, which were developed by the USGS for this investigation.

Previously installed monitoring wells NL5, NL6, and NL6A had been destroyed prior to the initiation of this investigation (table 1) and could not be used. The casing in wells NL1 and NL2 was bent, which limited the size of the devices (pumps, pressure transducers, water-level measurement tapes) that could be inserted in the wells. Only water levels could be collected from these wells.

Vertical aquifer profiling (VAP) was performed at four locations in the vicinity of the landfill (fig. 3) to characterize the hydrogeology and water quality with depth and to provide screening-level information that could be used to guide depth placement of the new monitoring wells. The locations of VAP1 and VAP2 were selected to characterize hydrogeology and water quality downgradient of the landfill in areas where historical sampling from wells open to the shallow part of the glacial drift aquifer (NL3 and NL4) indicated the presence of contamination. The location of VAP3 was chosen to characterize the aquifer near the creek. The location of VAP4 was chosen because it is hydraulically upgradient of the landfill, enabling characterization of ambient (that part of the aquifer not affected by landfill leachate) water quality in the aquifer. VAP consisted of lithologic sampling, water-quality sampling, water-level measurements, and slug testing at discrete depths along each boring. The first step in the VAP was to obtain a geologic profile by collecting lithologic samples. Lithologic samples were collected over an interval of 3 to 5 ft approximately every 10 ft (for example, at 5-10 ft and 15-20 ft) to a depth of 100 ft or sampler refusal, whichever came first. Lithologic samples were collected by use of a Geoprobe Systems Macrocore sampler with a length of 4 ft and a diameter of 1.5 in. The next step in the VAP was the collection of water-quality samples from four different depths at a second VAP location approximately 10 ft from the location of the lithologic profile. Water-level measurements and slug tests were performed in each of the VAP intervals where water-quality samples were collected (table 1). The depth intervals were selected to sample four approximately equally spaced intervals of the sand-and-gravel deposits identified by the lithologic profiling at each VAP location. Water-quality samples, water levels, and slug tests were collected by use of a temporary well screen with a length of 4 ft and a diameter of 1 in.

The five monitoring wells constructed for this investigation (GS1S, GS1D, GS2S, GS3D, and GS4D; fig. 3), hereafter referred to as the new monitoring wells, were installed July 7 and 8, 2005. The new monitoring wells were drilled using a 8.25 in. outside-diameter hollow-stem auger to the depth of the screened interval. The wells were installed at the VAP locations and are constructed of a flush-threaded, 2-in. diameter schedule 40 polyvinyl chloride (PVC) screen and riser pipe to be consistent with construction materials used for the previously installed monitoring wells. Screens were slotted, with a 0.010 in. opening and a length of 5 ft. The depth of the screens was selected based on the results of the water-quality and hydraulic data collected from the VAP at that location. A granular sand pack was placed in the annular space from the bottom of the auger typically to about 1.5 ft above the top of the screen. About 1 ft of filter sand was placed above the granular sand pack. About 2 ft of bentonite pellets were placed above the filter sand. Bentonite grout was used to fill the borehole annulus to about 1.5 ft below ground surface, where an above-grade protective well casing encased in concrete was installed. Waterproof locking caps were used to seal the top of the casing at each well.

Monitoring wells were developed using a combination of bailers, a 1.7 in. diameter hand pump, and a high-capacity submersible sampling pump. A minimum of five well volumes of water were removed during redevelopment of wells NL3 and NL4. A minimum of 25 well volumes were pumped during development of the new wells to remove water affected by drilling and well construction from the aquifer. The newly constructed wells were developed at least 5 days after construction and at least 13 days prior to sampling. Wells NL3 and NL4 were redeveloped 15 days prior to the first sampling event of this investigation.

Water levels were measured in the available previously installed monitoring wells on February 28, 2005, and in the new and previously installed monitoring wells on July 13, 27, 29, and August 15, 2005. Because water levels measured on all dates were similar to those measured on July 13, only the data from July 13 are discussed. Water levels were measured in each of the VAP test intervals at the time of sampling. All ground-water levels were measured by use of an electric water-level indicator graduated at 0.01 ft and were measured from the top of the riser pipe in each of the wells and VAP locations.

The elevation of the land surface and the top of the riser pipe at each well and VAP location was measured by USGS personnel using a standard, closed loop, rod-and-level technique. The latitude and longitude of each well was measured by USGS personnel by use of a hand-held global positioning device. All elevations are presented relative to an internal datum for the site. Ecology and Environment, Inc. (1986) used an internal datum for the site when they surveyed the elevation of various points on the previously installed wells. The identity of this datum was not specified. However, the elevation of the top of the inner casing of all of the previously installed wells was recorded. This elevation was measured to be 91.48 ft at well NL3. Because the top of the inner casing of well NL3 was used as the point of reference for the USGS surveying, the internal datum for this investigation is a horizontal plane located 91.48 ft below the top of the inner casing of well NL3.

The stage of the creek was measured on July 13, 2005 near well GS2S (fig. 3). The stage was measured by surveying the elevation of the water surface using the rod-and-level technique.

Horizontal hydraulic conductivity (K_h) of the deposits was calculated from data collected by USGS personnel during slug tests in previously installed monitoring wells NL3 and NL4, in all but one of the VAP test intervals, and in all of the new monitoring wells. Slug tests in the monitoring wells involved instantaneous lowering of the water level in the well by removal of a solid PVC cylinder from the water column and measurement of water-level rise (recovery) with time. Slug tests in the VAP locations involved use of compressed nitrogen gas to lower the water level then measurement of water-level rise with time when the pressure was instantaneously released. Recovery was measured in both monitoring wells and VAP locations by use of a calibrated 0-10 lb/in^2 pressure transducer connected to a datalogger. Water-level measurements collected from the monitoring wells were collected on a logarithmic time scale, whereas water-level measurements collected from the VAP locations were collected on a linear scale of 1 second or less. A minimum of two slug tests were done on each well and a minimum of three slug tests were done at each VAP test interval. Water levels were measured prior to the start of each test to ensure that levels had stabilized.

Data from many of the slug tests displayed a continuous decline in water levels through time and were analyzed using the technique of Bouwer and Rice (1976). This technique was developed for use in unconfined aquifers with wells that fully or partially penetrate an aquifer. Water levels from some of the slug tests exhibited an oscillatory response, which was analyzed using the van der Kamp (1976) technique. This technique was developed for analysis of slug-test data from highly permeable aquifers in which the effects of inertia of water in the well dominate the aquifer response (underdampened case). A fully penetrating well in a confined aquifer is assumed for this technique. The aquifer storage coefficient was assumed to be 0.001 when aquifer transmissivity was calculated. This storage coefficient is probably representative of confined conditions where the water level is above the well screen. The thickness of the aquifer was assumed to be equal to the saturated length of the well screen when K_h was calculated from the transmissivity values.

The van der Kamp and Bouwer and Rice techniques further assume the following conditions:

1. Drawdown of the water table in the vicinity of the well or VAP location is negligible,
2. Flow above the water table can be ignored,
3. Head losses as the water enters the well are negligible, and
4. The hydraulic unit tested is homogenous and isotropic.

These conditions appear to have been met or approximated in the locations tested during this study.

The following assumptions were made to analyze slug-test data:

1. The radius of the casing is equal to the radius of the inner casing of the monitoring wells (1 in.) or the riser pipe of the VAP locations (0.625 in.).
2. The length of the well through which water enters the aquifer is equal to the length of the well screen.
3. The borehole radius is equal to one-half the nominal outside diameter of the auger used to drill the well (8.25 in.) or one-half of the outside diameter of the VAP casing (1.125 in.).

Water samples were collected from the monitoring wells using standard low-flow techniques in accordance with the USEPA approved Field Sampling Plan for the investigation. Monitoring wells NL3 and NL4 were sampled on March 3, 2005. Monitoring wells GS1S, GS1D, GS2S, NL3, GS3D, NL4, and GS4D were sampled on July 27, 2005. A submersible sampling pump was used to purge and collect samples from the monitoring wells. Field parameters (temperature, pH, dissolved-oxygen concentration (DO), oxidation-reduction potential (ORP), and specific conductance) of the water were measured during purging by use of a calibrated Hydrolab Surveyor Datasonde 4 with an in-line flow-through cell attachment. Samples for field measurement of turbidity were collected at the end of purging and measured by use of a calibrated Hach Model 2100P portable turbidity meter. Samples for laboratory analysis were collected after a minimum of three well volumes had been purged and monitoring had established that the field parameters were changing by less than 10 percent for every half of a well volume.

On March 3, 2005, a water sample was collected from a surface seep near former wells NL6 and NL6A (fig. 3). A hole was dug about 20 in. deep at the seep and filled with about 8 in. of water in about 10 minutes after digging. The sample was then collected by partly submerging the bottles in the water. Field parameters were analyzed by submerging the probes in the water. The seep was not present during the July sampling event.

Samples from the VAP locations also were collected using standard low-flow techniques in accordance with the USEPA-approved Field Sampling Plan for the investigation. Water samples collected from the VAP locations were collected after purging 3-8 well volumes with a bladder pump. Temperature, pH, DO, ORP, and specific conductance of the water typically were measured during purging by use of a calibrated in-line flow-through cell. pH was not measured at VAP1, and pH, ORP, and DO was not measured in the deepest interval of VAP2 because the probes were broken. Water samples were collected from VAP locations after monitoring established that the field parameters were changing by less than 10 percent for every gallon of water purged and after at least three well volumes were purged. Chloride and total alkalinity concentrations of the samples at VAP2, VAP3, and VAP4 were measured in the field immediately prior to sampling by use of colorimetric titration techniques. Measurement of chloride and total alkalinity analyses from VAP locations was not planned originally. These analyses were added when the field kits containing the titration equipment became available, which was after VAP1 had been sampled. The chloride and total alkalinity analyses made with the field kits provide "field level" data. The concentration of chloride and total alkalinity determined from these samples is not considered to be highly accurate.

Water samples from the monitoring wells, the seep, and the VAP locations submitted for laboratory analysis were placed in pre-preserved bottles, immediately stored on ice in a cooler, and delivered to the laboratory typically within 72 hours of collection. The samples were not filtered. Samples collected from the VAP locations were analyzed by the laboratory only for volatile organic compounds (VOCs). Samples collected from the monitoring wells and the seep were analyzed for VOCs, metals, major cations, cyanide, pesticide compounds, sulfate, ammonia as nitrogen (ammonia), nitrate as nitrogen (nitrate), chloride, and alkalinity as calcium carbonate (alkalinity). All laboratory samples were analyzed by National Environmental Laboratory Accreditation Program (NELAP) certified laboratories that are approved for use by the IEPA. Samples were analyzed using USEPA standard methods according to the approved QAPP for the project. Rinsate blank, trip blank, matrix spike-matrix spike duplicate (MS/MSD), and duplicate samples were collected for quality assurance and quality control (QAQC) during the sampling efforts at the monitoring wells as specified in the QAPP. Proper chain-of-custody was maintained for all samples. Review of the QAQC data indicates the sampling results can be used without qualification. Because VAP data were used for screening purposes only, blank, MS/MSD, and duplicate samples were not collected at the VAP locations.

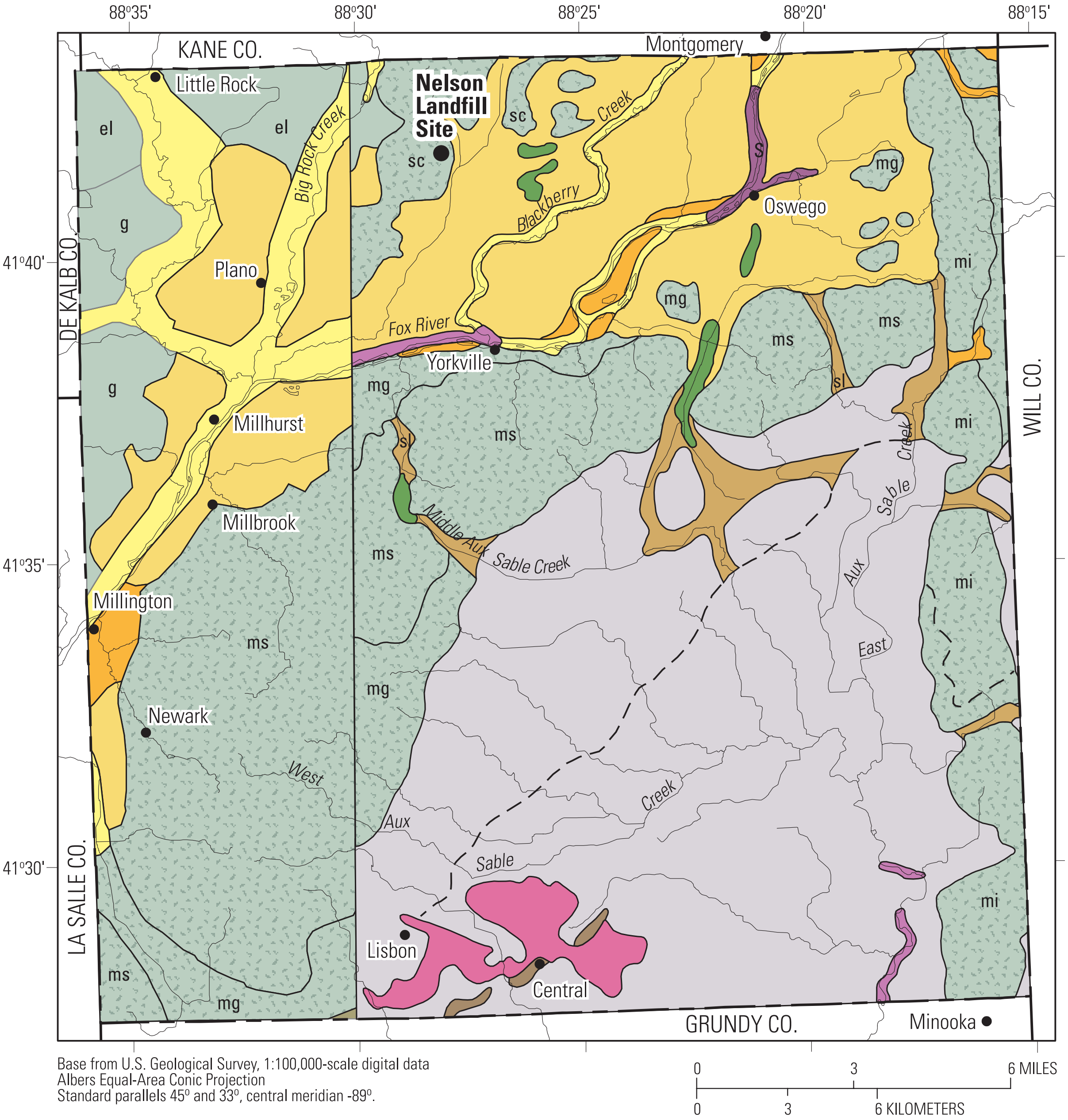
Concentrations of each constituent, including field parameters, detected in water samples from six or more monitoring wells during the July 2005 sampling event were plotted against each other, and the square of the Pearson product-moment correlation coefficient (r^2) for each plot was calculated by use of Microsoft Excel spreadsheet package. Two sets of plots were generated, one including and one excluding the data from wells NL4 and GS4D. Data from wells NL4 and GS4D were excluded because these wells are located directly hydraulically downgradient from the landfill and correlations excluding these wells are more likely to identify chemical processes occurring in the ambient aquifer. A high degree of correlation was considered to be present if the r^2 value for the plot exceeded 0.70 (Angel Martin, U.S. Geological Survey, oral commun., 2005).

Additional information on data collection and sampling methods is available from the USGS Illinois Water Science Center. Copies of the Field Sampling Plan, QAPP, and Work Plan also are available from the USGS Illinois Water Science Center.

GEOLOGY

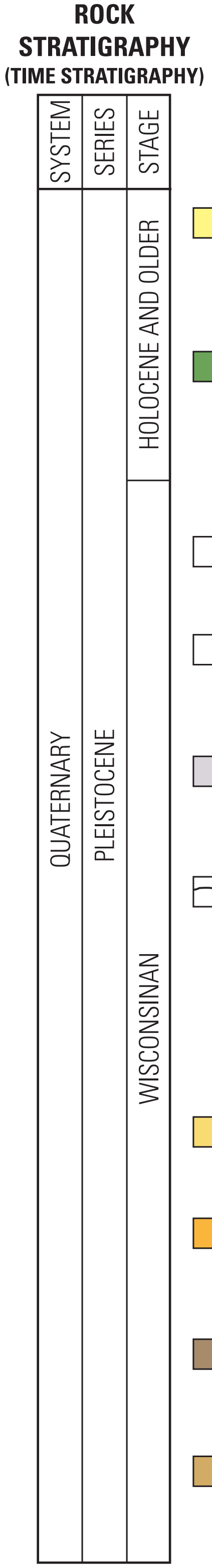
The study area is located at the surface contact between the Saint Charles Moraine, which is composed of the Yorkville Member of the Lemont Formation, and outwash plain deposits composed of the Batavia facies of the Henry Formation (fig. 4). Regionally, the Yorkville Member of the Lemont Formation is composed of clayey to silty till with local lenses of silt and sand and gravel deposited by glacial ice. The Batavia facies of the Henry Formation is composed of sand and gravel deposited in streams and rivers derived from glacial meltwater. These streams eroded into the bedrock surface during the Quaternary System, forming the Newark Bedrock Valley (fig. 1). These deposits interfinger throughout the Newark Bedrock Valley and are more than 100 ft thick in the vicinity of the site (fig. 1). The stratigraphic nomenclature used in this report is that of the Illinois State Geological Survey (Willman and others, 1975; Hansel and Johnson, 1996) and does not necessarily follow the usage of the USGS.

Lithologic logs obtained from previous site-specific investigations (Ecology and Environment, Inc., 1987) and as part of this study (table 2) indicate that the geologic deposits in the study area consist of two general types: sand and gravel, and silt and clay till. Although general patterns in distribution can be identified, these deposits are complexly distributed across the study area. Silt-and-clay rich till predominates within the upper 5 to 30 ft of land surface, with thin sand or sand and gravel interspersed (fig. 5). The shallow, predominantly till deposit tends to be thickest (about 25 to 30 ft) in the



Base from U.S. Geological Survey, 1:100,000-scale digital data
Albers Equal-Area Conic Projection
Standard parallels 45° and 33°, central meridian -89°.

Figure 4. Surficial geology of Kendall County, Illinois (west of 88° 30′ modified from Hansel and Johnson, 1996, plate 1, map scale 1:500,000; east of 88° 30′ and stratigraphic descriptions modified from Willman and Lineback, 1970, map scale 1:250,000, and Hansel and Johnson, 1996; mapping of the eastern and western parts of county at different levels of detail and presentation scale accounts for the apparent discontinuity in surficial geology). Illinois State Geological Survey stratigraphic nomenclature is used.



🟨 CAHOKIA FORMATION
Deposits in floodplains and channels of modern rivers and streams; mostly poorly sorted silt and sand containing local deposits of sandy gravel; in many places overlies relatively well-sorted glacial outwash of the Henry Formation.

🟩 GRAYSLAKE PEAT
Peat, muck, and locally marl; dominantly organic deposits with interbedded silt and clay in places; mostly in glacial lake basins; locally in lake basins on floodplains of major rivers.

MASON GROUP

1 PEORIA SILT -- See aslo EXPLANATION below
Generally wind deposited; light yellow tan to gray; grades from sandy on bluffs to clayey away from bluffs; lacks bedding structure; locally contains sand beds.

1 ROXANA SILT -- See also EXPLANATION below
Generally wind deposited; brownish red where oxidized, gray where water saturated; may contain up to 30 percent sand and 20 percent clay where it overlies sand deposits; generally massive with root traces

⬜ EQUALITY FORMATION
Lake plain deposits; floors of glacial lakes flattened by wave erosion and minor deposition in low areas; largely underlain by till; thin deposits of silt, clay, and sand present locally.

〰 *Lake shoreline*
Shorelines within lake plains; usually consists of a slight beach ridge or erosional scarp forming a terrace; discontinuous deposits of sand or pebbly sand of the Equality Formation generally present.

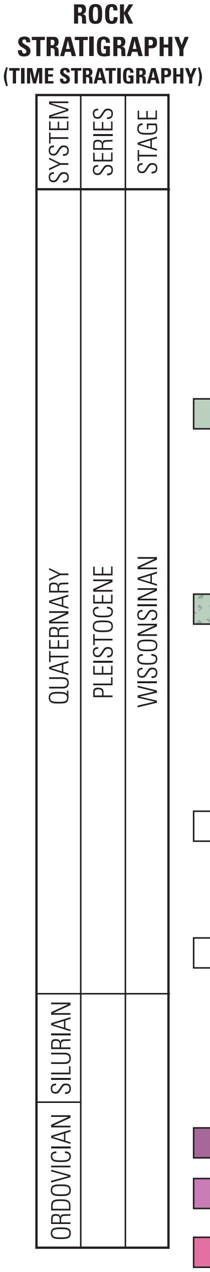
HENRY FORMATION
Sand and gravel with minor and local beds of silt; largely glacial outwash but locally includes deposits of outlet rivers of glacial lakes; outwash deposits generally have a thin cover of silt.

🟨 Batavia facies
Sand and gravel, well sorted; deposits in uplands; deposits of glacial meltwater rivers and streams in outwash plains.

🟧 Mackinaw facies
Sand and gravel, generally well sorted and evenly bedded; deposits in valleys; mostly glacial outwash in terraces--remnants of valley trains; includes similar deposits in glacial sluiceways.

🟫 Dolton facies
Largely shallow-water, near-shore lake sediments in beaches; dominantly medium-grained sand; contains beds of silt where gradational to quiet-water deposits; local lenses of sandy gravel along beaches.

🟫 *Glacial sluiceway*
Erosional channels; mostly outlets of glacial lakes where cut into till; contains local deposits (mostly bars) of sand and gravel of the Henry Formation.



WEDRON GROUP

LEMONT FORMATION
Gray, fine to coarse textured (silty clay to sandy loam) till with lenses of gravel, sand, silt, and clay, silt represents 30-50 percent of matrix; locally may overlie coarsening-upward sequences of sand and gravel.

🟩 Undivided
Mostly yellow, tan-gray, or slightly pinkish gravelly silt loam to loam till; typically derived from eroded Silurian dolomite; units laterally contiguous with other members of the Lemont Formation commonly are coarser than the units that compose those matters.

el	Elburn Morainic Complex
g	ground moraine, undifferentiated

▨ YORKVILLE MEMBER
Mostly gray to dark gray clayey till, locally silty clayey till; contains abundant small pebbles (especially in Marseilles Moraine), local lenses of silt, and less commonly lenses of sand and gravel.

mi	Minooka Moraine
ms	Marseilles Moraine
sc	St. Charles Moraine
mmg	Manhattan-Minooka Ground Moraine
mg	Marseilles Ground Moraine

1 BATESTOWN MEMBER -- See also EXPLANATION below
Gray, medium textured (loam) till; contains lenses of gravel, sand, silt, and clay; typically oxidizes brown, olive brown, or yellow brown.

1 TISKILWA MEMBER -- See also EXPLANATION below
Red gray to gray, medium textured (clay loam to loam) till with local lenses of clay, silt, sand, and gravel; oxidizes to red brown, brown, or yellow brown.

BEDROCK UNITS

🟪 Undifferentiated

🟪 MAQUOKETA GROUP
Shale and dolomite, predominantly.

🟥 GALENA GROUP
Dolomite, predominantly.

EXPLANATION

1 UNMAPPED -- These units are either surficial deposits that extensively cover Kendall County (Peoria Silt) or are largely or fully buried beneath younger deposits (Roxana Silt, Batestown, and Tiskilwa Members of the Lemont Formation.

Figure 4. Surficial geology of Kendall County, Illinois—continued.

western part of the study area at VAP4 and well NL2; of intermediate thickness (about 10 to 20 ft) at NL3/VAP2, NL4/VAP1 and NL6; and thinnest (0 to 5 ft) in the center of the study area at wells NL1 and NL5 and near the creek at VAP3. Sand-and-gravel deposits underlie the till and are the predominant lithology from a depth interval of about 16 ft to at least 80 ft at VAP1 and to at least 100 ft at VAP2 in the south-central part of the study area. Sand-and-gravel deposits predominate from about 25 to 71 ft below land surface at VAP4 in the northwestern part of the study area and in much of the interval from 0 to at least 83 ft at VAP3 near the creek. Till deposits were present from about 71 to 100 ft at VAP4 and were interspersed throughout the interval from about 49 to 76 ft at VAP3.

The type and heterogeneity of the geologic deposits in the study area are consistent with regional data for the Newark Bedrock Valley (Kay and others, 2005) and it is likely that the geologic conditions at the site are representative of the Newark Bedrock Valley as a whole. The vertical and horizontal heterogeneity of the glacial-drift deposits in the study area, an area approximately 2,000 ft in length and width, indicates that persons seeking to develop water supplies from the Newark Bedrock Valley will need to perform a detailed geologic exploration to locate the most productive water-bearing deposits.

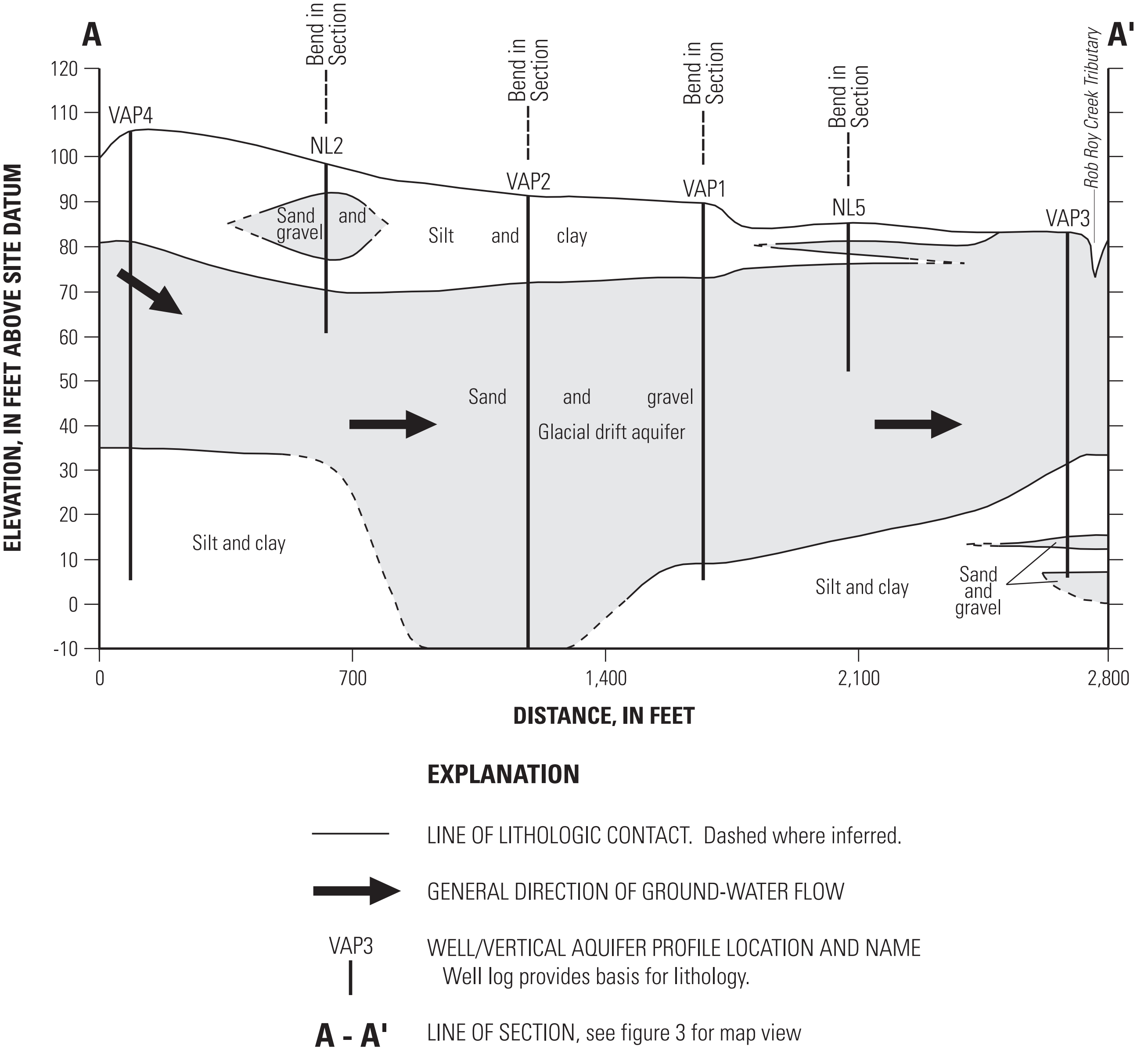


Figure 5. Hydrogeologic section A-A′ in the vicinity of the Nelson Landfill site.

Table 2. Lithologic logs for the vertical aquifer profile (VAP) locations in the vicinity of the Nelson Landfill site.

[ft, feet; cm, centimeter]

Depth (feet below land surface)	Description
	VAP1
5-10	5-6 ft is tan silt and clay till, some gravel, sandy. 6-8 ft is silt and clay till. 8-10 ft is silt, sand, and gravel. Dry.
15-20	15-16 ft is grey till composed of silt, clay, sand, gravel. 16-20 ft is tan sand and gravel. Saturated.
25-30	Grey sand and gravel.
36-40	Tan, well sorted, medium-grained sand.
49-53	Sand and gravel. Mostly gravel-sized grains from about 0.3 to 1 cm in size. Some silt and clay.
53-55	Soft zone when drilling.
57-61	Grey sand and gravel.
66-70	Predominately grey medium-grained sand. Some gravel up to about 1 cm in size.
78-81	78-80.5 is sand and gravel, grains up to about 3 cm in size. 80.5-81 is grey till.
83	Refusal.
	VAP2
7-10	Brown till. Gravel in sand and silt matrix.
17-20	17-17.5 ft is grey silty till. 17.5-19 ft is grey clay, plastic. 19-20 ft is tan coarse sand, about 1 percent gravel. Gravel composed of lithic fragments, subrounded, maximum size about 2 cm.
27-30	Tan coarse-grained sand.
37-40	Very coarse-grained sand. Mostly quartz, some lithic fragments. About 1 percent gravel up to 1 cm in size.
47-50	Tan, very coarse-grained sand. Mainly quartz, some lithic fragments.
57-60	Sand and gravel. Largest gravel grains larger than size of sampler (2 inches).
68-71	Grey sand and gravel. Mostly quartz (smaller size) and lithic fragments (larger size). Most grains from 0.1 to 3 cm in size.
79-82	79-81.5 ft is tan medium- to coarse-grained sand. 81.5-82 ft is sand and gravel.
99-102	Very coarse-grained sand and gravel. Composed of lithic fragments.
	VAP3
7-10	Grey silty sand and gravel. Gravel is rounded to subrounded lithic fragments. Maximum diameter about 1.5 inches. Moist.
17-20	Grey sand and gravel. Similar to 7-10 ft but little silt.
30-34	Same as 17-20 ft.
40-44	Same as 17-20 ft.
49-52	Tan sandy silt and very fine-grained sand.
57-60	Grey sandy silt.
61-67	Very hard drilling.
67-70	Tan coarse-grained sand and gravel, silty.
70.1-74.9	Very hard drilling.
75-78	75-76 ft is grey till. Cobbles in silt/sand/clay matrix, same as 80-80.5 ft at VAP1. 76-78 ft is grey sand and gravel. Indications of sand and gravel at about 79-82 ft but no sample.
	VAP4
8.5-12.5	8.5-11.5 ft is brown till. Gravel in sand and silt matrix. Dry. 11.5-12.5 ft is brown clay till, moist.
19-23	Brown clay and silt till. Wet.
29-33	Tan medium- to coarse-grained sand.
39-43	Tan sandy gravel. Gravel subrounded lithic fragments, mostly dolomite. Average gravel size about 1 cm.
49-53	Grey sand and gravel. Gravel lithic fragments. Maximum size beyond diameter of sampler. Mostly subrounded to rounded.
58-62	Grey sand and gravel. Same as 49-53 ft.
68-72	68-69 ft is grey coarse-grained sand. 69-71 ft is gravel. 71-72 ft is grey clayey, silty till. Gravel fragments up to 1 percent of sample. Plastic.
79-82	Brown silt till. Mostly silt, some sand and gravel in silt matrix. Gravel about 1 percent of sample. Often angular. Maximum gravel size about 1 cm. Plastic.
88-91	Same as 79-82 ft.
97-100	Same as 79-82 ft.

HYDROLOGY

The hydraulic unit of concern to this investigation is the aquifer composed of the sand-and-gravel deposits beneath the study area, hereafter referred to as the glacial drift aquifer (fig. 5). The silt and clay till deposits interspersed with the sand-and-gravel deposits are not considered part of the aquifer. The glacial drift aquifer was the focus of this investigation because it is a potential source for public-water-supply and because it is likely to be the primary pathway for contaminant migration associated with the landfill. On a regional basis, and beneath much of the study area, the glacial drift aquifer is under unconfined conditions. However, the glacial drift aquifer is locally confined by fine-grained deposits.

Water-level elevations measured in the previously installed monitoring wells on February 28, 2005, ranged from 77.32 ft above the internal site datum at well NL1 in the northern part of the study area to about 74.5 ft at wells NL3 and NL4 in the southern part of the study area (table 1). These water-level elevations indicate water in the glacial drift aquifer flows from north to south across the site.

Water levels were measured in all available monitoring wells and at the creek on July 13, 2005. Water-level elevations in wells open to the shallow part of the glacial drift aquifer (wells GS1S, NL1, GS2S, NL3, NL4, NL2) ranged from 77.78 ft above the internal site datum at well GS1S in the northwestern part of the study area to 73.68 ft at well GS2S in the southeastern part of the study area near the creek (table 1, fig. 6). Water-level elevations on July 13, 2005, are roughly consistent with patterns in surface topography (excluding the landfill itself) and indicate that water in the glacial drift aquifer flows from north to south across the study area, with a slight west-to-east component of flow. Comparison of water levels at well GS2S with the stage of the creek, which was nearly dry on this date, indicates virtually horizontal flow between the creek and the glacial drift aquifer, and good hydraulic interconnection between the aquifer and the creek. This interpretation assumes a margin of measurement error of about 0.04 ft. Northern Illinois, including the study area, was in a drought during the summer of 2005, and flow conditions defined by these measurements may not represent typical conditions.

Water levels measured in monitoring wells open to the deeper part of the glacial drift aquifer (wells GS1D, GS3D, GS4D) on July 13, 2005, ranged from 74.55 ft above the internal site datum at well GS1D in the northwestern part of the study area to about 73.80 ft at wells GS3D and GS4D in the southern part of the study area. Allowing for differences in the location and number of data points, flow direction in the deeper part of the glacial drift aquifer in the study area appears to be similar to flow direction in the shallow part. The flow direction is predominately from north to south with a slight west-to-east component (fig. 6).

The water-level elevation in the shallow part of the glacial drift aquifer was more than 3 ft higher than in the deep part of the aquifer at the GS1S/GS1D well cluster, indicating the potential for downward flow in this area (table 1, fig. 6). The water-level elevation in the shallow part of the glacial drift aquifer typically differed by less than 0.10 ft from the water level in the deep part of the aquifer at the NL3/GS3D and NL4/GS4D well clusters. Within the margin of measurement error (about 0.04 ft), water levels indicate the potential for downward or horizontal flow at the NL3/GS3D cluster and horizontal or perhaps slightly upward flow at the NL4/GS4D cluster. These water levels generally are consistent with the water-level data collected from the VAP intervals. Water-level elevation tended to increase in the deeper part of the glacial drift aquifer at VAP1 (the location of the NL4/GS4D cluster) and tended to decrease with depth at VAP2 and VAP4 (the locations of the NL3/GS3D and GS1S/GS1D clusters, respectively) (table 3). Water levels in VAP3 showed a tendency for horizontal flow, or to increase slightly with depth, indicating the potential for horizontal or perhaps upward flow of water from the glacial drift aquifer into the creek. The VAP data also is consistent with the water-level data from well GS2S and the creek (within the margin of measurement error). These data indicate that vertical flow in the glacial drift aquifer is complex.

If ground water from the glacial drift aquifer flows into the creek, it is likely that the creek defines the eastern boundary of any ground-water contamination leaching from the landfill. Furthermore, if contamination is present in the ground water in this area, the potential exists for that contamination to migrate into sediments and water in the creek, where they could impact the environment. Finally, if water flows into the creek, a large capacity well drawing water from the glacial drift aquifer in this area would have the potential to divert water from the creek, reducing the volume of streamflow. A reduction in streamflow could adversely affect the creek ecosystem.

The north to south direction of ground-water flow in the study area is generally consistent with the regional flow direction in the glacial drift aquifer (Kay and others, 2005). The flow direction in the study area on July 13, 2005, also is consistent with the flow direction determined from water levels measured in the previously installed monitoring wells on February 28, 2005 (table 1), and from measurements taken by Ecology and Environment, Inc., (1986, 1991) in the mid-1980s. These measurements indicate that the flow direction in the glacial drift aquifer beneath the study area has been consistent for at least the past 20 years. It is probable, there-

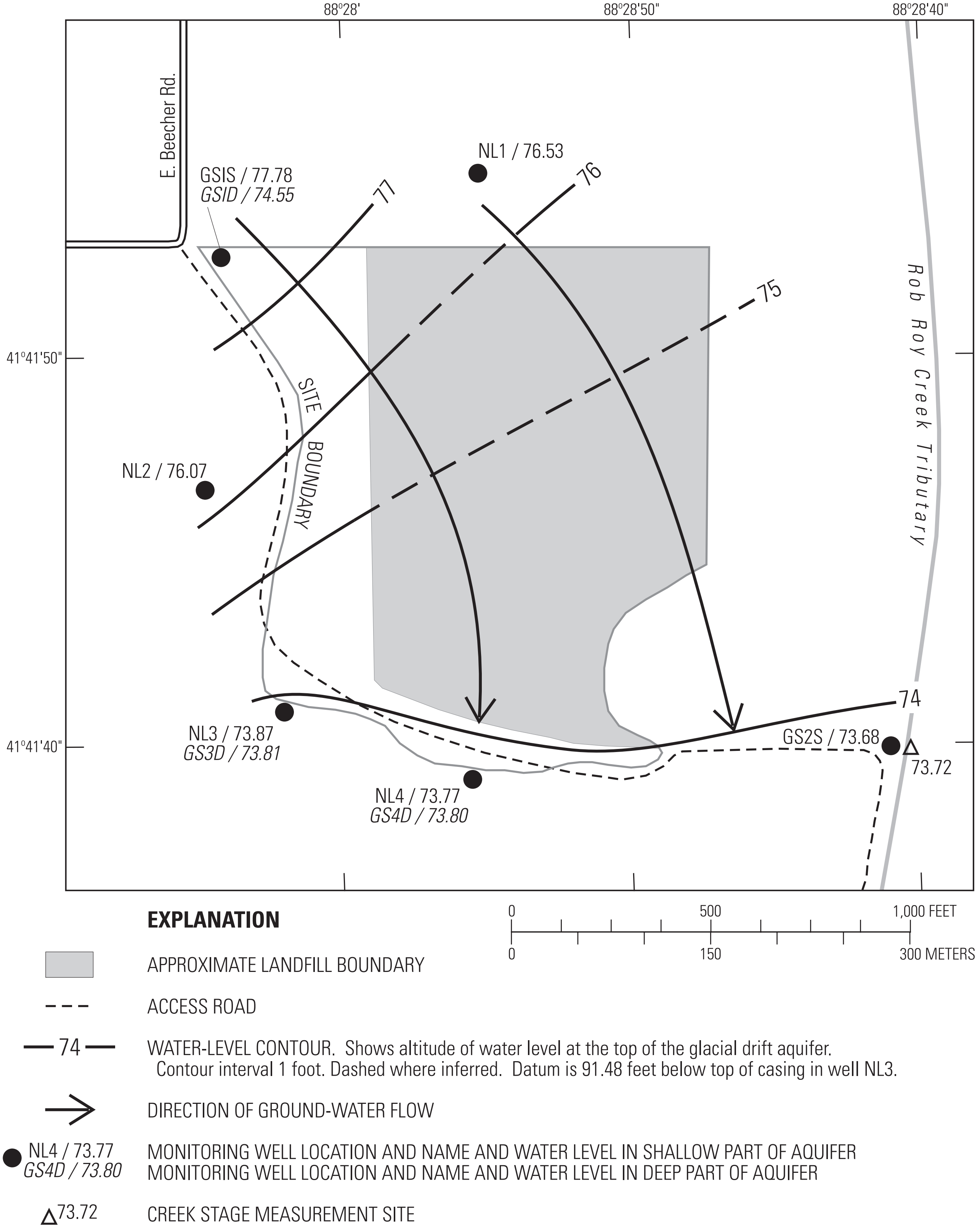


Figure 6. Water levels in monitoring wells and potentiometric surface of the shallow part of the glacial drift aquifer in the vicinity of the Nelson Landfill site, July 13, 2005.

Table 3. Vertical aquifer profile data in the vicinity of the Nelson Landfill site, April 4-27, 2005.

[Datum is 91.48 feet below top of inner casing in well NL3; nd, not detected; nm, not measured; minus, denotes that value is below datum]

Vertical aquifer profile location (shown in figure 3)	Depth of test interval (feet below land surface)	Elevation of test interval (feet above site datum)	Elevation of water level (feet above site datum)	Average horizontal hydraulic conductivity (feet per day)	Volatile organic compound detections	Temperature (degrees Celsius)	pH (standard units)	Specific conductance (microSiemens per centimeter)	Chloride concentration from field test (milligrams per liter)	Alkalinity from field test (milligrams per liter as Calcium Carbonate)
VAP1	38-42	49-45	74.36	11	nd	11.9	nm	1,320	nm	nm
VAP1	48-52	39-35	74.51	.27	nd	10.7	nm	1,510	nm	nm
VAP1	59-63	28-24	74.37	1.6	nd	11.9	nm	1,430	nm	nm
VAP1	75-79	12-8	76.52	.81	nd	13.0	nm	1,750	nm	nm
VAP2	37-41	52-48	74.47	.82	nd	13.2	7.6	770	30	280
VAP2	57-61	32-28	74.45	18	nd	13.2	7.7	790	40	320
VAP2	77-81	12-8	74.44	18	nd	13.0	7.5	830	40	340
VAP2	97-101	minus 8 - minus 12	74.40	.53	nd	nm	nm	nm	35	300
VAP3	19-24	61-57	74.32	2.6	nd	8.4	7.2	970	50	320
VAP3	29-33	51-47	74.30	48	nd	8.4	7.3	1,000	45	540
VAP3	39-44	41-36	74.34	.46	nd	8.0	7.1	930	50	270
VAP3	79-83	1 - minus 3	74.34	1.7	nd	11.8	7.1	1,130	100	330
VAP4	31-35	75-71	77.39	nm	nd	11.8	6.8	1,340	90	570
VAP4	40-44	66-62	76.71	31	nd	12.0	7.3	1,180	120	370
VAP4	50-54	56-52	76.66	34	nd	10.9	7.4	700	25	250
VAP4	65-69	41-37	76.69	36	nd	10.4	7.4	730	20	200

fore, that the general direction of contaminant migration also has not changed in that time.

Values for the K_h of the aquifer calculated from analysis of slug tests done in the VAP test intervals and the monitoring wells ranged from 0.27 to 640 ft/d (tables 1 and 3). The geometric mean K_h of all of the slug-test values is 9.2 ft/d. The K_h values displayed no obvious patterns with depth, location, or lithology (sand, sand-and-gravel, or gravel) but indicate that at least parts of the glacial drift aquifer are highly permeable and may be capable of sustaining withdrawals from individual wells in excess of 50 gal/min. However, the variation in K_h values, even within those parts of the glacial drift aquifer consisting of permeable sand-and-gravel deposits, indicates that the successful installation of high-capacity wells and well fields in this aquifer will require a detailed geologic exploration and hydrologic evaluation of the aquifer.

The horizontal hydraulic gradient for the shallow part of the glacial drift aquifer was calculated by dividing the change in the elevation of the potentiometric surface between two points on a line parallel to the direction of flow (4 ft, based on data from wells GS1S and NL4) by the horizontal distance of the flow path between the two points (approximately 1,600 ft)(fig. 6). The horizontal hydraulic gradient was calculated to be about 2.5×10^{-3} ft/ft based on the water levels measured on July 13, 2005. The low value of this gradient is consistent with a permeable aquifer. However, this gradient may be atypically low because of the effects of drought conditions at this time.

The ground-water velocity (v) in an aquifer is defined as

$$v = (K_h/n_e)I\ , \qquad (1)$$

where n_e is the effective porosity of the aquifer, in percent, and
I is the horizontal hydraulic gradient, in ft/ft.

If K_h is assumed to be equal to the geometric mean value for the glacial drift aquifer (9.2 ft/d), I is equal to the value determined from the July 13, 2005, data, (2.5 x 10^{-3} ft/ft), and n_e is assumed to be 30 percent, the average ground-water velocity in the glacial drift aquifer is calculated to be about 7.7 X 10^{-2} ft/d. This velocity is likely to be low because of the effects of the drought conditions on the hydraulic gradient. The velocity calculation indicates that the ground water underlying the landfill area when operations began in 1967 had migrated about 1,070 ft downgradient by the time this investigation occurred in 2005.

Lithologic and water-level data collected as part of the current and previous investigations indicate that the aquifer is about 100 ft thick in the southern part of the landfill. If the landfill has a width of about 1,000 ft perpendicular to the direction of ground-water flow, the volume of water (Q) flowing through 1 ft^2 of the aquifer is given by

$$Q = K_h AI, \qquad (2)$$

where A is the cross-sectional area through which water is flowing (1,000 ft by 100 ft), I is the horizontal hydraulic gradient (2.5 X 10^{-3} ft/ft), and K_h is the mean horizontal hydraulic conductivity (9.2 ft/d). Solving equation 2 indicates that about 2,300 ft^3/d (about 16,900 gal/d) of water flows through the aquifer beneath the landfill based on the hydraulic conditions present on July 13, 2005.

WATER QUALITY

Water-quality data collected from the seep and the monitoring wells fall into five general categories: field parameters, major ions, nitrogen compounds, organic compounds, and metals and cyanide. Water-quality data collected from the VAP locations consisted of field parameters, field measurements of concentration of chloride and alkalinity, and VOCs.

Water samples also were collected from three locations in the creek on June 17, 2005, by Civil and Environmental Consultants Inc. as part of a separate investigation (fig. 2). Creek samples were analyzed for VOCs, nitrogen compounds, cyanide, arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver (Gordon Stevens, Civil and Environmental Consultants Inc., written commun., 2005). Site SS1 is upstream of the creek area receiving ground water that may have been chemically affected by the landfill. Site SS2 is near well GS2S and in a part of the creek that may be receiving ground water that may have been chemically affected by the landfill. Site SS3 is about 900 ft south of the landfill, south of the confluence with the main branch of Rob Roy Creek, and in a part of the creek that also may have been receiving ground water that was chemically affected by the landfill.

Field Measurements

Turbidity is a measure of the clarity of water and is an indication of the amount of particulate matter in the sample. Because the presence of particulate matter in a sample container can affect the concentration of numerous dissolved constituents in the water, the most representative water samples should be those with minimal turbidity. Sample turbidity was not measured at the VAP locations, but the samples were highly turbid based on visual observation, and the buildup of particulate matter in the flow-through cell substantially affected the measurements of DO and ORP despite efforts to keep the probes clean. As a consequence, DO and ORP values from the VAP sampling are not reported. Although it was not quantified, the turbidity of the seep sample was high based on visual observation, which may have affected the sample results. The turbidity of the samples collected from the monitoring wells typically was less than 10 nephelometric turbidity units (NTUs) (table 4). Turbidity values showed a low degree of correlation with the value of every other constituent (table 5), indicating that the sample results are not substantially affected by interactions with particulate matter and can be considered representative of in-situ water quality.

Temperature of the water samples from the monitoring wells and the VAP locations ranged from 8.0 to 14.2 ^{o}C (tables 3 and 4). The temperature value likely was affected by the ambient air temperature during sampling. Therefore, temperature readings are not considered representative of in-situ water quality, and spatial or temporal patterns in water temperature were not determined.

Specific conductance is a measure of the capacity of a solution to conduct an electrical current and commonly is correlated with the total concentration of dissolved constituents. Specific conductance of the samples from the monitoring wells (fig. 7) and the VAP locations ranged from 520 to 1,970 µS/cm (tables 3 and 4). Specific conductance of samples from the monitoring wells and the VAP showed similar values and patterns. Because the data from the VAP locations is more extensive, this discussion focuses on the specific-conductance values collected during the VAP. Specific conductance values at VAP4, the location most likely to be representative of the ambient aquifer (fig. 3), ranged from 704 to 1,338 µS/cm and were almost a factor of two higher in the shallow part of the aquifer than in the deep part. Specific conductance values from VAP1, VAP2, and VAP3 (fig. 3) tended to be slightly higher in the deeper part of the aquifer at each location, but showed no large, easily identified variations with depth. Specific conductance values tended to be lower overall at VAP2 and

Table 4. Field parameters measured in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site.

[nm, not measured]

Sample location (shown in figure 3)	Temperature (degrees Celsius)	Dissolved oxygen concentration (milligrams per liter)	Oxidation-reduction potential (millivolts)	pH (standard units)	Specific conductance (microSiemens per centimeter)	Turbidity (nephelometric turbidity units)
March 3, 2005						
NL3	9.2	2.24	80	7.0	530	5.6
NL4	10.2	.30	69	6.7	1,190	1.1
Seep	3.3	5.54	86	6.8	715	nm
July 27, 2005						
GS1S	12.6	3.53	80	6.8	1,175	5.5
GS1D	12.1	.09	-102	7.4	720	2.4
GS2S	12.2	.14	-71	7.1	1,040	5.7
NL3	14.2	2.79	59	7.2	520	12.3
GS3D	12.4	.13	-18	7.1	890	3.9
NL4	11.1	.16	-20	6.9	1,145	3.1
GS4D	12.9	.10	-72	6.8	1,970	2.1

higher overall at VAP1 (located directly downgradient of the landfill) than in the remaining VAP locations. Specific conductance of the seep sample was 715 µS/cm. This moderate value is assumed to indicate the presence of recent snowmelt mixed with leachate at the seep during sample collection.

pH is the negative log of the concentration of hydrogen ions in the water and can affect the concentration of a number of constituents in water from the aquifer. The pH of ground-water samples from the monitoring wells and the VAP locations ranged from 6.7 to 7.7 (tables 3 and 4). The pH of the seep sample was 6.8. These values are all near the neutral pH of 7.0.

The oxidation-reduction potential (ORP) of water is a measure of electron activity and is an indicator of the relative tendency of a solution to accept electrons from, or lose electrons to, the surrounding environment during chemical reactions. The lower the ORP value, the more reducing, or electron accepting, the solution. The ORP of a solution affects the valence state of iron, nitrogen, sulfur, and other constituents, which in turn can affect their form and solubility. The ORP of a solution also affects, and is affected by, the type and activity of the microbiologic communities in an aquifer, which can affect the breakdown of nitrogen compounds and some types of VOCs. ORP values in samples from the monitoring wells ranged from -102 to 80 mv (fig. 7, table 4). ORP values decrease with depth at each monitoring well cluster, indicating that the aquifer becomes more reducing with depth below the water table. The positive values for ORP detected in wells NL3 and GS1S, which are open to the upper 15 ft of the aquifer (table 1), coupled with the negative ORP value at well GS2S, which is open to the shallow part of the aquifer but is screened about 25 ft below the top of the aquifer (table 1), indicates that the transition between the more oxidizing and more reducing conditions may occur in an interval within the upper 15-25 ft of the aquifer. Comparison of ORP values at wells NL3 and NL4 from March and July 2005 indicates that values were lower in July, especially at well NL4. This decrease in ORP may reflect the effects of reduced recharge of oxygenated precipitation during the drought on geochemical conditions in the shallow part of the aquifer. The ORP of the seep sample was 86 mv.

Concentrations of dissolved oxygen (DO) are an indication of ORP conditions in an aquifer. DO concentrations of about 0.5 mg/L or less typically are considered representative of anoxic (oxygen deficient) waters. The removal of DO from the aquifer can be a precursor to the development of more reducing geochemical conditions, which can have both beneficial and detrimental effects on water quality. The DO concentration in the water from the monitoring wells ranged from 0.09 to 3.53 mg/L (table 4, fig. 7). DO concentrations greater than 2 mg/L in wells NL3 and GS1S indicate the presence of moderate amounts of dissolved oxygen in the shallow (upper 15 ft) (table 1) part of the ambient aquifer. DO values less than 0.5 mg/L in the water from shallow well NL4 may reflect the effects of the landfill on water chemistry. DO values decreased with depth in the aquifer, particularly at NL3/GS3D and GS1S/D clusters, where substantial concentrations of DO were present in the shallow ground water. This pattern is consistent with the patterns in ORP. DO and ORP showed a high degree of positive correlation (r^2 greater than 0.80) (table 5) both with and without the data from wells NL4 and GS4D. This result is further indication

Table 5. Correlation of selected constituents in ground water from monitoring wells in the vicinity of the Nelson Landfill site, July 27, 2005. (Excludes data from wells NL4 and GS4D. Bold denotes correlation coefficient values greater than 0.70; positive values denote direct correlation, negative values denote inverse correlation)

Constituent	Barium	Iron	Manganese	Strontium	Calcium	Magnesium	Sodium	Potassium	Alkalinity	Sulfate	Chloride	Dissolved oxygen	Oxidation-reduction potential	Specific conduc-tance	pH
Barium	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Iron	**0.82**	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Manganese	.50	0.20	-	-	-	-	-	-	-	-	-	-	-	-	-
Strontium	.51	.43	**0.75**	-	-	-	-	-	-	-	-	-	-	-	-
Calcium	.43	.11	.48	0.19	-	-	-	-	-	-	-	-	-	-	-
Magnesium	.40	.11	.41	.20	**0.97**	-	-	-	-	-	-	-	-	-	-
Sodium	.00	-.09	-.02	-.22	.32	0.33	-	-	-	-	-	-	-	-	-
Potassium	**.83**	.51	.87	**.77**	.58	.54	0.00	-	-	-	-	-	-	-	-
Alkalinity	.10	.00	.19	.00	**.82**	**.78**	.68	0.18	-	-	-	-	-	-	-
Sulfate	**.90**	**.74**	.32	.25	.32	.25	.00	.59	0.07	-	-	-	-	-	-
Chloride	.59	.44	**.82**	**.99**	.28	.28	-.15	**.86**	.02	0.32	-	-	-	-	-
Dissolved oxygen	-.32	-.41	-.42	**-.88**	-.02	-.04	.43	-.47	.06	-.11	**-0.81**	-	-	-	-
Oxidation-reduction potential	-.33	-.58	-.16	-.63	.00	-.02	.34	-.31	.11	-.13	-.56	**0.86**	-	-	-
Specific con-ductance	.25	.04	.21	.04	**.92**	**.92**	.59	.31	**.91**	.19	.08	.00	0.00	-	-
pH	.00	.14	-.02	.04	-.40	-.32	**-.71**	.00	**-.78**	.00	.05	-.37	-.53	-0.53	-
Turbidity	-.10	-.09	-.17	-.35	-.27	-.41	-.01	-.25	-.11	.00	-.35	.36	.40	-.24	-0.01

that conditions are more reducing in the deep part of the aquifer. These data further indicate that biologic and chemical reactions are consuming dissolved oxygen in the shallow part of the aquifer, and that this process appears to be completed by the time water travels to the deeper part. Furthermore, DO values less than 0.5 mg/L in the water from well GS2S, which is screened more than 25 ft below the top of the aquifer (table 1), may indicate that the oxygen-consuming reactions are occurring within the upper 15-25 ft of the aquifer. The concentration of DO in the seep sample was 5.54 mg/L (table 4), which is indicative of the recent snowmelt source and perhaps the exchange of oxygen from the atmosphere into water in the shallow soil.

Major Ions

For the purposes of this report, the major cations are calcium, magnesium, sodium, and potassium. The major anions are chloride, sulfate, and alkalinity as milligrams per liter of calcium carbonate (table 6). Alkalinity in these samples is likely to be primarily due to the presence of the bicarbonate ion (HCO_3^-).

Chloride concentrations in the samples from the seep and monitoring wells ranged from 1.9 to 220 mg/L (table 6) and were similar between sampling events at wells NL3 and NL4. Chloride concentrations in the samples from the VAP locations ranged from 20 to 120 mg/L (table 3) and show poor to good correlation with the data from the monitoring wells, where similar intervals were tested. Chloride concentrations increased by at least a factor of four with depth at the GS1S/D, NL3/GS3D and NL4/GS4D well clusters (fig. 8), although they showed a less definite relation at the VAP locations. Overall chloride concentrations were higher at the NL4/GS4D well cluster, particularly well GS4D,

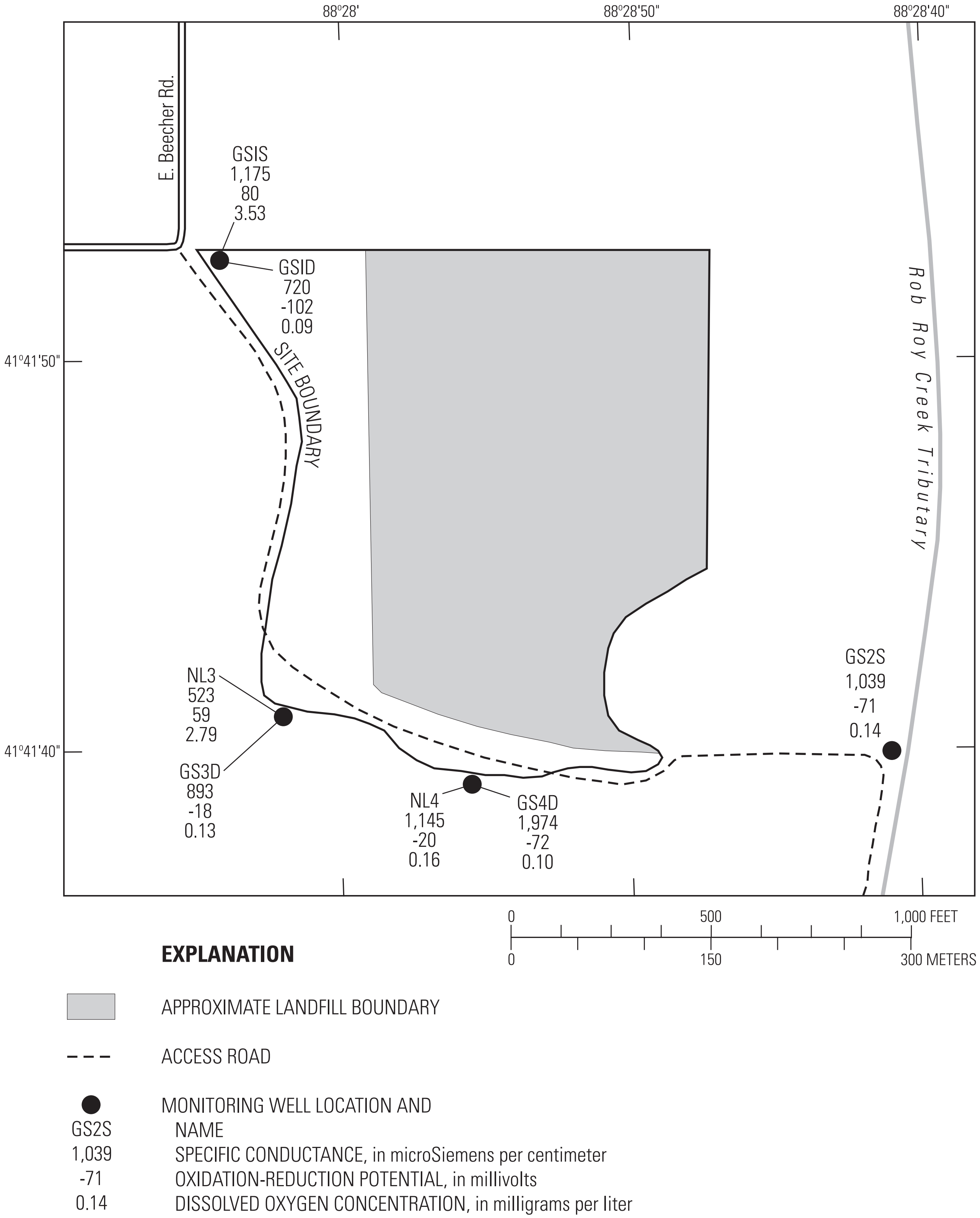


Figure 7. Value of selected field parameters in samples from monitoring wells in the vicinity of the Nelson Landfill site, July 27, 2005.

Table 6. Concentration of major ions and nitrogen compounds in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site.

[dup, duplicate sample; mg/L, milligrams per liter; na, none available; <, denotes constituent not detected and detection limit; MCL, U.S. Environmental Protection Agency Maximum Contaminant Level; SMCL, U.S. Environmental Protection Agency Secondary Maximum Contaminant Level]

	Locations sampled on March 15, 2005					
Constituent	NL3	NL4	NL4 (dup)	Seep	MCL	SMCL
Calcium (mg/L)	69.1	103	104	96.3	na	na
Magnesium (mg/L)	32.4	67.4	68.3	50.8	na	na
Alkalinity (mg/L as calcium carbonate)	290	500	490	280	na	na
Sulfate (mg/L)	19	99	100	70	na	250
Chloride (mg/L)	1.9	54	55	28	na	250
Ammonia (NH) as nitrogen (mg/L)	<.02	3	3.1	.51	na	na
Nitrate as nitrogen (mg/L)	<.10	<.10	<.10	<.10	10	na

	Locations sampled on July 27, 2005								
Constituent	GS1S	GS1D	GS2S	NL3	GS3D	NL4	GS4D	MCL	SMCL
Calcium (mg/L)	133	90.1	137	69	122	101	110	na	na
Magnesium (mg/L)	56.4	43.2	56.3	31.1	51.7	63.6	80.7	na	na
Sodium (mg/L)	44.9	5.86	12.3	5.29	8.72	59.1	147	na	na
Potasium (mg/L)	1.4	1.49	3.33	.63	2.51	8.48	36	na	na
Alkalinity (mg/L as calcium carbonate)	420	280	360	270	360	490	690	na	na
Sulfate (mg/L)	34	14	190	19	21	15	34	na	250
Chloride (mg/L)	5.2	22	39	2.1	36	46	220	na	250
Ammonia (NH) as nitrogen (mg/L)	<.02	.03	.98	<.02	<.02	3.4	31.7	na	na
Nitrate as nitrogen (mg/L)	3.8	<.10	<.10	<.10	<.10	<.10	<.10	10	na

than in the rest of the aquifer. Wells NL4 and GS4D are located directly downgradient of the landfill.

Alkalinity in the samples from the seep and monitoring wells ranged from 270 to 690 mg/L as $CaCO_3$, making it the dominant anion in the aquifer (table 6). Alkalinity concentrations at wells NL3 and NL4 were similar between sampling events. Alkalinity values in the samples from the VAP locations ranged from 200 to 570 mg/L as $CaCO_3$ (table 3) and show moderate to good correlation with the data from the monitoring wells, where similar intervals were tested. Alkalinity values increased with depth at the NL3/GS3D and NL4/GS4D well clusters but decreased with depth at the GS1S/D cluster (fig. 8). No easily identified relation between alkalinity and depth was present at the VAP locations. Alkalinity values tended to be higher at the NL4/GS4D well cluster, particularly in well GS4D, than in the rest of the aquifer. Wells NL4 and GS4D are located directly downgradient of the landfill.

Sulfate concentrations in the samples from the seep and monitoring wells ranged from 14 to 190 mg/L (table 6, fig. 8). Sulfate concentrations were identical between sampling events at well NL3 but varied by more than a factor of six between sampling events at well NL4. Sulfate concentrations varied by less than a factor of three at the GS1S/D, NL3/GS3D and NL4/GS4D well clusters and showed no consistent patterns with depth during the July 2005 sampling. Sulfate concentrations were as much as an order of magnitude higher at well GS2S, located near the creek, than in the rest of the aquifer.

Calcium concentrations in the samples from the seep and monitoring wells ranged from 69 to 137 mg/L (table 6), making it the dominant cation in the aquifer. Calcium concentrations were similar between sampling events at wells NL3 and NL4. Calcium concentrations varied by less than a factor of two at the GS1S/D, NL3/GS3D and NL4/GS4D well clusters and showed no consistent patterns in concentration with depth or location during the July 2005 sampling (fig. 8).

Magnesium concentrations in the samples from the seep and monitoring wells ranged from about 31 to 81 mg/L and were similar between sampling events at wells NL3 and NL4 (table 6). Magnesium concentrations varied by less than a factor of two at the GS1S/D, NL3/GS3D, and NL4/GS4D well clusters and showed no consistent patterns in concentration with depth (fig. 8). Magnesium concentrations tended to be higher at the NL4/GS4D well cluster than in the rest of the aquifer. Wells NL4 and GS4D are located directly downgradient of the landfill. Calcium, magnesium, and alkalinity concentrations typically show a high degree of positive correlation, particularly if the data from wells NL4 and GS4D are not included in the analysis (table 5). This correlation indicates that these constitutes are derived

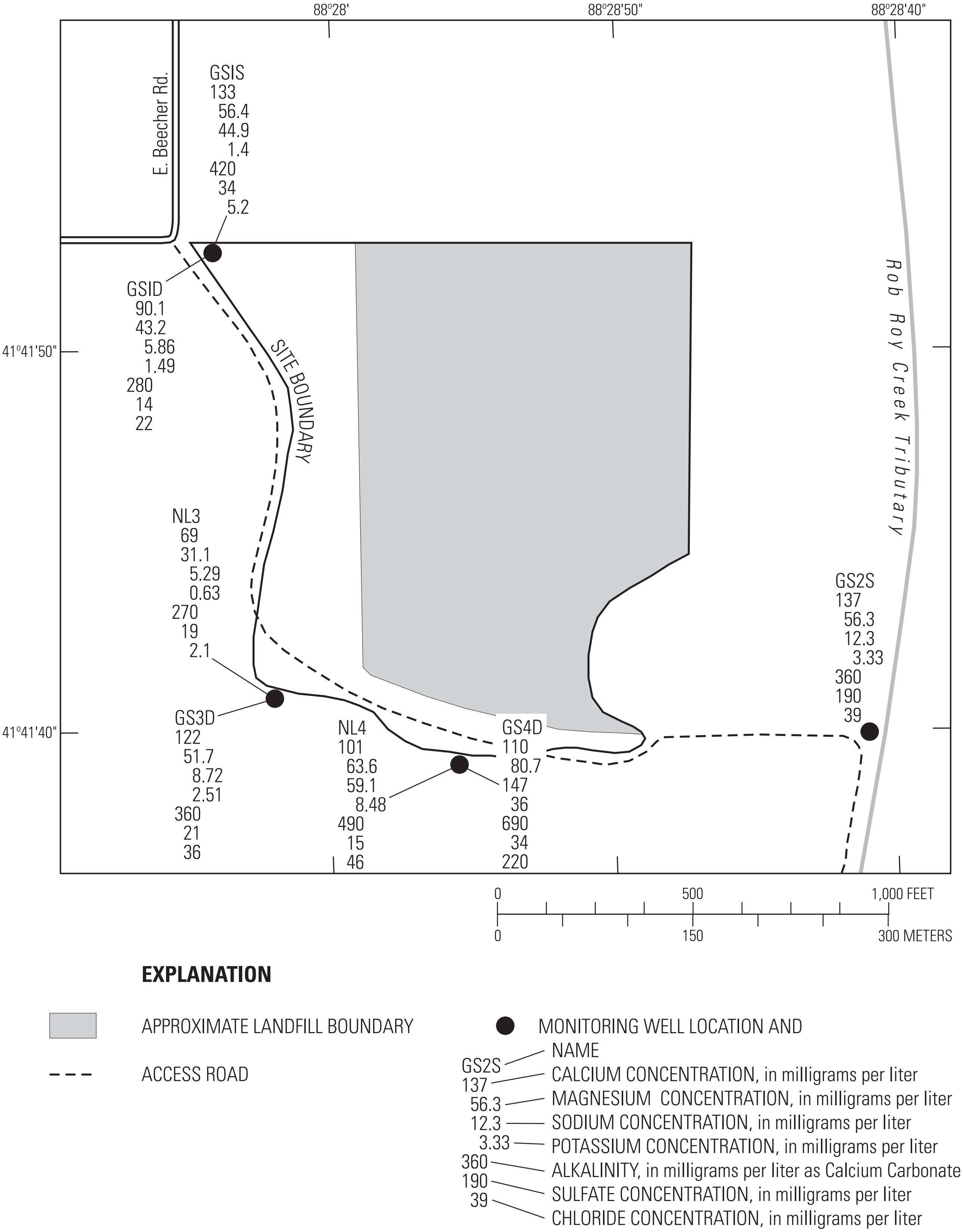


Figure 8. Concentration of major ions in samples from monitoring wells in the vicinity of the Nelson Landfill site, July 27, 2005.

primarily from the dissolution of limestone and dolomite minerals in the ambient aquifer.

Sodium concentrations in the samples from the monitoring wells ranged from about 5 to 147 mg/L during the July 2005 sampling (table 6, fig. 8). Sodium concentrations in wells NL4 and GS4D, particularly well GS4D, were higher than in the remaining wells, by an order of magnitude for some wells. Wells NL4 and GS4D are located directly downgradient of the landfill. Sodium concentrations varied by less than a factor of two at the NL3/GS3D cluster, but showed more variability at the GS1S/D and NL4/GS4D well clusters. Sodium concentrations showed no consistent patterns in concentration with depth during the July 2005 sampling.

Potassium concentrations in the samples collected from the monitoring wells in July 2005 ranged from less than 1 to 36 mg/L (table 6, fig. 8). Potassium concentrations were similar with depth at the GS1S/D well cluster but were substantially higher in the deeper part of the aquifer at the NL3/GS3D and NL4/GS4D clusters during the July 2005 sampling. Potassium concentrations in wells NL4 and GS4D, located downgradient of the landfill, were higher than in the remaining wells, by more than an order of magnitude in some wells. Potassium concentrations showed a high degree of positive correlation with chloride concentrations (table 5), indicating potassium in the ambient aquifer may be partly derived from road salts.

Nitrogen Compounds

Nitrogen-containing compounds occur primarily as ammonia (NH_3) or nitrate (NO_3^-) in natural water. The form of the compounds depends on the geochemical environment of the solution, with ammonia being the dominant form in more reducing (anoxic) water and nitrate predominating in more oxidizing water. Concentrations of nitrogen compounds greater than 5 mg/L as nitrogen are often associated with septic wastes (one of the wastes disposed of at the landfill) and the use of agricultural fertilizer.

Nitrate as nitrogen was detected at a concentration of 3.8 mg/L in the sample from shallow upgradient well GS1S (table 6, fig. 9), below the USEPA MCL of 10 mg/L. Nitrogen in the form of ammonia was detected at the seep and in the samples from wells GS1D, GS2S, NL4, and GS4D at concentrations ranging from 0.51 to nearly 32 mg/L. Ammonia concentrations were substantially higher in wells NL4 and GS4D, and particularly well GS4D, than in the remaining wells. These wells are located directly downgradient of the landfill. There is no USEPA or Illinois drinking-water standard for ammonia.

The form of the nitrogen in this aquifer is affected by the presence of oxygen and the ORP in the aquifer. Nitrate predominates in the oxygenated, higher ORP water in the shallow part of the aquifer at well GS1S. Ammonia predominates in the anoxic, lower ORP water more than about 25 ft below the top of the aquifer at wells GS1D, GS2S, GS3D, NL4, and GS4D (table 1), indicating that the processes responsible for the conversion of nitrate to ammonia in the ambient part of the glacial drift aquifer are occurring in a zone within the upper 15-25 ft of the aquifer. Nitrogen compounds were not detected in the oxygenated, higher ORP water at well NL3. The presence of detectable concentrations of nitrogen compounds in shallow upgradient well GS1S indicates that the surface application of agricultural fertilizers may affect ambient water quality in the glacial drift aquifer. However, nitrate as nitrogen was detected in only one well and at a concentration less than half its MCL.

Ammonia as nitrogen was detected at concentrations of about 0.60 mg/L in the creek at all sampling locations (Gordon Stevens, Civil and Environmental Consultants Inc., written commun., 2005). Nitrate as nitrogen was not detected at site SS1 or SS2, but was detected at a concentration of 1.34 mg/L at site SS3. Concentrations of nitrogen compounds in the creek do not appear to be affected by the landfill.

Organic Compounds

Organic compounds (pesticides and VOCs) were not detected in samples collected from the seep, monitoring wells, or the VAP locations during this investigation (tables 3, 7, 8). VOCs were not detected in any of the samples collected from the creek (Gordon Stevens, Civil and Environmental Consultants Inc., written commun., 2005). VOCs were among the primary types of potential contaminants of concern at the site and their absence from the wells and the VAP sample locations, coupled with the apparent stability in flow directions through the years, indicates that capping of the landfill and natural attenuation processes have reduced the size of the VOC plume in the glacial drift aquifer in this area or eliminated it altogether. The absence of detectable concentrations of pesticide compounds in the monitoring wells also indicates that the agricultural application of these compounds does not appear to have a substantial effect on ambient water quality in the glacial drift aquifer in this area. However, the pesticide compounds analyzed for as part of this investigation are used to control insects (insecticides). Pesticide compounds used to control plants (herbicides) were not included in these analyses. The presence of herbicides or pesticide degradates in the glacial drift aquifer has not been determined.

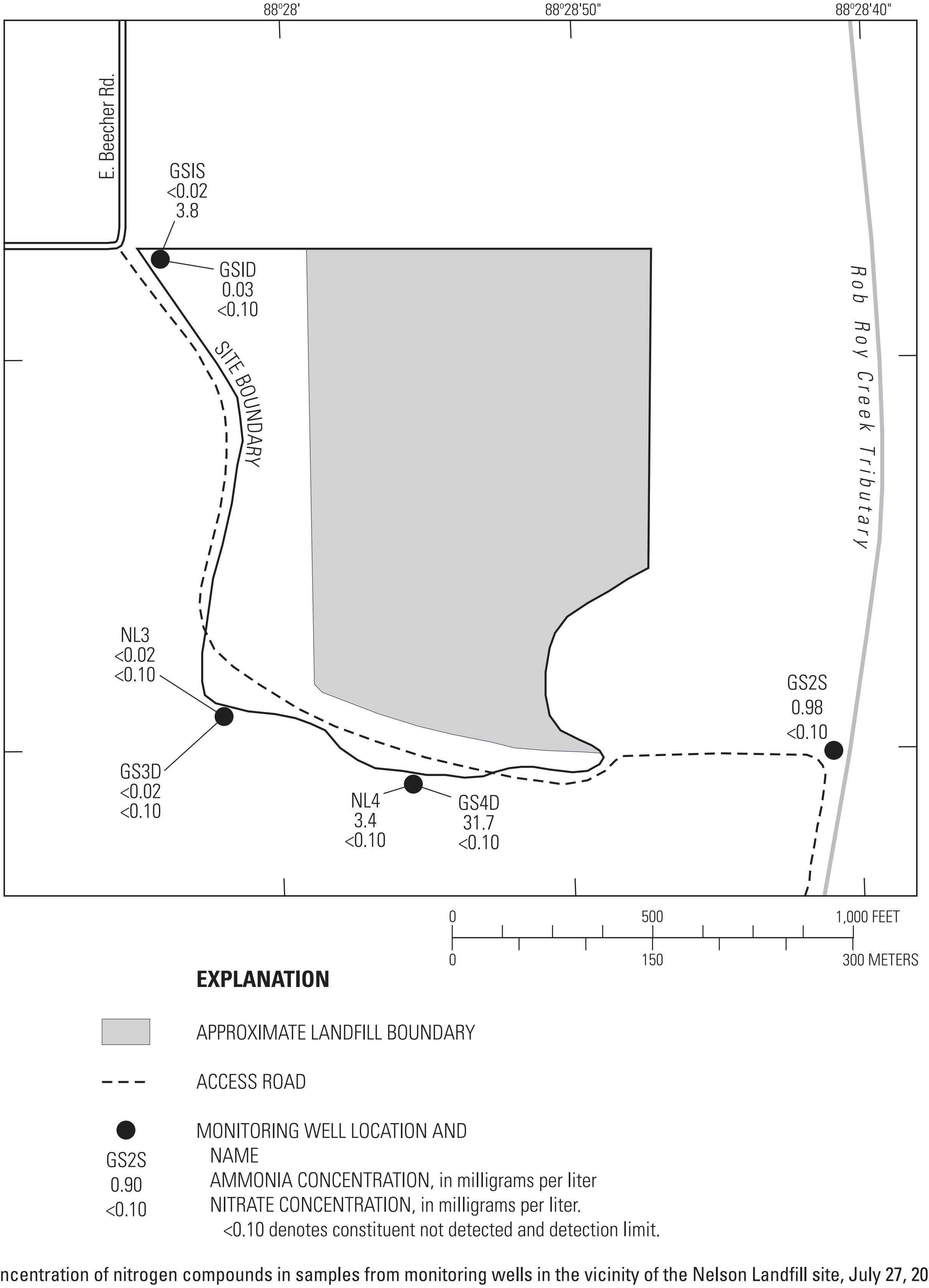


Figure 9. Concentration of nitrogen compounds in samples from monitoring wells in the vicinity of the Nelson Landfill site, July 27, 2005.

Table 7. Concentration of volatile organic compounds in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site.

[dup, duplicate sample; MCL, U.S. Environmental Protection Agency Maximum Contaminant Level; ug/L, micrograms per liter; <, denotes constituent not detected and detection limit; na, not available]

	Locations sampled on March 15, 2005				
Constituent	NL3	NL4	NL4 (dup)	Seep	MCL
Acetone (ug/L)	<1	<1	<1	<1	na
Benzene (ug/L)	<1	<1	<1	<1	5
Bromodichloromethane (ug/L)	<1	<1	<1	<1	na
Bromoform (ug/L)	<1	<1	<1	<1	na
Bromomethane (ug/L)	<1	<1	<1	<1	na
2-Butanone (ug/L)	<1	<1	<1	<1	na
Carbon disulfide (ug/L)	<5	<5	<5	<5	na
Carbon tetrachloride (ug/L)	<1	<1	<1	<1	5
Chlorobenzene (ug/L)	<1	<1	<1	<1	100
Chloroethane (ug/L)	<1	<1	<1	<1	na
Chloroform (ug/L)	<1	<1	<1	<1	na
Chloromethane (ug/L)	<1	<1	<1	<1	na
Dibromochloromethane (ug/L)	<1	<1	<1	<1	na
1,1-Dichloroethane (ug/L)	<1	<1	<1	<1	na
1,2-Dichloroethane (ug/L)	<1	<1	<1	<1	5
1,1-Dichloroethene (ug/L)	<1	<1	<1	<1	7
trans-1,2-Dichloroethene (ug/L)	<1	<1	<1	<1	100
cis-1,2-Dichloroethene (ug/L)	<1	<1	<1	<1	70
1,2-Dichloropropane (ug/L)	<1	<1	<1	<1	5
cis-1,3-Dichloropropene (ug/L)	<1	<1	<1	<1	na
trans-1,3-Dichloropropene (ug/L)	<1	<1	<1	<1	na
Ethylbenzene (ug/L)	<1	<1	<1	<1	700
2-Hexanone (ug/L)	<10	<10	<10	<10	na
Methylene chloride (ug/L)	<5	<5	<5	<5	na
Methyl tert-butyl ether (ug/L)	<5	<5	<5	<5	na
4-Methyl-2-pentanone (ug/L)	<10	<10	<10	<10	na
Styrene (ug/L)	<1	<1	<1	<1	100
1,1,2,2-Tetrachloroethane (ug/L)	<1	<1	<1	<1	na
Tetrachloroethene (ug/L)	<1	<1	<1	<1	5
Toluene (ug/L)	<1	<1	<1	<1	1,000
1,1,1-Trichloroethane (ug/L)	<1	<1	<1	<1	200
1,1,2-Trichloroethane (ug/L)	<1	<1	<1	<1	5
Trichloroethene (ug/L)	<1	<1	<1	<1	5
Vinyl acetate (ug/L)	<1	<1	<1	<1	na
Vinyl chloride (ug/L)	<1	<1	<1	<1	2
m,p-xylene (ug/L)	<2	<2	<2	<2	10,000 for all xylene isomers
o-xylene (ug/L)	<1	<1	<1	<1	

Table 7. Concentration of volatile organic compounds in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site—continued.

[dup, duplicate sample; MCL, U.S. Environmental Protection Agency Maximum Contaminant Level; ug/L, micrograms per liter; <, denotes constituent not detected and detection limit; na, not available]

	Locations sampled on July 27, 2005							
Constituent	GS1S	GS1D	GS2S	NL3	GS3D	NL4	GS4D	MCL
Acetone (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Benzene (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
Bromodichloromethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Bromoform (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Bromomethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
2-Butanone (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Carbon disulfide (ug/L)	<5	<5	<5	<5	<5	<5	<5	na
Carbon tetrachloride (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
Chlorobenzene (ug/L)	<1	<1	<1	<1	<1	<1	<1	100
Chloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Chloroform (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Chloromethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Dibromochloromethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
1,1-Dichloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
1,2-Dichloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
1,1-Dichloroethene (ug/L)	<1	<1	<1	<1	<1	<1	<1	7
trans-1,2-Dichloroethene (ug/L)	<1	<1	<1	<1	<1	<1	<1	100
cis-1,2-Dichloroethene (ug/L)	<1	<1	<1	<1	<1	<1	<1	70
1,2-Dichloropropane (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
cis-1,3-Dichloropropene (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
trans-1,3-Dichloropropene (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Ethylbenzene (ug/L)	<1	<1	<1	<1	<1	<1	<1	700
2-Hexanone (ug/L)	<10	<10	<10	<10	<10	<10	<10	na
Methylene chloride (ug/L)	<5	<5	<5	<5	<5	<5	<5	na
Methyl tert-butyl ether (ug/L)	<5	<5	<5	<5	<5	<5	<5	na
4-Methyl-2-pentanone (ug/L)	<10	<10	<10	<10	<10	<10	<10	na
Styrene (ug/L)	<1	<1	<1	<1	<1	<1	<1	100
1,1,2,2-Tetrachloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Tetrachloroethene (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
Toluene (ug/L)	<1	<1	<1	<1	<1	<1	<1	1,000
1,1,1-Trichloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	200
1,1,2-Trichloroethane (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
Trichloroethene (ug/L)	<1	<1	<1	<1	<1	<1	<1	5
Vinyl acetate (ug/L)	<1	<1	<1	<1	<1	<1	<1	na
Vinyl chloride (ug/L)	<1	<1	<1	<1	<1	<1	<1	2
m,p-xylene (ug/L)	<2	<2	<2	<2	<2	<2	<2	10,000 for all xylene isomers
o-xylene (ug/L)	<1	<1	<1	<1	<1	<1	<1	

Table 8. Concentration of pesticide compounds in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site.

[dup, duplicate sample; ug/L, micrograms per liter; <, denotes constituent not detected and detection limit; MCL, U.S. Environmental Protection Agency Maximum Contaminant Level; na, none available]

	Locations sampled on March 15, 2005				
Constituent	NL3	NL4	NL4 (dup)	Seep	MCL
Aldrin (ug/L)	<0.51	<0.51	<0.51	<0.51	na
alpha-BHC (ug/L)	<.51	<.51	<.51	<.51	na
beta-BHC (ug/L)	<.51	<.51	<.51	<.51	na
gamma-BHC (lindane) (ug/L)	<.51	<.51	<.51	<.51	na
delta-BHC (ug/L)	<.51	<.51	<.51	<.51	na
alpha-Chlordane (ug/L)	<.51	<.51	<.51	<.51	na
gamma-Chlordane (ug/L)	<.51	<.51	<.51	<.51	na
Chlordane (ug/L)	<.10	<.10	<.10	<.10	2
4,4’-DDD (ug/L)	<.51	<.51	<.51	<.51	na
4,4’-DDE (ug/L)	<.51	<.51	<.51	<.51	na
4,4’-DDT (ug/L)	<.51	<.51	<.51	<.51	na
Dieldrin (ug/L)	<.51	<.51	<.51	<.51	na
Endosulfan I (ug/L)	<.51	<.51	<.51	<.51	na
Endosulfan II (ug/L)	<.51	<.51	<.51	<.51	na
Endosulfan sulfate (ug/L)	<.51	<.51	<.51	<.51	na
Endrin (ug/L)	<.51	<.51	<.51	<.51	2
Endrin aldehyde (ug/L)	<.51	<.51	<.51	<.51	2
Endrin ketone (ug/L)	<.51	<.51	<.51	<.51	2
Heptachlor (ug/L)	<.51	<.51	<.51	<.51	.4
Heptachlor epoxide (ug/L)	<.51	<.51	<.51	<.51	.4
Methoxychlor (ug/L)	<.10	<.10	<.10	<.10	40
Toxaphene (ug/L)	<.51	<.51	<.51	<.51	3

	Locations sampled on July 27, 2005							
Constituent	GS1S	GS1D	GS2S	NL3	GS3D	NL4	GS4D	MCL
Aldrin (ug/L)	<0.48	<0.48	<0.48	<0.48	<0.48	<0.48	<0.48	na
alpha-BHC (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
beta-BHC (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
gama-BHC (lindane) (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
delta-BHC (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
alpha-Chlordane (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
gamma-Chlordane (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Chlordane (ug/L)	<.096	<.096	<.096	<.096	<.096	<.096	<.096	2
4,4’-DDD (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
4,4’-DDE (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
4,4’-DDT (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Dieldrin (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Endosulfan I (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Endosulfan II (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Endosulfan sulfate (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	na
Endrin (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	2
Endrin aldehyde (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	2
Endrin ketone (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	2
Heptachlor (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	.4
Heptachlor epoxide (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	.4
Methoxychlor (ug/L)	<.096	<.096	<.096	<.096	<.096	<.096	<.096	40
Toxaphene (ug/L)	<.48	<.48	<.48	<.48	<.48	<.48	<.48	3

Table 9. Concentration of metals and cyanide in water from monitoring wells and seeps in the vicinity of the Nelson Landfill site.

[dup, duplicate sample; mg/L, micrograms per liter; <, denotes constituent not detected and detection limit; MCL, U.S. Environmental Protection Agency Maximum Contaminant Level; SMCL, U.S. Environmental Protection Agency Secondary Maximum Contaminant Level; na, none available]

	Locations sampled on March 15, 2005					
Constituent	NL3	NL4	NL4 (dup)	Seep	MCL	SMCL
Aluminum (mg/L)	0.142	<0.050	<0.050	42.6	na	0.05 to 02
Antimony (mg/L)	<.050	<.050	<.050	<.050	na	na
Arsenic (mg/L)	<.020	<.020	<.020	<.020	0.05	na
Barium (mg/L)	.038	.12	.121	.38	2	na
Beryllium (mg/L)	<.005	<.005	<.005	<.005	.04	na
Boron (mg/L)	<.050	.234	.241	<.050	na	na
Cadmium (mg/L)	<.005	<.005	<.005	<.005	.005	na
Chromium (mg/L)	.011	<.010	<.010	.049	.1	na
Cobalt (mg/L)	<.030	<.030	<.030	<.030	na	na
Copper (mg/L)	<.010	<.010	<.010	.039	1,300	na
Iron (mg/L)	.219	<.010	.076	36.8	na	.3
Lead (mg/L)	<.050	<.050	<.050	<.050	na	na
Manganese (mg/L)	.012	.937	.952	.803	na	.05
Molybdenum (mg/L)	<.050	<.050	<.050	<.050	na	na
Nickel (mg/L)	<.010	.015	.016	.041	na	na
Selenium (mg/L)	<.020	<.020	<.020	<.020	.05	na
Silver (mg/L)	<.010	<.010	<.010	<.010	na	.1
Strontium (mg/L)	.056	.163	.166	.172	na	na
Tin (mg/L)	<.050	<.050	<.050	<.050	na	na
Titanium (mg/L)	<.050	<.050	<.050	.978	na	na
Vanadium (mg/L)	<.050	<.050	<.050	.076	na	na
Zinc (mg/L)	<.010	<.010	<.010	.21	na	na
Cyanide (mg/L)	<.005	<.005	<.005	<.005	.2	na
Mercury (mg/L)	<.001	<.001	<.001	<.001	.002	na

	Locations sampled on July 27, 2005								
Constituent	GS1S	GS1D	GS2S	NL3	GS3D	NL4	GS4D	MCL	SMCL
Aluminum (mg/L)	<0.050	<0.050	<0.050	0.059	<0.050	<0.050	<0.050	na	0.05 to 02
Antimony (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Arsenic (mg/L)	<.020	<.020	<.020	<.020	<.020	<.020	<.020	0.05	na
Barium (mg/L)	.067	.076	.180	.039	.081	.116	.713	2	na
Beryllium (mg/L)	<.005	<.005	<.005	<.005	<.005	<.005	<.005	.04	na
Boron (mg/L)	<.050	<.050	<.050	<.050	.057	.237	.677	na	na
Cadmium (mg/L)	<.005	<.005	<.005	<.005	<.005	<.005	<.005	.005	na
Chromium (mg/L)	<.010	<.010	<.010	<.010	<.010	<.010	<.010	.1	na
Cobalt (mg/L)	<.030	<.030	<.030	<.030	<.030	<.030	<.030	na	na
Copper (mg/L)	<.010	<.010	<.010	<.010	<.010	<.010	<.010	1,300	na
Iron (mg/L)	<.050	1.34	2.81	.092	.234	.055	2.26	na	.3
Lead (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Manganese (mg/L)	.043	.032	.199	<.010	.211	.901	.217	na	.05
Molybdenum (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Nickel (mg/L)	<.010	<.010	<.010	<.010	<.010	<.010	.012	na	na
Selenium (mg/L)	<.020	<.020	<.020	<.020	<.020	<.020	<.020	.05	na
Silver (mg/L)	<.010	<.010	<.010	<.010	<.010	<.010	<.010	na	.1
Strontium (mg/L)	.056	.144	.198	.057	.193	.160	.316	na	na
Tin (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Titanium (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Vanadium (mg/L)	<.050	<.050	<.050	<.050	<.050	<.050	<.050	na	na
Zinc (mg/L)	.014	<.010	<.010	<.010	.013	<.010	<.010	na	na
Cyanide (mg/L)	<.005	<.005	<.005	<.005	<.005	<.005	<.005	.2	na
Mercury (mg/L)	<.001	<.001	<.001	<.001	<.001	<.001	<.001	.002	na

Metals and Cyanide

Aluminum, chromium, copper, barium, boron, iron, manganese, nickel, strontium, titanium, vanadium, and zinc were detected in at least one of the ground-water samples (well or seep) collected for this investigation (table 9). Only barium, boron, iron, manganese, and strontium were detected in the samples from three or more wells. None of these constituents were detected at a concentration above its MCL for drinking water, although the Secondary Maximum Contaminant Level (SMCL) was exceeded for aluminum, iron, and manganese in at least one sample (U.S. Environmental Protection Agency, 2004). The SMCL is a non-enforceable standard based on aesthetic considerations such as color, odor, and taste. None of the remaining metal constituents were detected in any sample collected for this investigation.

Aluminum was detected in the sample from well NL3 during the March and July 2005 sampling at a concentration of 0.142 and 0.059 mg/L, respectively (table 9). Aluminum was not detected in any other sample collected for this investigation. There is no MCL for aluminum, but concentrations at well NL3 are within the range of the USEPA's SMCL for drinking water from 0.05 to 0.2 mg/L

Concentrations of barium in the ground-water samples collected for this investigation ranged from 0.039 mg/L in well NL3 to 0.713 mg/L in well GS4D (table 9, fig. 10). Barium concentrations were higher in wells NL4 and GS4D, located directly downgradient of the landfill, and well GS2S located near the creek, than in the remaining wells. Concentrations of barium in the creek ranged from about 0.77 mg/L at sites SS1 and SS3 to 0.102 mg/L at site SS2. The barium concentration in the creek at site SS2 was similar to the concentration in ground water at well GS2S (0.18 mg/L).

Concentrations of boron in the ground-water samples collected for this investigation ranged from below the detection limit in wells GS1S, GS1D, GS2S, and NL3 to 0.677 mg/L in well GS4D (table 9). Boron concentrations were consistent between sampling rounds at wells NL3 and NL4. Boron concentrations were substantially higher in wells NL4 and GS4D, located directly downgradient of the landfill, than in the remaining wells.

Chromium was detected in the sample from well NL3 during the March 2005 sampling at a concentration of 0.011 mg/L, approximately on order of magnitude below its MCL (table 9). Chromium was not detected in any other sample collected for this investigation, including the sample from well NL3 collected in during July 2005.

Copper was detected in the seep sample during the March 2005 sampling at a concentration of 0.039 mg/L (table 9). Copper was not detected in any other sample collected for this investigation.

Concentrations of iron in the ground-water samples collected from the monitoring wells for this investigation ranged from below the detection limit in well GS1S and one of the samples collected from well NL4 to 2.81 mg/L in well GS2S (table 9, fig. 10). These concentrations are more than an order of magnitude lower than the iron concentration in the seep sample. Iron concentrations showed moderate variation within and between sampling rounds at wells NL3 and NL4 and increased with depth at the GS1S/1D, NL3/GS3D, and NL4/GS4D well clusters. Although the value of r^2 did not exceed 0.70, iron concentrations tended to decrease as ORP and DO values increase, indicating that iron concentrations are affected by geochemical conditions in the aquifer. There is no MCL for iron, but many of these samples exceed the SMCL for drinking water of 0.3 mg/L, indicating that water-supply systems that utilize the glacial drift aquifer (particularly the deeper part) may need to treat that water to remove iron.

Concentrations of manganese in the ground-water samples collected for this investigation ranged from below the detection limit in well NL3 to more than 0.9 mg/L in well NL4 (table 9, fig. 10). Manganese concentrations were consistent between sampling events at wells NL3 and NL4 and showed no consistent patterns with depth. There is no MCL for manganese, but many of these samples exceed the SMCL for drinking water of 0.05 mg/L, indicating that water-supply systems that utilize the glacial drift aquifer may need to treat that water to remove manganese.

Concentrations of nickel in the ground-water samples collected for this investigation ranged from below the detection limit in most of the wells to 0.041 mg/L at the seep (table 9). Nickel concentrations were moderately consistent between sampling events at wells NL3 and NL4. Nickel concentrations in wells NL4 and GS4D, located directly downgradient of the landfill, were higher than in the remaining samples collected from the monitoring wells.

Concentrations of strontium in the ground-water samples collected for this investigation ranged from about 0.056 mg/L in wells GS1S and NL3 to 0.316 mg/L at well GS4D (table 9, fig. 10). Strontium concentrations were consistent between sampling events at wells NL3 and NL4 and increased with depth at the GS1S/1D, NL3/GS3D and NL4/GS4D well clusters. Strontium concentrations in well GS4D, located directly downgradient of the landfill, were slightly higher than in the remaining samples collected from the monitoring wells. Strontium concentrations showed a high degree of positive correlation with chloride concentrations (table 5) regardless of whether wells NL4 and GS4D were included in the analysis, indicating strontium may be partly derived from road salts (Granato, 1996).

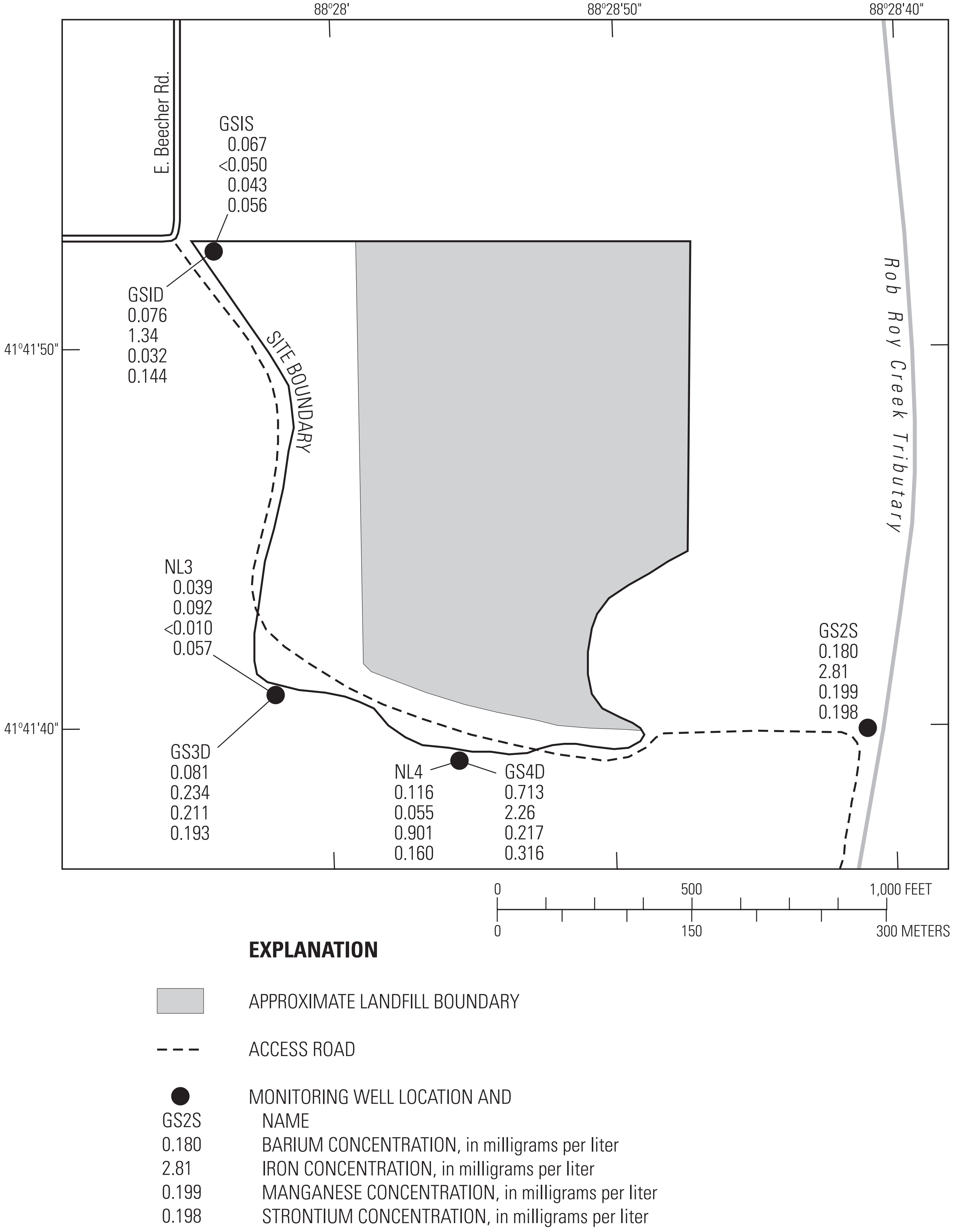


Figure 10. Concentration of selected metals in samples from monitoring wells in the vicinity of the Nelson Landfill site, July 27, 2005.

Titanium was detected in the seep sample during the March 2005 sampling at a concentration of 0.978 mg/L (table 9). Titanium was not detected in any other sample collected for this investigation.

Vanadium was detected in the seep sample during the March 2005 sampling at a concentration of 0.076 mg/L (table 9). Vanadium was not detected in any other sample collected for this investigation.

Zinc was detected in the seep during the March 2005 sampling at a concentration of 0.21 mg/L and was detected during the July 2005 sampling in wells GS1S and GS3D at concentrations of 0.014 and 0.013 mg/L, respectively (table 9). Zinc concentrations were consistent at wells NL3 and NL4 between sampling events and showed no patterns with depth or location.

Cyanide was not detected in any sample collected as part of this investigation (table 9). Cyanide was not detected in any of the samples collected from the creek (Gordon Stevens, Civil and Environmental Consultants Inc., written commun., 2005). The absence of detectable concentrations of cyanide from wells where it had been detected during a prior investigation (Ecology and Environment, Inc., 1991) is further indication that natural attenuation processes have reduced cyanide concentrations in this aquifer in the years since the previous investigations were performed.

SUMMARY AND CONCLUSIONS

The U.S. Geological Survey, in cooperation with the United City of Yorkville and the U.S. Environmental Protection Agency, characterized the geology, hydrology, and water quality of the glacial drift aquifer in the Newark Bedrock Valley in the vicinity of the Nelson Landfill site in Yorkville, Illinois. The investigation was performed to evaluate the potential for development of the aquifer as a water-supply source and the effect of land use, including landfill operations, on the quality of water in the aquifer.

There are two geologic deposits of concern in the study area-a shallow deposit composed primarily of silt and clay till, and a deeper sand-and-gravel deposit. The sand-and-gravel deposit constitutes the glacial drift aquifer, which is from about 35 to greater than 85 feet thick and has a geometric mean horizontal hydraulic conductivity of 9.2 feet per day. The aquifer appears to be sufficiently permeable for use as a source of public-water supply, but because the geology is heterogeneous, the development of the aquifer for water supply would require site-specific studies to optimally locate any public-supply well or well field. The overall direction of flow in the aquifer is from north to south, with a slight west to east component toward the tributary to Rob Roy Creek. The creek appears to be in good hydraulic connection with the glacial drift aquifer so that the flow of the creek could be affected (reduced) by any future high-capacity pumping from the aquifer. Ground-water velocity and the volume of ground water flowing through the aquifer were calculated to be about 7.7 X 10^{-2} feet per day and 2,300 cubic feet per day, respectively.

There is no indication that the water in the aquifer in the vicinity of the site contains concentrations of any constituent high enough to represent a threat to human health or the environment. Contaminants of potential concern detected during ground-water sampling in the mid-1990s, including volatile organic compounds, pesticide compounds, and cyanide, were not detected in samples of ground water collected during this investigation. No constituent, including nitrate as nitrogen and metals, was detected above its U.S. Environmental Protection Agency Maximum Contaminant Level. However, iron, manganese, and aluminum were detected at concentrations above their Secondary Maximum Contaminant Level in at least one sample, indicating that this water may require treatment if used for public-water supply. There is no indication that water quality in the creek is being degraded by the inflow of ground water.

Of the major ions, nitrogen compounds, and metals detected in the ground water beneath the study area, the highest concentration of magnesium, sodium, potassium, alkalinity, chloride, ammonia as nitrogen, barium, boron, manganese, nickel, and strontium were detected in the sample from two wells located directly downgradient of the landfill. Nitrogen as nitrate also was detected in a shallow well located upgradient of the landfill at a concentration below its Maximum Contaminant Level, indicating that agricultural practices may be affecting the concentrations of nitrogen compounds in the glacial drift aquifer.

Oxidation-reduction conditions in the glacial drift aquifer become more reducing with depth. This change is reflected by the type and concentration of the nitrogen compounds and the concentration of dissolved oxygen and iron in the glacial drift aquifer. Concentrations of some of the major ions and metals may be affected by dissolution of carbonate minerals and possibly road salts.

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