

# Water Chemistry Near the Closed Norman Landfill, Cleveland County, Oklahoma, 1995

# Water-Resources Investigations Report 00-4238



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

**Cover**: Photograph of the top of the Norman Landfill with the Canadian River in the background.



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By Jamie L. Schlottmann

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## **Conversion Factors and Datum**

Multiply	Ву	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m <sup>2</sup> )	1.196	square yard (yd <sup>2</sup> )
hectare (ha)	2.471	acre
hectare (ha)	0.003861	square mile (mi <sup>2</sup> )
	Flow rate	
cubic meter per second (m <sup>3</sup> /s)	35.31	cubic foot per second ( $ft^3/s$ )
liter per second (L/s)	15.85	gallon per minute (gal/min)

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

°F = (1.8 x °C) + 32

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

Elevation, as used in this report, refers to distance above or below sea level.

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

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

# **Local Identifier**

The well locations are specified by latitude and longitude to the nearest second and by a local identifier, which is based on the public land survey. The local identifier includes the township and range followed by the section and a series of letters that designate the quarter-section subdivisions from largest to smallest. The order of the quarter-section subdivisions is opposite of that used by the public land survey. A sequence number is added to make the local identifier unique in the U.S. Geological Survey data base. As illustrated in the following diagram, the public land survey description of the site indicated by the dot is NW 1/4 NE 1/4 SE 1/4 sec. 18, T. 08 N., R. 02 W., which is denoted by the local identifier 08N-02W-18 DAB. If the sequence number is 5, the complete identifier is 08N-02W-18 DAB 5.



# Water Chemistry Near the Norman Landfill, Cleveland County, Oklahoma, 1995

By Jamie L. Schlottmann

## Abstract

The Norman Landfill was selected for study as part of the U.S. Geological Survey Toxic Substances Hydrology Program in 1994. The landfill is located south of the City of Norman on alluvial deposits of the Canadian River. Type of waste deposited in the landfill from 1922 to 1973 was largely unrestricted and may include substances now recognized as hazardous. Dissolved and suspended substances leached from wastes in the closed and capped landfill are now in ground water extending toward the Canadian River as a plume of leachate.

Water samples were collected from two stock wells, one domestic well, temporary drive-point wells, the Canadian River, and a small intermittent stream hydraulically downgradient of the capped landfill known as the slough. Most constituent concentrations were greater in ground water downgradient from the capped landfill than in background ground water and were greater in the slough than in the Canadian River. Concentrations of most constituents in the Canadian River, other than sulfate, manganese, and iron, were similar to concentrations in background ground water.

Some constituents measured in ground-water for this investigation are potential indicators of leachate contamination. Potential indicators that could be used to differentiate leachate contaminated water from uncontaminated ground water of the alluvial aquifer include specific conductance, chloride, alkalinity, dissolved organic carbon, boron, and  $\delta D$ . Specific conductance and chloride were greater in water from wells downgradient of the landfill than water from background wells. Dissolved organic carbon and boron also were greater in the leachate contaminated ground water than in background ground water.

## Introduction

The U.S. Geological Survey began a multidisciplinary investigation in 1994 at the Norman Landfill (fig. 1), as part of the Toxic Substances Hydrology program, in collaboration with scientists at the University of Oklahoma, Oklahoma State University, and the U.S. Environmental Protection Agency. The contamination of the shallow alluvial aquifer at the Norman Landfill provides an opportunity to study the spatial variability of biogeochemical processes and the resulting effects on the fate of degradable contaminants in the leachate plume. The emphasis of this multidisciplinary research project is on developing an understanding of the processes controlling contaminant distribution and migration. Results of the investigation of water chemistry, described in this report, will provide basic information that may be used to direct future research on biological, chemical, mineral, and hydrologic processes. The results presented describe the quality of ground and surface water at the site during the period between October 25, 1995, and December 7, 1995.

The Norman Landfill has been identified in Suflita and others (1988) as a source of dissolved organic and inorganic compounds in ground water that is known as leachate and geophysical measurements have shown that a leachate plume extends southwest from the landfill toward the Canadian River. Robertson and others (1974) found more than 40 industrial and commercial organic chemicals leaching from the landfill, including p-cresol, phthalates, and volatile fatty acids. Anderson (1972) determined that unfiltered water samples from within the landfill contained greater concentrations of aluminum, chromium, copper, iron, lead, and manganese than did samples of ground water from the surrounding alluvial aquifer. Surface geophysical measurements made at the Norman Landfill site during January and February 1995 (Lucius and Bisdorf, 1995, p. 13) showed an area of increased conductivity attributed to a leachate plume, extending toward the Canadian River, between the west cell of the capped landfill and a small intermittent stream southwest of the capped landfill referred to as the slough. These measurements did not indicate clearly whether the plume extended beyond the slough.

Investigations of microbial activity and microbial degradation of organic matter in ground water between the slough and the capped landfill have indicated anaerobic conditions (Suflita and others, 1988). In anaerobic conditions organisms can use organic chemicals or inorganic anions such as nitrate, sulfate, or carbonate as electron acceptors. Suflita and others (1988) report that sulfate reduction was the dominant process southwest of the west cell and methanogenisis was the dominant process south of the west cell of the capped landfill (fig. 1).

Characteristics of landfill leachate have been described in investigations of other landfills (Baedecker and Back, 1979, Nicholson and others, 1983, Christensen and Kjeldsen, 1989, and Hackley and others, 1996). These investigations indicated that concentrations of dissolved sulfide, methane, major ions, organic nitrogen, ammonia, iron, manganese, organic carbon, and zinc commonly are greater in leachate-contaminated water



Figure 1. Location of study area.

than in uncontaminated ground water. These inorganic constituents may be leached from refuse or dissolved from aquifer minerals. Chloride generally is a conservative indicator of leachate contamination if other significant sources of chloride are not present. Degradation of organic compounds in leachate by bacteria depletes oxygen in the water and produces carbon dioxide, increasing alkalinity and decreasing pH. Conditions commonly are reducing in the most-concentrated part of an established plume. Oxygen content increases near the water table and the perimeter of the plume. Baedecker and Back (1979) and Hackley and others (1996) report that deuterium and <sup>13</sup>carbon stable isotopes tend to be more enriched in leachatecontaminated water.

### **Purpose and Scope**

This report describes results of an initial investigation of the ground- and surface-water chemistry at the Norman Landfill. Digitized aerial photographs show the development of the landfill and changes in river-channel position. This report includes maps that show the geology of the area and sampling locations. Tables list sample collection and preservation methods and results of the geochemical analyses. Stiff diagrams illustrate the major-ion composition of the water and other graphs show compositional variations between sample types. Selected constituents that indicate the presence of leachate in the ground water are presented.

## **Description of the Norman Landfill Area**

The Norman Landfill is located south of the City of Norman on alluvial deposits of the Canadian River (fig. 1). The landfill includes regions where waste was buried before 1970 and a clay-capped sanitary landfill where waste was deposited from 1970 to 1985. The total landfill area is about 314,000 square meters. The capped landfill comprises about 186,000 square meters of the total landfill area and rises to an elevation 12 to 15 meters above the alluvium. The capped landfill includes two cells, the east cell and the west cell (fig. 1), separated by a buried wastewater discharge line from a wastewater treatment plant 750 meters to the north. The relatively flat topped cells are vegetated with Bermuda grass.

The Canadian River flows southeast and currently (1998) lies about 600 meters southwest of the landfill. The river is separated from the landfill by a relatively flat area, with low sand dunes vegetated with small willow and cottonwood trees, shrubs, and grasses. The location of the river channel varies over time. In the mid-1980s the river flowed along the base of the landfill. The river flooded in 1986 and the channel shifted 600 meters to the west.

The slough lies 50 to 100 meters southwest of the landfill and flows to the southeast. The slough is probably a previous location of the main river channel. The slough does not have an obvious source and is presumed to be fed by ground-water discharge and precipitation. Discharge from the Norman Wastewater Treatment Plant flows into the slough at the south end of the west cell and the combined water flows south to the Canadian River (fig. 1).

The Canadian River alluvium surrounding the capped landfill slopes gently towards the river at about 1 meter per 500 meters. The northern extent of the alluvium is 670 to 945 meters north of the capped landfill. Land surface elevation increases by about 7.6 meters over a distance of 30 to 60 meters in transition from the alluvium to a river terrace.

## Hydrogeologic setting

The geologic setting of the landfill is characterized by moderately permeable alluvial and terrace deposits with a shallow water table that overlie low permeability rocks of Permian age. The Canadian River alluvial and the low terrace deposits of Quaternary age, on which the landfill is located, are the youngest geologic units (fig. 2). The alluvium has been continually eroded, transported, and deposited by the river and is about 12 meters thick in the landfill area (Stacy, 1961, p. 46-47). The alluvial and low terrace deposits consist of unconsolidated quartz sand, silt, mud, and lenticularly bedded gravel. Clay in the mud is dominantly smectitic (George Breit, U.S. Geological Survey, oral commun., 1997). The alluvial and low terrace deposits are moderately permeable. Hydraulic conductivity is estimated to range from 8.4x10<sup>-7</sup> to 2.8x10<sup>-4</sup> meters per second (Scholl and Christenson, 1998, p. 18). Regional ground-water flow is toward the Canadian River. Northeast of the slough, the hydraulic gradient is toward the southwest (fig. 3). South of the slough the hydraulic gradient becomes southeast. The hydraulic gradient is about 1.4 meters per kilometer south of the capped landfill (Scholl and Christenson, 1998, p. 1-3). Depth to ground water in the deposits varies seasonally and ranges from at land surface to about 4 meters. The alluvial and low terrace deposits unconformably overlie red-bed siltstones, mudstones, and sandstones of the Hennessey Group of Permian age.

High terrace deposits of Quaternary age, which are older than the alluvial deposits, crop out on topographically higher areas north of the landfill. The terrace deposits are 9 to 23 meters thick. The upper 4 to 11 meters dominantly are moderate brown and light brown silty or sandy clay (Stacy, 1961 and James Warram, Terracon, unpub. data, 1991). The lower 11 meters consist of unconsolidated light brown fine to medium sand. Water levels in the high terrace deposits are 5 to 9 meters below land surface (Stacy, 1961, and James Warram, Terracon, written commun., 1991). The deposits are moderately permeable, and, where sufficient saturated thickness is present, yield moderate quantities of water to wells (0.3 to 2 liters per second, Wood and Burton, 1968, p. 25). The high terrace deposits also are underlain by rocks of the Hennessey Group, which crop out about 400 meters east of the landfill along Bishop Creek (fig. 2).

Permeability is low in the underlying Hennessey Group, considered a confining unit, composed mainly of mudstone. It is about 60 meters thick in the Norman Landfill area. Wells



Figure 2. Geology of the Norman Landfill area.



Figure 3. Potentiometric surface near the Norman Landfill area (based on Scholl and Christenson, 1998).

#### 6 Water Chemistry Near the Norman Landfill, Cleveland County, Oklahoma, 1995

completed in the Hennessey Group supply small amounts of water for domestic and stock use, mostly from a zone of weathered material above relatively unaltered mudstone, and lesser amounts from fractures and cavities left by dissolution of soluble materials (Wood and Burton, 1968, p. 22). The Hennessey Group conformably overlies the Garber Sandstone, a major freshwater aquifer in central Oklahoma.

#### Site History

The Norman Landfill is a closed municipal landfill that accepted solid wastes from 1922 until 1985 (Robertson and others, 1974). The landfill was closed in 1985 and covered with a clay cap (Dixon and Popoola, 1992, p. 8). Waste deposited in the landfill prior to 1973 was largely unrestricted and may include substances now recognized as hazardous. Dissolved and suspended substances leached from wastes have been found in a plume that extends hydraulically downgradient from the landfill, toward the Canadian River (Scholl and Christenson, 1998, p. 1).

Historical aerial photographs from 1951, 1966, 1978, 1985, and 1995 (figs. 4-8) and records of the landfill (Dixon and Popoola, 1992, p. 6) indicate that solid waste was deposited at many sites in the Norman Landfill area, primarily south of Bratcher-Miner Road, north of the 1995 Canadian River location, east of Chautauqua Avenue, and west of Jenkins Avenue. Solid waste also was deposited in an area extending about 350 meters west of Chautauqua Avenue (figs. 1 and 5). The Canadian River was north of the location of the modern capped landfill in the 1930s (Dixon and Popoola, 1992, p. 6). Cleveland County residents may have dumped trash north of the location of the capped landfill at that time. Small roads and trails north of the landfill site on the first aerial photograph, 1951, indicate that waste was dumped and possibly buried with alluvial sand just south of Bratcher-Miner Road (labeled D in fig. 4). The City began dumping waste in trenches, dug down to the water table, and subsequently buried the waste with about 15 centimeters of alluvial sand in 1960. Trenching and burial is most apparent in the 1966 photograph southwest of the intersection of Bratcher-Miner Road and Chautauqua Avenue (labeled with a white T in fig. 5). East-west oriented ridges resulting from collapse of the buried waste in the trenches are still apparent in the 1995 aerial photograph (fig. 8). The 1966 aerial photograph (fig. 5) also shows activities associated with burying the discharge line from the wastewater treatment plant that separates the west and east cells of the landfill. The Norman Landfill was an open dump until 1969 when the City of Norman leased the property to Norman Asphalt Co. for the operation of a sanitary landfill.

Trenching was abandoned in 1971 and lifts of sand were constructed to assure that waste was deposited at least 0.6 meter above the water table. Waste was buried daily with alluvial sand. Locations of linear ridges in the 1978 aerial photograph (fig. 6) suggest that much of the post-1966 trenching was within the area of and just north of the west cell. Lifts had been constructed over the east cell and the southern part of the west cell.

The landfill was closed in 1985 (fig. 6). The landfill was covered by a 0.9-meter cap of compacted clay in June 1989.

The aerial photographs illustrate the extent with which the river channel and local streams have varied in location. The peak flow of record at the Canadian River at Bridgeport gage (4,500 cubic meters per second for the period of 1945–1998) was measured 120 kilometers upstream by the U.S. Geological Survey on June 23, 1948. The photograph in figure 4, taken in 1951, shows a wide light-colored scar, including an area within the west cell of the modern landfill, where the river scoured the valley. The main river channel was southwest of the 1995 location and Bishop Creek discharged in the east cell of the landfill.

The downstream migration of a meander in the Canadian River can be followed on the aerial photographs in figures 5 through 8. The river channel was within 50 meters southwest of the west cell in 1966 (fig. 5). Light colored areas where vegetation was scoured away by the river during high flow are apparent south and west of the cells. The cutbank of a meander was 800 meters to the northwest of the landfill, along the southern bank of the river. The river still flowed near the landfill in 1978 (fig. 6). No scarring was apparent within the area of the modern capped landfill at that time. The cutbank of the meander had advanced 150 meters to the southeast. The river was flowing along the southern border of the west cell by 1985, and the southwestern border of the east cell (fig. 7). The cutbank had migrated 950 meters further south. After a flood in 1986, the river shifted south of the landfill and by 1995, the cutbank of the meander was more than 500 meters south of the landfill.

#### Acknowledgments

This work could not have been completed without the help of Robert Puls and Cindy Paul of the U.S. Environmental Protection Agency, who provided the chemical analyses of the samples by the U.S. Environmental Protection Agency, National Risk Management Research Laboratory in Ada, Oklahoma. Thanks are extended to John Potts and Walter Vandenburg for providing access to their wells and property. Thanks also to the City of Norman, especially Jim Berry, the Director of Public Works, for allowing research to be conducted at this site, and the Norman Asphalt Company for allowing access to the landfill.

## Method of Investigation

Background and downgradient chemistry of ground and surface water at the Norman landfill was characterized by sampling 26 wells, the Canadian River, and the slough in fall 1995. Ground-water samples were collected from 17 wells in a network of shallow, small-diameter, temporary drive-point wells that were installed during fall 1995 to determine potentiometric-surface elevations (fig. 9). Water samples collected from the



EXPLANATION

Outline of the base of the capped landfill (1995).

D Location where solid waste apparently was dumped during the early 1950's.

**Figure 4.** Digitized aerial photograph taken on March 22, 1951, showing the approximate location of the modern capped Norman Landfill and location where waste was deposited. Numerous small roads north and west of the landfill also may lead to dumping locations.





**Figure 5.** Digitized aerial photograph taken on November 11, 1966, showing the approximate location of the modern capped Norman Landfill and location of waste disposal in trenches in the Canadian River alluvium.



EXPLANATION Outline of the base of the capped landfill (1995). Location of active landfill in 1978.

**Figure 6.** Digitized aerial photograph taken on October 29, 1978, showing the approximate location of the modern capped Norman Landfill and location of active landfill in 1978.



Outline of the base of the capped landfill (1995).





EXPLANATION Outline of the base of the capped landfill (1995). 

AERIAL PHOTO COPYRIGHT ACE AERIAL PHOTO SERVICE, Oklahoma City, Oklahoma

Figure 8. Digitized aerial photograph taken on February 24, 1995, showing the approximate location of the modern capped Norman Landfill.



**Figure 9.** Location of background and downgradient surface-water collection sites and wells sampled near the Norman Landfill, October 25, 1995, through December 7, 1995. (Shallow wells are less than 2 meters below the water table, deep wells are more than 2 meters below the water table.)

potentiometric-surface network wells were designated with the letters PS followed by a site number that reflects the order of well installation (fig. 9). Two of the shallow potentiometric-surface network wells on either side of the slough (PS38 and PS54) were driven to depths ranging from about 2 meters below the water table to the base of the alluvium (12 meters) to determine whether the plume extended at depth southwest of the slough and to characterize deeper water downgradient of the landfill. Specific conductance was determined about every 1.2 meters and water samples were collected about every 3.3 meters below the depth accessed by the shallow wells at each site. Suffixes B, C, and D designate the three samples collected at increasing depths. Additional information on background ground-water quality for deeper wells was obtained by sampling water from one domestic well (WS1) and two stock wells (JPESW and JPWSW) (fig. 9). The river was sampled upgradient of the landfill to characterize background surface-water quality and the chemistry of this potential ground-water source. The slough

### **Selection of Sampling Sites**

leachate discharge.

Twelve wells (PS04, PS06, PS07, PS08, PS10B, PS12, PS17, PS18, PS22, JPESW, JPWSW, and WS1) were selected for collection as nonleachate contaminated or background samples. The term background is used for the well or sample location not hydraulically downgradient from the capped landfill. Waste has been buried at many locations in the area since 1922, and those locations were not well documented. However, because the sampled wells were outside the areas where aerial photos suggest trash had been buried, water from the background wells is thought to have been representative of background ground-water quality, except PS06, which was located next to a petroleum-production tank battery, and may have been contaminated by oilfield brine.

was sampled to determine the quality of surface water receiving

Eight shallow wells (PS35, PS36, PS37, PS38, PS39, PS40, PS43B, and PS54) were sampled to characterize the quality of ground water hydraulically downgradient from the capped landfill (fig. 3). Six wells (PS35 through PS39 and PS54) were along a transect, known as the PS35 transect, that extended southwest from the landfill. The transect began within the zone of sulfate reduction identified by Suflita and others (1988, p. 192). PS40 and PS43B were south of the west cell near the edge of the landfill cap in the shallow methanogenic part of the leachate plume.One surface-water sample was collected from the Canadian River upstream of the capped landfill. A second surface-water sample was collected from the slough at a location between the two vertical-depth-profile sites.

### **Quality Assurance**

Two duplicate samples and two blank-water samples were collected for quality assurance. Duplicate samples were collected from wells PS18 and PS38B after initial sample collec-

tion to provide information on repeatability. One quality assurance ambient blank sample (AB01) was collected at site 38 after PS38C was collected. Trace-element-free water from the U.S. Geological Survey Water Quality Support Unit in Ocala, Florida, was poured into sample bottles under ambient conditions to determine whether wind-blown dust or chemical vapors contaminated the samples. An equipment blank (FB01) was collected at the PS54 site by pumping trace-element-free water through a fresh piece of silicone tubing used for sampling and a fresh filter for dissolved analyses to determine whether the sampling equipment contaminated the samples. The tubing and filter were rinsed with about 500 milliliters of blank water before sampling. Bottles used for the blank samples were rinsed twice with blank water before filling. Blank-water samples were preserved for analyses using identical procedures to those used to preserve regular samples (table 1).

### **Sample Collection and Preservation**

Ground-water samples were collected using sampling and preservation methods described in Koterba and others (1995) and low-flow pumping procedures described in Puls and Barcelona (1996). Surface-water samples were collected using methods described in Wells and others (1990). Sample collection and preservation procedures used are summarized in table 1.

The PS background and downgradient ground-water samples were collected from drive-point wells. The wells consisted of 2.64-centimeter inside-diameter schedule-40 threaded stainless-steel pipe attached to stainless-steel screened sandpoints with a screen length of 0.76 meter and a slot width of 0.15 millimeter. Nickel-Teflon tape was used to cover the pipe threads and keep them from adhering to the couplings. The wells were driven using an electric jack hammer until the well screens were just below the water table.

The drive-point wells were developed and purged prior to collection of samples to ensure that the water was representative of water chemistry in that depth-interval of the aquifer. Water was pumped from the drive-point wells with a variable-speed peristaltic pump through silicone tubing, which was lowered into each well to the level of the screened interval. The wells were developed by pumping until sand production decreased and the water appeared clear. After the well was developed, specific conductance, water temperature, pH, and dissolved-oxygen concentration were measured at 10- to 15-minute intervals in a flow-through cell with probes attached to portable meters. Water temperature was measured with the temperature sensor on the specific conductance meter. Water samples from the drive-point wells were collected after at least two consistent readings were obtained for specific conductance, pH, and dissolved-oxygen concentration. Subsequent tests have shown that silicone tubing is permeable to oxygen, making the reported dissolved oxygen concentration suspect. Unfiltered samples for total analysis and field ammonia were collected first. Samples for dissolved major-cation and trace-element determinations

Table 1. Field procedures for collection and preservation of water samples for laboratory analysis used at the Norman Landfill, Oklahoma

[Dissolved, concentration in filtered sample; total, total recoverable concentration including suspended and colloidal solids; C, degrees Celsius]

Constituents	Filter type used	Collection-bottle type	Preservation method
Major cations, iron and manganese, and trace elements, dissolved	0.45-micrometer cartridge filter	125-milliliter acid- rinsed polyethylene	Ultrapure nitric acid to pH 2 or less
Major cations, iron and manganese, and trace elements, total	Unfiltered	125-milliliter acid- rinsed polyethylene	Ultrapure nitric acid to pH 2 or less
Major anions, dissolved	Unfiltered	250-milliliter poly- ethylene	Chill to $4^{\circ}C$
Nutrients and organic carbon, total	Unfiltered	250-milliliter amber polyethylene	Sulfuric acid to pH less than 2, chill to $4^{\circ}C$
Organic carbon, dissolved	0.45-micrometer silver filter	125-milliliter baked amber glass	Chill to 4 <sup>°</sup> C
Uranium	0.45-micrometer cartridge filter	125-milliliter acid- rinsed polyethylene	Nitric acid to pH 2 or less
Field ammonia	Unfiltered	125-milliliter baked amber glass	Hydrochloric acid to pH less than 6, chill to 4 C
Mercury, total	Unfiltered	125-milliliter baked amber glass	0.1 milliliter of 1,000-microgram per liter gold solution, 0.5 milliliter nitric acid, chill to 4 °C
Stable isotopes of hydrogen and oxygen	Unfiltered	60-milliliter glass with polyseal cap	None

were filtered through a prewashed 0.45 micrometer cartridge filter.

Three well-water samples (WS1, JPESW, and JPWSW) were collected using submersible pumps. Two well volumes were purged prior to sampling. During purging, specific conductance, water temperature, pH, and dissolved-oxygen concentration were recorded at 10- to 15-minute intervals until consistent readings were obtained. Discharge from the stock wells (JPESW and JPWSW) was too great to allow attachment of the flow-through cell so the water properties were measured in water discharged into a plastic bucket using probes attached to portable meters. The water samples from the stock wells were analyzed only for specific conductance, water temperature, pH, dissolved-oxygen, sulfide, ferrous iron, total major ions, and trace elements concentrations.

A cross-sectional equal-width depth-integrated composite sample was collected from the Canadian River at CR01. The water sample was placed in a churn splitter, in which specific conductance and pH were measured. Water temperature was measured in the river using the temperature sensor on the specific conductance meter. Samples for dissolved constituent determination were filtered from the churn splitter using a peristaltic pump and a 0.45 micrometer prewashed cartridge filter. A grab sample was collected from the river for stable-isotope analysis.

Field measurements of specific conductance, pH, water temperature, dissolved oxygen, sulfide, alkalinity, and ferrous iron were made at the sampling sites. Filtered and unfiltered samples were collected for determination of dissolved and total major cations, phosphate, organic carbon, iron, manganese, and trace elements, and dissolved major anions, nutrients, and uranium (table 1). Unfiltered samples were collected to determine the hydrogen and oxygen isotope composition of the water.

#### **Analytical Methods**

Field analyses for alkalinity, dissolved oxygen, ammonia, ferrous iron, and sulfide were made on site or within 24 hours of sample collection. Alkalinity was measured by incremental titration (Wells and others, 1990, p. 53-56) using 0.1639 normal

sulfuric acid with a 25 or 10-milliliter aliquot of a filtered sample.

Dissolved-oxygen concentration was measured using several techniques in an attempt to determine the best method. Oxygen concentration was monitored by a probe in the flowthrough cell attached to an electrometric meter. This method was generally acceptable for water with greater than 0.5 milligram per liter dissolved oxygen concentration. A modified Winkler titration (Hach, 1989, p. 436) was used to verify dissolved oxygen concentration detected by the meter. The Winkler method could not be used on samples from downgradient wells because alkalinity was too great. Dissolved oxygen concentration less than 0.8 milligram per liter also was measured using the Hach low range AccuVac ampule indigo carmine method and a spectrophotometer (Hach, 1989, p. 445). Dissolved-oxygen concentration in water also was analyzed using Chemetrics visually read CHEMettes kits. The lowest dissolved-oxygen concentration readings were reported.

Field-ammonia, ferrous iron, and sulfide concentrations were determined as soon after sample collection as possible using specific-ion and spectrophotometric techniques. Field ammonia concentration was determined within 24 hours of sample collection. Ionic strength and pH were adjusted with a sodium-hydroxide based ionic-strength adjusting solution prior to ammonia determination by a specific ion electrode. Ferrous iron concentration was determined colorimetrically on site using the Hach AccuVac ampule phenanthroline method and a portable spectrophotometer (Hach, 1989, p. 311). Sulfide concentration was measured colorimetrically using the methylene-blue method (Hach, 1989, p. 572) and a portable spectrophotometer. Many of the sulfide determinations did not include a sample-water blank to correct for turbidity. False-positive sulfide detections may have resulted.

Laboratory determinations, with the exception of uranium and stable isotopes of hydrogen and oxygen, were done at the U.S. Environmental Protection Agency, National Risk Management Research Laboratory, in Ada, Oklahoma. Samples collected for determination of major cations and trace elements were microwave-digested and analyzed using inductively coupled argon plasma atomic-emission spectroscopy. Mercury was determined using cold-vapor atomic-absorption spectroscopy. Arsenic and selenium concentrations were measured using graphite-furnace atomic-absorption spectroscopy. Sulfate, chloride, fluoride, bromide, nitrite, and nitrate concentrations were measured using Waters capillary electrophoresis method number N-601. Ammonia concentration was determined using U.S. Environmental Protection Agency method 350.1 (U.S. Environmental Protection Agency, 1993, p. 350.1-1). Orthophosphate concentration was measured using U.S. Environmental Protection Agency colorimetric method 365.1 (U.S. Environmental Protection Agency, 1993, p. 365.1-1). Total phosphorous was determined using U.S. Environmental Protection Agency colorimetric method 365.4 (U.S. Environmental Protection Agency, 1979, p. 365.4-1). Total and dissolved organic carbon were analyzed using a Dohrmann DC-80 carbon analyzer. Samples were acidified with phosphoric acid and

purged with nitrogen gas to remove inorganic carbon. Organic carbon was converted to carbon dioxide gas by high-temperature combustion and the carbon dioxide gas was detected using a nondispersive infrared detector.

Hydrogen-isotope-ratio determinations were made by the U.S. Geological Survey National Water Quality Laboratory in Arvada, Colorado, (now located in Lakewood, Colorado) using a hydrogen equilibration technique at 30 degrees Celsius (Coplen and others, 1991). This technique measures deuterium activity rather than concentration. Oxygen-isotope ratios were measured using the carbon dioxide equilibration technique at 25 degrees Celsius (Epstein and Mayeda, 1953).

Uranium concentration was determined by the U.S. Geological Survey, Geologic Division, Denver, Colorado, by laser induced fluorescence with a Scintrex instrument. Hexametaphosphate was added to the water sample and the resulting mixture was exposed to a pulsed laser. The amount of resulting fluorescence was measured to determine uranium concentration. Samples with high organic carbon content were heated to dryness with hydrogen peroxide, then redissolved with 0.16 normal nitric acid prior to analysis. All uranium determinations were made using standard additions.

### **Statistical Method**

A Mann Whitney U test (P-STAT, Inc., 1990, p. 45.17) was used to determine if concentrations of water-quality constituents were different between background and downgradient ground-water samples. The Mann Whitney U test is a nonparametric test that calculates the probability that two independent sample groups come from the same population. Concentrations and nondetects less than the greatest detection limit for each group were censored and set to a value slightly below the maximum detection limit to compensate for the effects of multiple detection limits (Helsel and Hirsch, 1992, p. 369). Only constituents with fewer than 60 percent censored values in either group were statistically compared. The null hypothesis is that concentrations in both groups have the same distribution. A small exact *p*-value (less than 0.05) suggests that the null hypothesis can be rejected, based on a significance level of 5 percent ( $\alpha$ =0.05). Rejections of the null hypothesis indicate content of a water-quality constituent is significantly different between background and downgradient categories.

## **Quality Assurance Results**

Most analytical results for quality assurance duplicate samples from wells PS 18 and PS 38B were in agreement to within 10 percent; however, there was a substantial difference between unfiltered concentrations of 11 elements as well as filtered concentrations of two elements. Results for total and dissolved potassium, the field determined nitrogen as ammonia, total aluminum, total arsenic, dissolved boron, total cobalt, total copper, total germanium, total molybdenum, total silver, total tellurium, and total thallium varied by more than 10 percent in at least one of the duplicate sample sets. The variation in total concentrations between these duplicate samples likely was a result of inclusion of one or more element-rich suspended particle in one sample as a result of natural variability. The variation in dissolved potassium, field nitrogen as ammonia, and dissolved boron may represent natural variation in concentration or may result from analytical error.

The ambient blank, AB01, contained detectable amounts of total phosphate, total organic carbon, and total titanium, suggesting a wind-blown contaminant source or contaminated bottles for the three unfiltered samples. The phosphate and titanium results were similar to the smallest sample concentrations, whereas, organic carbon was considerably less than sample concentrations.

Sampling equipment may have contaminated some samples with calcium, total phosphate, total organic carbon, dissolved aluminum, dissolved antimony, total arsenic, dissolved silver, total strontium, dissolved tellurium, and dissolved vanadium. The equipment blank, FB01, contained detectable total calcium, total phosphate, and total organic carbon at concentrations well below amounts detected in samples. The equipment blank also contained dissolved aluminum, dissolved antimony, total arsenic, dissolved silver, total strontium, dissolved tellurium, and dissolved vanadium. Concentrations of dissolved aluminum, antimony, lead, silver, tellurium, and vanadium in the blank indicate contamination by the sampling equipment that could substantially increase concentrations of these elements in the samples. Thus, concentrations listed for these elements may have been greater than the true concentration. The equipment blank contained 21 micrograms per liter dissolved lead; however, lead was not detected in samples. The presence of arsenic and strontium in the unfiltered split, and not in the filtered split, indicates these elements were not introduced by the filter and probably were not introduced by the sample tubing.

## **Ground- and Surface-Water Chemistry**

The chemistry of the water samples reflects the source of the water and the chemical processes that result from landfill leachate contamination. Most constituent concentrations were greater in ground water from wells downgradient from the landfill than in water from background wells and were greater in the slough than in the Canadian River. Concentrations of many constituents in the slough were less than concentrations in ground water downgradient of the landfill, but were greater than concentrations in background ground water. The most substantial changes in shallow ground water chemistry tended to occur between PS36 and PS37, a region referred to as the transition zone. Concentrations of most constituents in the Canadian River, other than sulfate, manganese, and iron, were similar to concentrations in background ground water.

#### **Physical Properties**

Physical properties of the water samples varied by sample source and with distance downgradient from the landfill. Specific conductance generally was less in water from background wells than in water from downgradient wells (fig. 10, table 2). The greatest conductance measured for water from a background well was for well PS06, which may be affected by a small oil-field-brine spill (fig. 11). Specific conductance and alkalinity were greater in the slough than in the river (table 2). Specific conductance of the river was near the 75th quartile of background ground water, whereas specific conductance of the slough was near the 25th quartile of specific conductance in downgradient ground water near the slough (fig. 12). Specific conductance along the transect was greater than 5,500 microsiemens per centimeter within 40 meters of the landfill, dropped to 3,170 in the transition zone, then decreased with increasing distance from the landfill (fig. 12). pHs of water samples from the river and slough were greater than pHs measured in ground water. pH values generally were greater in water from background wells than in water from downgradient wells (fig. 10 and table 2). Lesser pHs in downgradient ground water probably are due to carbon dioxide generated by degradation of organic matter and organic acids in the landfill leachate. River alkalinity was less than in background ground water (table 2). Alkalinity in the slough was similar to the lowest alkalinity measured in downgradient ground water. The high alkalinity of the contaminated ground water probably was a result of carbon dioxide produced during aerobic and anaerobic oxidation of organic compounds and the presence of organic acids. Alkalinity values decreased away from the landfill, possibly as a result of dilution by recharge (fig. 12). Dissolved oxygen concentration generally increased away from the landfill and was high in the slough, probably as a result of plant photosynthesis and uptake of atmospheric oxygen (fig. 12).

### **Major lons**

Major ion concentrations generally were less in water from background wells than in water from downgradient wells (fig. 13, tables 2 and 3). Calcium and bicarbonate were the dominant ions in water from background wells, except PS06 and WS1 (fig. 14). The dominance of sodium and chloride in water from well PS06 is consistent with its location near a possible oil-field brine spill. Sodium, chloride, and sulfate concentrations in water from WS1 may be indicative of typical water type at depth in the alluvium. Water from downgradient wells within 20 meters of the landfill (PS35, PS36, PS40, and PS43B) was typically a sodium-bicarbonate-chloride type (fig. 15). Water from wells further downgradient of the landfill, with the exception of water from well PS54, was a calcium-sodiumbicarbonate-chloride type (fig. 16). Water type at depth in the vertical profiles on either side of the slough was mixed, ranging from sodium-calcium-magnesium-bicarbonate-chloride to calcium-sodium-magnesium-bicarbonate-chloride (fig. 16). Water

# Table 2. Summary statistics for concentrations of selected water-quality constituents in water samples collected from background and downgradient wells near the Norman Landfill, Oklahoma

[All concentrations in micrograms per liter unless otherwise noted;  $\mu$ s/cm, microsiemens per centimeter; °C, degrees Celsius; dissolved, determined in filtered sample or by a technique that determines only the dissolved fraction; total, determined in unfiltered sample; CaCO<sub>3</sub>, calcium carbonate; as N, indicates concentration reported as nitrogen; PO<sub>4</sub>, phosphate; N, number of analyses; min, minimum concentration; max, maximum concentration; Percent censored, percent of samples with concentration less than the group maximum reporting level; *p*-value, indicates the exact 1-tailed attained significance level from a Mann Whitney U test comparing background and downgradient water samples; --, no results for this category; N.D., none detected; <, indicates actual *p*-value is less than value shown]

		Background ground water							Downgradient ground water								Surface water				
Name		D	etected		Non- detec	cted	Per- cent		l	Detected		de	Non- tected	Per- cent	Water exact		Detec	ted	N	ondetected	
	Ν	Min	Median	Max	Ν	Max	cen- sored	Ν	Min	Median	Max	Ν	Max	cen- sored	<i>p</i> -value	N	River	Slough	N	Range	
Specific conductance (µs/cm)	13	465	993	2,180	0		0	15	1,861	5,350	7,710	0		0	< 0.001	2	1,618	2,770	0		
Eh (volts)	11	0.055	0.101	0.35	10		0	15	0.075	0.177	0.216	50		0	0.004	0			0		
pH (standard units)	13	6.99	7.12	7.40	Ů Ő		0	15	6.58	6.86	7.34	0		0	< 0.001	2	8.61	8.22	0		
Water temperature (°C)	13	16.4	19.0	21.4	0		0	15	16.4	18.1	19.9	0		0	0.007	2	10.8	16.0	0		
Oxygen, dissolved	13	0.004	0.60	6.6	0		0	15	0.25	0.58	1.3	0		0	0.301	1		13.	0		
Sulfide, dissolved	12	0.003	0.016	0.18	0		0	11	0.002	0.01	0.28	4	0.01	0	0.333	0			0		
Alkalinity as CaCO <sub>3</sub> , dissolved	13	248	474	646	0		0	15	632	1,980	2,870	0		0	< 0.001	2	218	790	0		
Calcium, dissolved	11	67	117	163	0		0	15	108	298	489	0		0	< 0.001	2	176	103	0		
Calcium, total	13	66	121	334	0		0	15	99	297	444	0		0	< 0.001	2	181	209	0		
Magnesium, dissolved	11	15	31	36	0		0	15	35	116	177	0		0	< 0.001	2	57	98	0		
Magnesium, total	13	15	33	62	0		0	15	36	108	173	0		0	< 0.001	2	58	108	0		
Sodium, dissolved	11	4.1	49	260	0		0	15	172	515	652	0		0	< 0.001	2	105	322	0		
Sodium total	13	4.3	74	277	0		0	15	185	526	690	0		0	< 0.001	2	110	329	0		
Potassium dissolved	9	0.35	3.0	47	2	23	46	9	9.5	62	394	6	2.8	40	0.018	1	N.D.	13	1	2.8	
Potassium, total	11	0.38	2.8	44	2	1.0	23	15	2.7	15	352	0		0	< 0.001	2	2.6	18	0		
Bicarbonate, dissolved	13	303	578	788	0		0	15	771	2,420	3,500	0		0	< 0.001	2	256	964	0		
Carbonate, dissolved	13	0	0	0	0		0	15	0	0	0	0		0		2	5	0	0		
Sulfate, dissolved	13	3.0	61	169	0		0	6	3.4	90.8	458	9	5.5	73		2	486	133	0		
Chloride, dissolved	12	2.9	26.2	509	1	5.5	23	15	206	821	1,080	0		0	< 0.001	2	124	300	0		
Fluoride, dissolved	4	1.0	1.48	2.2	9	5.5	100	0		N.D.		15	5.5	100		0	N.D.	N.D.	2	3.0 - <b>S</b> 5	
Bromide, dissolved	3	3.0	3.3	3.8	10	5.5	100	1		10		14	5.5	93		0	N.D.	N.D.	2	3.0 - <b>9</b> .5	
Nitrite as N. dissolved	0		N.D.		7	0.05	100	0		N.D.		10	0.05	100		0	N.D.		1	0.05 <b>a</b>	
Nitrate as N. dissolved	2	0.12	0.42	0.72	5	0.05	71	0		N.D.		10	0.05	100		1	1.7		0	<u>a</u>	
Nitrite plus nitrate as N. dissolved	6	0.09	0.19	0.31	0		0	5	0.29	0.38	0.49	0		0	0.006	1		6.3	0	ŭ	
Ammonia as N, dissolved	12	0.11	1.38	2.5	1	0.05	8	15	0.87	20	321	0		0	< 0.001	1	N.D.	4.8	1	0.05 <b>face</b>	
Ammonia, field as N, dissolved	10	0.1	1.03	2.0	1	0.01	9	15	1.2	14	310	0		0	< 0.001	1	N.D.	2.8	1	0.01 <b>'</b>	
Phosphate, total as PO4, dissolved	8	0.06	0.76	1.3	0		0	15	0.07	0.1	1.0	0		0	0.075	2	0.09	0.91	0	ter	
Orthophosphate as PO <sub>4, dissolved</sub>	10	0.05	0.56	1.3	3	0.05	23	5	0.27	0.58	1.1	10	0.05	67		1	N.D.	0.75	1	0.05 <b>C</b>	
Organic carbon, dissolved	13	0.2	3.5	5.2	0		0	15	24	102	182	0		0	< 0.001	2	2.8	26	0	ner	
Organic carbon, total	13	0.5	5.3	30.1	0		0	15	23.3	119.	302.	0		0	< 0.001	2	3.6	75.	0	nistr	
Iron, dissolved	10	1.1	2.38	3.8	1	0.003	9	15	1.4	13	24	0		0	< 0.001	0	N.D.	N.D.	2	-	
Ferrous iron, dissolved	13	0.05	1.7	2.9	0		0	14	1.4	10.8	16	0		0	< 0.001	1		0.01	0		
Iron, total	13	1.3	3.08	22	0		0	15	1.5	13	25	0		0	0.002	2	0.99	17	0		
Manganese, dissolved	11	0.02	1.0	2.1	0		0	15	0.27	0.84	1.3	0		0	0.399	2	0.02	0.02	0		
Manganese, total	13	0.05	1.1	2.1	0		0	15	0.28	0.86	1.3	0		0	0.358	1	N.D.	0.83	1	0.08	



Figure 10. Background and downgradient ground-water properties near the Norman Landfill.



**Figure 11.** Relationship of alkalinity to specific conductance and chloride in ground water. The sample from well PS06 may be contaminated by oil-field brine.



**Figure 12.** Changes in water properties along a transect of shallow wells and surface water from the Norman Landfill to just southwest of the slough.

#### Table 3. Significant findings on water chemistry near the Norman Landfill, Oklahoma

 $[\mu s/cm, microsiemens per centimeter; Dissolved, determined in filtered sample or by a technique that determines only the dissolved fraction; mg/L, milligrams per liter; µg/L, micrograms per liter; to$ tal, determined in unfiltered sample; CaCO<sub>3</sub>, calcium carbonate; as N, indicates concentration reported as nitrogen; PO<sub>4</sub>, phosphate; --, no data available; ND, none detected; <, less than; Significantlygreater or Less, indicates comparison of downgradient and upgradient attained significance based on a Mann Whitney U test at a significant level of 5 percent; Greater, Similar, or Less, concentrationwas greater, similar or less than that in compared with, but was not tested statistically; just upgradient of the slough, greater than 20 meters from the landfill]

Constituent name	Back- ground ground- water range	Down- gradient ground- water range	Relation of downgradient concentrations to background concentrations in ground water	Location of greatest concentrations	Relation of slough concentration to Canadian River concentration	Relation of Canadian River concentration to ground-water background concentrations	Relation of slough concentration to ground-water background concentrations	Relation of slough concentration to concentrations in shallow ground water just upgra- dient of the slough
			Wa	ter properties				
Specific conductance (µs/cm)	465 - 2,180	1,861 – 7,710	Significantly greater	PS40 PS38C	Greater	Similar	Greater	Greater
Eh (volts)	0.055 - 0.351	0.075 - 0.216	Significantly greater	PS06, PS54D				
pH (standard units)	6.99 - 7.40	6.58 - 7.34	Significantly less	CR01, SLOUGH	Less	Greater	Greater	Greater
Sulfide, dissolved (mg/L)	0.003 - 0.18	0.002 - 0.28	Similar	PS40, PS38C				
Alkalinity as CaCO <sub>3</sub> , dissolved (mg/L)	248 - 646	632 - 2,870	Significantly greater	PS40, PS38C, PS35	Greater	Less	Greater	Less
				Major ions				
Calcium, dissolved (mg/L)	67 - 163	108 - 489	Significantly greater	PS38D, PS54D	Less	Greater	Similar	Less
Magnesium, dissolved (mg/L)	15 - 36	35 – 177	Significantly greater	PS38B, PS38C	Greater	Greater	Greater	Greater
Sodium, dissolved (mg/L)	4.1 - 260	172 - 652	Significantly greater	PS40. PS38D	Greater	Similar	Greater	Greater
Potassium, dissolved (mg/L)	0.35 – 47	<2.8 - 394	Significantly greater	PS40, PS36	Greater		Similar	Less
Bicarbonate, dissolved (mg/L)	303 - 788	771 – 3,500	Significantly greater	PS40, PS38C	Greater	Less	Greater	Less
Sulfate, dissolved (mg/L)	3.0 - 169	<3.0 - 458	Less	CR01, PS37	Less	Greater	Similar	Less
Chloride, dissolved (mg/L)	2.9 - 509	206 - 1.080	Significantly greater	PS38D.PS38C	Greater	Similar	Similar	Greater
		,,		Nutrients				
Nitrate or Nitrite plus nitrate as N, dissolved (mg/L)	< 0.05 - 0.72	0.29 - 0.49	Significantly greater	SLOUGH	Greater	Greater	Greater	Greater
Ammonia as N, dissolved (mg/L)	< 0.05 - 2.5	0.87 - 321	Significantly greater	PS40, PS36	Greater	Less	Greater	Less
Phosphate, total as PO <sub>4</sub> , dissolved (mg/L)	< 0.01 - 2.0	0.07 - 1.0	Similar	PS18, PS17, PS39	Greater	Similar	Similar	Less
Orthophosphate as PO <sub>4</sub> , dissolved (mg/L)	<0.02 - 1.3	< 0.05 - 1.1	Similar	PS18, PS39, PS17	Greater		Similar	Less
			Or	ganic carbon				
Organic carbon, dissolved (mg/L)	0.2 - 5.2	24 - 182	Significantly greater	PS40. PS38C	Greater	Similar	Greater	Similar
Organic carbon, total (mg/L)	0.5 - 30.1	23.3 - 302	Significantly greater	PS35, PS36	Greater	Similar	Greater	Less
			Tra	ace elements				
Arsenic, dissolved (µg/L)	<1-5	<1 - 24	Significantly greater	PS40, PS35	Less	Greater	Similar	Greater
Arsenic, total (µg/L)	<1-10	4 - 34	Significantly greater	PS35, PS38	Greater	Similar	Greater	Similar
Barium, dissolved (mg/L)	.105 –.628	.150 - 12.800	Significantly greater	PS40, PS38B	Greater	Similar	Greater	Greater
Boron, dissolved (mg/L)	.074 –.714	1.070 - 9.580	Significantly greater	PS38C, PS54C	Greater	Similar	Greater	Less
Cadmium, dissolved (µg/L)	<2-4	<2-13	Significantly greater	PS38D, PS38C, PS54C	Less	Greater		Less

Table 3. Significant findings on water chemistry near the Norman Landfill, Oklahoma—Continued

Constituent name	Back- ground ground- water range	Down- gradient ground- water range	Relation of downgradient concentrations to background concentrations in ground water	Location of greatest concentrations	Relation of slough concentration to Canadian River concentration	Relation of Canadian River concentration to ground-water background concentrations	Relation of slough concentration to ground-water background concentration	Relation of slough concentration to concentrations in shallow ground water just upgra- dient of the slough
			Trace e	lements—Continued				
Cobalt, dissolved (µg/L)	<2-4	<3-23	Greater	PS40, PS38C, PS38D	ND			Less
Iron, dissolved (mg/L)	< 0.003 - 3.8	1.4 – 24	Significantly greater	PS38C, PS40	ND			Less
Ferrous iron, dissolved (mg/L)	0.05 - 2.9	1.4 – 16	Significantly greater	PS54C, PS38D			Less	Less
Iron, total (mg/L)	1.3 – 22	1.5 – 25	Significantly greater	PS38C, PS40, PS04	Greater	Less	Similar	Greater
Manganese, dissolved (mg/L)	0.02 - 2.1	0.27 – 1.3	Similar	PS12, PS17	Equal	Similar	Similar	Less
Manganese, total (mg/L)	0.05 - 2.1	0.28 - 1.3	Similar	PS12, PS17	Greater		Similar	Similar
Nickel, dissolved (µg/L)	<7 - 64	14-49	Significantly greater	PS38D, PS38B	Less	Similar	Similar	Less
Selenium, dissolved (µg/L)	<1-4	<1-7	Significantly greater	PS37, PS38, PS38D	Equal	Similar	Similar	Greater
Strontium, dissolved ( $\mu g/L$ )	.657 – 2.37	1.570 - 7.73	Significantly greater	PS38C, PS38B	Greater	Similar	Similar	Greater
Uranium, dissolved (µg/L)	<0.5 - 5.1	< 0.5 - 0.8	Similar	PS04, CR01	Less	Similar	Similar	Greater
Zinc, dissolved (µg/L)	<1-76	<3 - 19	Less	PS06, PS04	ND			ND
			S	table isotopes				
δ.deuterium (per mil)	-45.827.9	-34.51.0	Significantly greater	PS40, PS54	Greater	Similar	Greater	Greater
$\delta^{18}$ oxygen (per mil)	-7.054.67	-6.38 0.55	Similar	PS54, SLOUGH	Greater	Greater	Greater	Greater



**Figure 13.** Dissolved major-ion concentrations in ground-water collected from background and downgradient wells near the Norman Landfill, October 25, 1995, through December 7, 1995.



**Figure 14.** Major-ion concentrations in water from background wells near the Norman Landfill, October through December 1995. Well locations are shown in figure 8.



**Figure 15.** Major-ion concentrations in water from shallow downgradient wells near the Norman Landfill, October through December 1995. Well locations are shown in figure 8.



**Figure 16.** Major-ion concentrations in water from vertical-profile wells on either side of the slough near the Norman Landfill, October through December 1995. Depth profile sites 38 and 54 are shown in figure 8.

from the Canadian River was a mixed calcium-magnesiumsodium-sulfate type whereas the slough was a sodium-magnesium-bicarbonate-chloride type (fig. 17).

Major ion concentrations varied along the 35 transect. Dissolved calcium concentration increased from background concentrations of about 140 milligrams per liter to 292 milligrams per liter in the transition zone and then progressively decreased, returning to background concentrations in the slough (fig. 18). The largest magnesium concentration was in the slough and in water from wells upgradient from the transition zone (fig. 18). Sodium concentration was greatest near the landfill in water from well PS36, declined by about 40 percent in the transition zone, then further decreased towards the slough. The sodium concentration in the slough was greater than the sodium concentration in water from adjacent wells (fig. 18). Potassium concentration was greatest in shallow ground water near the landfill and decreased by 80 percent in the transition zone. The decrease in dissolved solids and chloride concentrations along the transect (figs. 15 and 18) may represent dilution by recharge. Sulfate was detected at low concentration in water from wells near the landfill (fig. 18). The lack of sulfate is consistent with sulfate-reducing or methanogenic conditions observed by Suflita and others (1988, p. 192). Sulfate concentration in the river was 3.6 times as great as the sulfate concentration in the slough (table 2).

#### Nutrients

Ammonia-nitrogen concentration was greater in water from wells upgradient from the transition zone and was much less downgradient from the transition zone and was least in the slough and water from PS54 (fig. 19). Ammonia is a common product of anaerobic decomposition of refuse and is commonly detected in landfill leachate (Baedecker and Back, 1979, p. 432). Nitrite plus nitrate concentrations in the slough (table 2) exceeded those in the river and in ground water. Lesser nitrite plus nitrate concentrations in water from well PS54 downgradient from the slough indicate that reducing processes in the bed or banks of the slough may occur.

Total phosphate and orthophosphate concentrations were similar in background ground water, indicating that most phosphate was in the form of orthophosphate (fig. 20). Total phosphate and orthophosphate concentrations downgradient of the landfill and in the slough were similar to background concentrations (tables 2 and 3). Total phosphate concentration in the river was less than in the slough and similar to concentrations in ground water. Decreases in total phosphate concentrations with depth (fig. 21) probably indicates removal by sorption on alluvial sediments.

### **Organic Carbon**

Dissolved and total organic carbon concentrations in water from wells downgradient of the landfill were significantly greater than in water from background wells (fig. 20 and tables 2 and 3). Organic carbon concentration in the river was similar to the median concentration in background ground water. The organic carbon concentration was greater in the slough than in the river (table 2). Greater concentration of total organic carbon than dissolved organic carbon indicate that some organic carbon is transported in a suspended or colloidal form.

#### Iron and Manganese

Iron concentrations were statistically greater in water from wells downgradient of the landfill than in background well water (fig. 20, tables 2 and 3). Greater total iron concentration than dissolved iron concentration in water from some of the background wells indicates that some iron may have been in the form of suspended solids or colloids. Dissolved iron and total iron concentrations were nearly the same in water from downgradient wells. Dissolved iron increased with depth in both vertical profiles, and decreased near the base of the alluvium (fig. 21). Dissolved iron was not detected in either the Canadian River or the slough. Total iron concentration was greater in the slough than in the river (table 2).

Dissolved and total manganese concentrations in water from wells downgradient of the landfill were similar to background ground water concentrations (fig. 20, tables 2 and 3). Manganese concentration in shallow ground water increased away from the landfill to well PS38 and then decreased (fig. 22). Dissolved manganese concentration was low in both the slough and the river, similar to those in ground water (table 2). Total manganese was greater in the slough than in the river (table 2).

#### Trace Elements

Trace element concentrations varied by source with the greatest concentrations generally in water from downgradient wells. Dissolved and total concentrations of trace elements in water from background wells generally were low (fig. 23). Dissolved concentrations of arsenic, barium, boron, cadmium, chromium, cobalt, nickel, and strontium were greater or significantly greater overall in water from wells downgradient of the landfill than in water from the background wells (fig. 23 and tables 3 and 4). Concentrations of barium, boron, chromium, cobalt, and nickel in shallow ground water generally decreased toward the slough (fig. 24). Measured concentrations of dissolved arsenic, barium, boron, germanium, and strontium increased with depth and then decreased near the base of the alluvium at sites 38 and 54, (figs. 25a and 25b). Trace element concentrations generally were greater in the slough than in the Canadian River (table 4). Most detectable trace element concentrations in the river were similar to or less than background ground-water concentrations, whereas, most trace-element concentrations detected in the slough, except for dissolved chromium concentration, were in the upper range of or greater than background ground-water concentrations. Dissolved selenium was detected in both the river and the slough at 4 micrograms



**Figure 17.** Major-ion concentrations in water from the Canadian River and the slough near the Norman Landfill, October through December 1995.



**Figure 18.** Changes in dissolved major-ion concentrations along a transect of shallow wells and surface water from the Norman Landfill to just southwest of the slough.



**Figure 19.** Changes in dissolved nutrient and organic carbon concentrations along a transect of shallow wells and surface water from the Norman Landfill to just southwest of the slough.



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Figure 20. Nutrient, carbon, iron, and manganese concentrations in water from background and downgradient wells near the Norman Landfill, October 25, 1995, through December 7, 1995.



**Figure 21.** Changes in nutrient, dissolved organic carbon, iron, and manganese concentrations with depth in vertical profiles on either side of the slough near the Norman Landfill.



**Figure 22.** Changes in iron and manganese concentrations along a transect of shallow wells an surface wager from the Norman Landfill just southwest of the slough.



**Figure 23**. Dissolved trace element concentraions in ground water collected from background and downgradient wells near the Norman Landfill, October 25, 1995, through December 7, 1995.

# Table 4. Summary statistics for concentrations of trace elements and δ-values for water samples collected from background and downgradient wells near the Norman Landfill, Oklahoma

[All concentrations are in micrograms per liter unless otherwise noted; dissolved, determined in filtered sample; total, determined in unfiltered sample; mg/L, milligrams per liter; N, number of analyses; min, minimum concentration; max, maximum concentration; Percent censored, percent of samples with concentration less than the group maximum detection limit and censored for statistical analysis; Ground water exact *p*-value, indicates the attained significance level from a Mann Whitney U test comparing concentration in background and downgradient ground water samples; --, no results for this category; N.D., none detected; <, indicates actual *p*-value is less than value shown]

	Background ground water							Dov	wngradien	t ground w			Cround	Surface water						
		D	etected		Nondetected F		Per-		D	etected		Nond	etected	Per-	Water		Detec	ted	No	ndetected
Name	N	Min	Median	Max	N	Max	cent cen- sored	N	Min	Median	Max	N	Max	cent cen- sored	exact p-value	N	River	Slough	N	Detection limit
Aluminum, total Antimony, total Arsenic, dissolved Arsenic, total Barium, dissolved (mg/L)	5 1 10 12 11	1,050  1 1 .105	2,410 66 2. 3.5 .427	20,600  5 10 .628	8 12 1 1 0	200 96 1 1 	62 100 9 8 0	13 0 11 15 15	142  5 4 .150	690 N.D. 14 19 3.18	2,810  24 34 12.8	$\begin{array}{c}2\\15\\4\\0\\0\end{array}$	200 96 1 	27 100 27 0 0	 0.019 < 0.001 0.004	2 0 2 2 2	1,900 N.D. 7 8 .150	17,800 N.D. 3 21 .805	0 2 0 0 0	38 - 79  
Barium, total (mg/L) Boron, dissolved (mg/L) Boron, total (mg/L) Cadmium, dissolved Cadmium, total	13 11 13 7 10	.102 .074 .078 2 3	.438 .212 .259 3 4	1.74 .714 .689 4 11	$\begin{array}{c} 0\\11\\0\\4\\3\end{array}$	  2 2	$     \begin{array}{c}       0 \\       0 \\       36 \\       23     \end{array}   $	15 15 15 15 14	.165 1.07 .955 2 4	2.88 5.07 4.74 9 8	11.500 9.58 8.95 13 13	$     \begin{array}{c}       0 \\       15 \\       0 \\       0 \\       1     \end{array} $	  2	0 0 0 7	0.006 < 0.001 < 0.001 < 0.001 < 0.001	2 2 2 1 2	.180 .280 .261 5 5	1.21 1.23 1.32 N.D. 6	$\begin{array}{c} 0 \\ 0 \\ 0 \\ 1 \\ 0 \end{array}$	
Chromium, dissolved Chromium, total Cobalt, dissolved Cobalt, total Copper, dissolved	5 7 3 4 0	1 2 3	2 5 3.5 N.D.	4 230 4 11	6 6 8 9 11	2 2 6 6 76	73 46 100 92 100	6 9 13 14 9	3 4 9 11 23	5.5 12 17 18 63	9 23 23 22 72	9 6 2 1 6	8 11 6 4 76	87 53 13 7 100	0.001	$     \begin{array}{c}       1 \\       1 \\       0 \\       1 \\       1     \end{array} $	N.D. N.D. N.D. N.D. 23	2 19  10 	1 1 2 1 1	8 11 3 - 6 7 76
Copper, total Germanium, dissolved Germanium, total Lead, total Molybdenum, dissolved	1 2 5 1 7	110 25  5	233 128.5 84 89 6	147 180  17	12 9 8 12 4	110 100 100 37 15	92 82 85 92 82	8 9 0 5	46 77 44  7	89 128 124 N.D. 12	111 177 241  14	7 6 6 15 10	110 100 150 37 21	93 53 87 100 100	   	0 1 1 0 1	N.D. N.D. N.D. N.D. N.D.	N.D. 130 160 N.D. 8	2 1 1 2 1	43 - 76 63 150 21 - 34 21
Molybdenum, total Nickel, dissolved Nickel, total Selenium, dissolved Selenium, total	7 8 9 7 3	5 9 11 1 1	$     \begin{array}{r}       10 \\       19.5 \\       24 \\       2 \\       1     \end{array} $	67 64 1,520 4 2	6 3 4 4 9	15 7 10 1 1	85 27 31 36 75	13 15 15 13 8	7 14 12 1 1	22 34 33 3 3	28 49 52 7 5	2 0 0 2 7	10  1 1	20 0 13 47	0.004 0.008 0.013	2 2 2 2 2	$10 \\ 15 \\ 14 \\ 4 \\ 4 \\ 4$	$     \begin{array}{r}       10 \\       10 \\       28 \\       4 \\       2     \end{array} $	0 0 0 0 0	   
Silver, total Strontium, dissolved Strontium, total Tellurium, total Thallium, total	5 11 13 5 4	13 657 645 33 11	14 1.180 1.220 46 16.5	221 2.370 2.900 142 66	8 0 0 8 9	17  90 49	85 0 92 92	12 15 15 9 12	19 1.560 1.490 58 20	36 3.710 3.770 89 38	57 7.730 7.430 181 61	3 0 0 6 3	17  91 49	20 0 73 80	<0.001 <0.001  	$     \begin{array}{c}       1 \\       2 \\       2 \\       0 \\       1     \end{array} $	13 1.860 2.000 N.D. N.D.	N.D. 2.310 2.730 N.D. 33	$     \begin{array}{c}       1 \\       0 \\       0 \\       2 \\       1     \end{array} $	17  40 - 74 21
Titanium, total Uranium, dissolved Vanadium, total Zinc, dissolved Zinc, total	5 3 1 3 7	$     \begin{array}{r}       19 \\       0.6 \\       \\       35 \\       1     \end{array} $	33 2 42 38 25	243 5.1 76 3,380	8 8 12 8 6	20 0.5 32 12 180	69 72 92 73 92	6 6 1 5 7	$     \begin{array}{c}       6 \\       0.5 \\       \\       3 \\       4     \end{array} $	19 0.6 16 6 11	48 0.8  19 41	9 9 14 10 8	20 0.5 32 12 180	80 60 100 87 100	   	$     \begin{array}{c}       2 \\       2 \\       0 \\       0 \\       2     \end{array} $	32 5 N.D. N.D. 12	295 1.8 N.D. N.D. 35	0 0 2 2 0	 14 - 27 3 - 12 
δ deuterium (per mil) δ <sup>18</sup> oxygen (per mil)	11 11	-45.8 -7.05	-34.2 -5.61	-27.9 -4.67	$\begin{array}{c} 0 \\ 0 \end{array}$		0 0	15 15	-34.5 -6.38	-12.9 -5.47	-1.0 0.55	$\begin{array}{c} 0 \\ 0 \end{array}$		$\begin{array}{c} 0 \\ 0 \end{array}$	<0.001 0.380	2 2	-29.6 -4.31	-12.1 -1.53	$\begin{array}{c} 0 \\ 0 \end{array}$	



**Figure 24.** Changes in trace element concentrations along a transect of shallow wells and surface water from the Norman Landfill to just southwest of the slough.





**Figure 25b.** Changes in selected trace element concentrations with depth in vertical profiles on either side of the slough near the Norman Landfill.

per liter (table 4), a concentration that can cause reproductive failure or mortality of some fish and breeding waterfowl (Lemly and Smith, 1988, p. 9).

#### Stable isotopes

The ratio of stable hydrogen isotopes (deuterium [ $\delta$ D]) in water from background wells differed from those of ground water collected downgradient of the landfill (tables 3 and 4). The ratios of stable oxygen isotopes ( $\delta^{18}$ O) in water from background wells were similar to those in ground water downgradient of the landfill (tables 3 and 4).  $\delta$ D and  $\delta^{18}$ O of water from background wells plot along or just to the right of the Central Oklahoma regional ground water line determined by Parkhurst and others (1996, p. C25) (fig. 26), which indicates that the isotopic composition of background water had changed little since entering the alluvial aquifer. Isotopic ratios of water from the slough and Canadian River are shifted to the right of the line (fig. 26) indicating that evapotranspiration had affected the isotopic composition.

 $\delta D$  and  $\delta^{18}O$  of water from downgradient wells generally plot above the Central Oklahoma regional ground water line (fig. 26), which indicates an enrichment of deuterium. The greatest deuterium enrichment was in shallow ground water from the methanogenic zone (PS40). Deuterium enrichment relative to the regional ground-water line in this sample exceeded 30 per mil. Baedecker and Back (1979, p. 436) observed a similar deuterium enrichment in landfill leachate, and speculated that the apparent enrichment was either a result of decomposition of landfill materials enriched in deuterium, or of bacterial processes preferentially consuming the lighter hydrogen isotope. Hackley and others (1996, p. 834) reported 30 to 60 per mil enrichment in deuterium in leachate from three landfills in Illinois, and speculated that most of the enrichment was a result of methanogenisis (equation 1, Chapelle, 1993, p. 86), with some enrichment resulting from isotopic exchange with hydrogen sulfide (equation 2, Clark and Fritz, 1997, p. 22):

coenzyme M methylreductase 
$$2HDO + H_2O$$
  
 $\rightarrow 2CH_4 + 2H_2CO_3 + D_2O$  (1)

$$DS^- + H_2O \rightarrow HS^- + DHO$$
(2)

Deuterium enrichment at the Norman Landfill site also is likely to have been a result of preferential consumption of the lighter hydrogen isotope by bacterial processes such as methanogenisis during breakdown of organic matter.

 $\delta D$  values indicate that the slough either was more evaporated than the river or that the slough receives water from different sources than the river. The  $\delta D$  value for the river was in the range of background ground-water  $\delta D$  values (table 4). The  $\delta D$  value of the slough, however, was similar to the high  $\delta D$  values for ground water from downgradient wells within 20 meters of the landfill and downgradient wells greater than 4 meters in depth at sites 38 and 54 on either side of the slough, indicating that deuterium was enriched by evaporation in the slough or that some of the deeper leachate-contaminated ground water upgradient of the slough discharged into the slough.

## **Indicators of the Leachate Plume**

Some constituents measured in ground-water for this investigation are potential indicators of leachate contamination. Potential indicators that could be used to differentiate leachate contaminated water from uncontaminated ground water of the alluvial aquifer include specific conductance, chloride, alkalinity, dissolved organic carbon, boron, and  $\delta D$ .

Specific conductance in water collected downgradient from the landfill was significantly greater than in water collected from uncontaminated background wells (table 3). The specific conductance of the leachate contaminated ground water resulted from alkalinity and, to a lesser extent, from chloride concentration.

Chloride concentration, like specific conductance, was significantly greater in water collected downgradient from the landfill than in water collected from uncontaminated background wells (table 3). Chloride is a conservative indicator (Baedecker and Back, 1979, p. 436), and only dilution should decrease the concentration. However, there are potential sources of chloride other than the landfill, including oil field brine and water from the slough upstream of the landfill.

Additional indicators are needed to differentiate greaterthan-background specific conductance or chloride concentration values resulting from leachate contamination from increased values resulting from other contaminant sources. Alkalinity is a good indicator of leachate because it was significantly greater in leachate contaminated ground water than in background ground water or from PS06, which was 290 milligrams per liter (table 3). A combined analysis of specific conductance or chloride and alkalinity differentiates oil-field brine contaminated water from leachate contaminated water (fig. 11).

Dissolved organic carbon and boron also are good indicators of leachate contamination. The gradual decrease in dissolved organic carbon and boron concentrations in shallow ground water from the transition zone, through the slough, and into ground water downgradient of the slough suggests that these constituents are affected primarily by dilution (figs. 19 and 24). The concentrations also may be affected by bacterial degradation of dissolved organic matter or by adsorption of boron on aquifer clay minerals. Dissolved organic carbon concentration was significantly greater in leachate contaminated ground water than in background ground water (table 3). This large difference in concentration range indicates that dissolved organic carbon may be the most effective indicator of the leachate plume. Dissolved boron also was significantly greater in leachate contaminated ground water than in background ground water and may be used to indicate leachate contamination (table 3). The use of boron as an indicator of landfill leachate was recently illustrated by Erdman and Christenson



**Figure 26.** Relation between  $\delta$  deuterium and d<sup>18</sup> oxygen for water samples from the Canadian River alluvium near the Norman Landfill during fall 1995.

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(2000), who reported that boron taken up from the ground water by vegetation can be used to locate a leachate plume by collecting vegetation samples rather than ground-water samples.

 $\delta$  values for isotopes of hydrogen and oxygen also can be used to differentiate between leachate-contaminated water, water from the slough, and noncontaminated ground water because the  $\delta$ D values of leachate-contaminated water are shifted upward 2 to 30 per mil by deuterium enrichment.  $\delta$  values for most contaminated water plot above the regional ground-water line (fig. 26). Contaminated water downgradient of the slough plots more than 2 per mil to the right of the line, indicating enrichment in <sup>18</sup>O possibly as a result of evaporation in the slough. Enrichment greater than 0.5 per mil in <sup>18</sup>O relative to regional ground water may be an indicator of recharge from the slough in water downgradient from the slough.

## Summary

The Norman Landfill was selected for study as part of the U.S. Geological Survey Toxic Substances Hydrology Program in 1994. The closed and capped municipal landfill is a source of dissolved organic and inorganic compounds in ground water that have leached from the deposited waste and now extend toward the Canadian River as a plume of leachate.

Solid waste was deposited at many sites south of Bratcher-Miner road, north of the Canadian River, and west of Jenkins Avenue including an area extending about 350 meters west of Chautauqua Avenue. Waste was dumped and possibly buried with alluvial sand as of 1951. The City began dumping waste in trenches, dug down to the water table, and subsequently burying the waste with about 15 centimeters of alluvial sand in 1960. Trenching was abandoned in 1971 and lifts of sand were constructed to assure that waste was deposited at least 0.6 meter above the water table. The landfill was closed in 1985. A cap of 0.9 meter of compacted clay was completed by June 1989.

This investigation of the geochemistry of ground- and surface-water was done by sampling water from two existing stock wells, one domestic well, and temporary drive-point wells during fall 1995. Surface-water samples from the Canadian River and the slough also were collected. Twelve wells upgradient of the capped landfill, were selected for collection of background samples. Eight wells were sampled to characterize ground water downgradient from the landfill. Six of those wells were on a transect along a ground water flowpath that extended southwest from the sulfate-reduction zone at the base of the capped landfill. Two wells on either side of the slough along the transect were driven to increasing depths, ranging from about 2 meters below the water table to the base of the alluvium. These wells were used to measure variations in water quality with depth and to determine whether the plume extends southwest of the slough. The river was sampled upstream from the landfill to characterize background conditions. The slough was sampled between the two ground-water depth profiles to determine the quality of surface water receiving leachate.

Concentrations of most constituents were greater in water from downgradient wells than in water from background wells. The greatest specific conductance measured in water from a background well were for well PS06, which may be affected by a small oil-field brine spill. Shallow-ground-water chemistry changed with distance downgradient from the landfill and the slough affected ground-water chemistry downgradient.

Ground-water concentrations of most detected constituents increased with depth along vertical profiles on either side of the slough and then decreased slightly near the aquifer base, indicating that the leachate plume is present through the entire thickness of the aquifer and extends beyond the slough.

Concentrations of most constituents in the river were similar to those in background ground water. Water quality differs between the Canadian River, sampled upstream of the Norman Landfill and the slough, sampled downgradient of the landfill. Concentrations of most constituents were greater in the slough than in the river. However, pH and dissolved sulfate, arsenic, cadmium, nickel, and uranium concentrations were greater in the river.

 $\delta D$  and  $\delta^{18}O$  of water from background wells is similar to Central Oklahoma regional ground-water.  $\delta D$  of water from downgradient wells is enriched in deuterium relative to hydrogen by as much as 30 per mil, probably as a result of bacterial processes such as methanogenisis during breakdown of organic matter.  $\delta D$  values for deep downgradient ground-water samples were enriched relative to regional ground-water values, indicating the presence of leachate.

 $\delta D$  and  $\delta^{18}O$  values in the Canadian River and the slough plot to the right of the Central Oklahoma regional ground-water line, indicating that evaporation had affected the isotopic composition of the surface water. The  $\delta D$  value in the slough was enriched relative to background ground-water values, indicating that either deuterium in the slough was enriched by evaporation, some of the deeper leachate-contaminated ground water discharged into the slough, or that deuterium was enriched by processes occurring at the ground water-slough interface.

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