HYDROGEOLOGY, WATER QUALITY, AND POTENTIAL FOR TRANSPORT OF ORGANOCHLORINE PESTICIDES IN GROUND WATER AT THE NORTH HOLLYWOOD DUMP, MEMPHIS, TENNESSEE

Prepared by the
U.S. GEOLOGICAL SURVEY

in cooperation with the
CITY OF MEMPHIS, TENNESSEE

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By Robert E. Broshears and Michael W. Bradley

U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 91-4022

Nashville, Tennessee
1992

U.S. DEPARTMENT OF THE INTERIOR
MANUEL LUJAN, JR., Secretary

U.S. GEOLOGICAL SURVEY
Dallas L. Peck, Director

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CONVERSION FACTORS AND VERTICAL DATUM

Multiply	*By*	*To obtain*
inch (in.)	25.4	millimeter
inch per year (in/yr)	2.54	centimeter per year
foot (ft)	0.3048	meter
foot per day (ft/d)	3.528×10^{-6}	meter per second
foot squared per day (ft^2/d)	0.0929	meter squared per day
cubic foot per second (ft^3/s)	0.0283	cubic meter per second
mile (mi)	1.609	kilometer
square mile (mi^2)	2.590	square kilometer
acre	0.4047	hectare
gallon per minute (gal/min)	3.785	liter per minute
gallon per minute per foot [(gal/min)/ft]	0.2070	liter per second per meter

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

$$°F = 1.8 \times °C + 32$$

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.

Use of trade names in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

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ABSTRACT

Geologic, hydrologic, and water-quality data indicate that ground-water contamination is confined to shallow horizons within the unconfined aquifer underlying the North Hollywood Dump in Memphis, Tennessee. The dump is a closed municipal-industrial landfill that has been ranked as Tennessee's potentially most dangerous hazardous-waste site. Toxic constituents of concern at the dump include residues from the manufacture of organochlorine pesticides. The dump overlies an unconfined aquifer of unconsolidated sands, silts, and clays. During average hydrologic conditions, ground water flows beneath the dump at a mean velocity of approximately 3 feet per day and discharges to the Wolf River. Leachate from the dump mixes with underlying ground water, resulting in increased concentrations of dissolved solids and organic carbon downgradient from the dump. The mobility of chlordane, a representative organochlorine pesticide, is limited by its low solubility and its strong affinity for sand, silt, and clays of the aquifer. Degradation of chlordane may occur slowly, if at all, in the aquifer. Based on estimates of mean ground-water velocity and retardation of the pesticide due to sorption, mean travel times for chlordane migrating from the dump to the ground-water discharge zone are of the order of 50 to 500 years. Simulations of chlordane concentration resulting from the discharge of contaminated ground water and complete mixing in the Wolf River are sensitive to assumptions about chlordane persistence in the unconfined aquifer. If the half-life of chlordane in the aquifer is assumed to be 30 years or less, the simulated concentration of chlordane in the Wolf River under average flow conditions is less than the most stringent water-quality criterion.

INTRODUCTION

The North Hollywood Dump in Memphis, Tennessee, is a closed municipal-industrial landfill that has been ranked as Tennessee's potentially most dangerous hazardous-waste site (U.S. Environmental Protection Agency, 1986). Because ground water constitutes the water supply for almost a million people in the Memphis area, the potential movement of contaminants from the dump and into ground water has been a concern of local, state, and Federal health, regulatory, and scientific agencies. Although other toxic materials are known to have been placed at the dump, concern has focused primarily on residues from the manufacture of organochlorine pesticides. These compounds, which include chlordane, heptachlor, aldrin, endrin, dieldrin, and lindane, are highly toxic and potentially carcinogenic (U.S. Environmental Protection Agency, 1980a).

Several investigations before 1985 have documented the occurrence and distribution of organochlorine pesticides in soils, sediments,

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biota, surface water, and ground water at and near the North Hollywood Dump. In 1985, the U.S. Environmental Protection Agency (USEPA) requested studies to assess potential rates and directions of contaminant migration in local ground water. This assessment required a comprehensive hydrogeologic investigation, as well as a characterization of geochemical and geomicrobial factors influencing pesticide mobility in ground water. In 1985, the U.S. Geological Survey (USGS), in cooperation with the City of Memphis, began such an investigation. In 1987, the USGS began a supplemental cooperative research project with Memphis State University. The latter investigation focused on the behavior and fate of chlordane, a representative organochlorine pesticide. Well-construction, geologic, and hydrologic data collected during the projects were described by Broshears and Bradley (1990).

Purpose and Scope

This report describes the hydrogeology, water quality, and potential for the transport of organochlorine pesticides in ground water near the North Hollywood Dump. Understanding the potential for pesticide transport requires a description of the geology, hydrology, and chemistry of the local aquifer. This report provides an account of the direction and rate of ground-water flow near the dump and describes processes affecting the behavior of pesticides within the flow system. A characterization of all effects of the landfill on the quality of local ground water was beyond the scope of the investigation. The potential transport analysis was limited to chlordane, a representative organochlorine pesticide frequently detected in ground water near the dump.

Description of the Study Area

The city of Memphis is on the eastern alluvial plain of the Mississippi River and its tributaries (fig. 1). Relief in the area varies from 180 feet above sea level along the river bank to approximately 325 feet above sea level in local uplands. Precipitation averages 49 in/yr (National Oceanic and Atmospheric Administration, 1973). Surface drainage flows to local streams and to the Mississippi River.

Underlying the Memphis area is the Mississippi embayment, a sequence of unconsolidated sedimentary deposits occupying nearly 3,000 feet of a deep structural trough in the Gulf Coastal Plain (Boswell and others, 1965, 1968). Cradled in Paleozoic bedrock, the trough has its origin in a failed rift system (McKeown and Pakiser, 1982). Changes in sea level and continuing deformations of the failed rift have resulted in a sequence of marine transgressions into the embayment. This sequence is recorded stratigraphically as a series of sand-rich sediments deposited during periods of freshwater drainage through the area, alternating with clay-rich sediments deposited under shallow marine conditions. The sandy deposits constitute aquifers; the clay-rich horizons form confining units.

Several major stratigraphic units underlie Memphis (table 1). The most recent deposits are alluvial and fluvial and are characterized by a range of particle sizes. Within these deposits an unconfined aquifer averages 30 to 40 feet in thickness. Because of its limited thickness and inferior water quality, the unconfined aquifer is seldom used for drinking-water supply at Memphis. In upland areas, alluvial and fluvial deposits are blanketed by loess.

The unconfined aquifer is underlain by a confining unit that is often called the Jackson Formation but also includes the Cook Mountain and Cockfield Formations of the upper part of the Claiborne Group (Parks and Carmichael, 1990a). This unit, which may be as much as 360 feet thick, is clay-rich; where it is continuous and thick, it is relatively impervious to flow (Graham and Parks, 1986). In such areas, the Jackson Formation and the upper part of the Claiborne Group effectively confine the underlying Memphis

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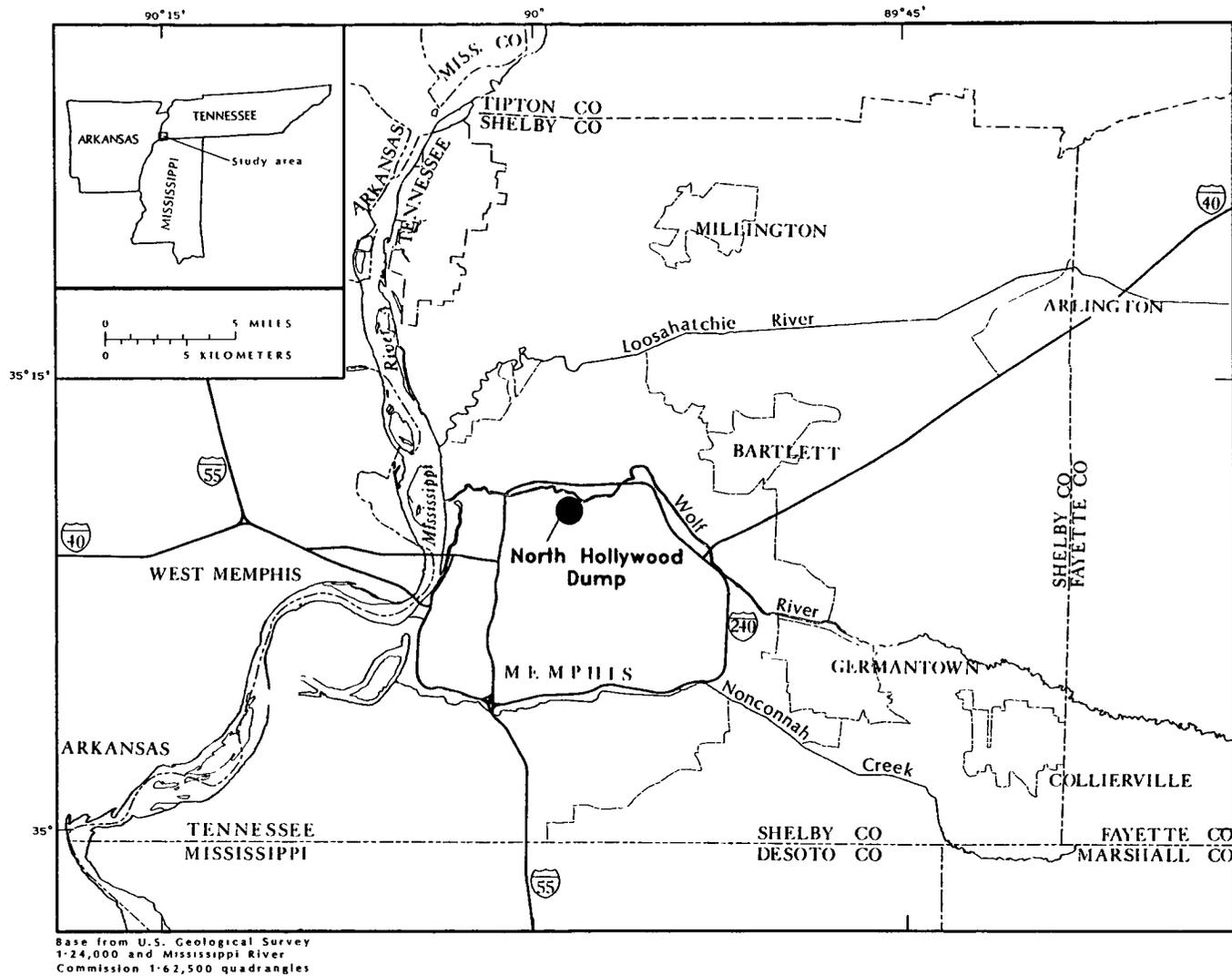
Figure 1.—Location of the North Hollywood Dump, Memphis, Tennessee.

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Table 1.--*Geologic units underlying the Memphis area and their hydrologic significance*

[Modified from Criner and Parks, 1976]

System	Series	Group	Stratigraphic unit	Thickness, in feet	Lithology and hydrologic significance
Quaternary	Holocene and Pleistocene		Alluvium	0-175	Sand, gravel, silt, and clay. Underlies the Mississippi River alluvial plain and the flood plains of other streams in the area. Supplies water to a few domestic and industrial wells. Could be an important source of water for irrigation and some industrial uses.
	Pleistocene		Loess	0-65	Wind-deposited silt; silty clay and minor sand. Forms a blanket over the fluvial deposits in upland area; topographically higher than alluvium. Thickest on the bluffs that border the Mississippi River alluvial plain; generally thinner towards the east. Not a source of ground water.
Quaternary and Tertiary (?)	Pleistocene and Pliocene (?)		Fluvial deposits (terrace deposits)	0-100	Sand and gravel; minor ferruginous sandstone. Underlies the upland areas in a broad, irregular belt east of the Mississippi River alluvial plain; may be locally absent. Supplies water to many shallow, small-capacity wells in suburban and county areas.
Tertiary	Ecocene	(?)	Jackson Formation and upper part of Claiborne Group ("capping clay")	0-360	Gray, bluish-gray, greenish-gray, and tan clay; subordinate beds of fine-sands and fine-grained lignite. Supplies water to some small-capacity wells. Generally considered to be of low perme-ability and to confine water in Memphis Sand. Absent in southeastern part of Memphis area.
		Claiborne	Memphis Sand ("500-foot" sand)	500-880	Fine to coarse sand; subordinate lenses of clay and minor amounts of lignite. Thick clay bed locally in lower part; coarse sand lenses locally at base. Very good aquifer supplies over 98 percent of water used in Memphis area.
		Wilcox	Flour Island Formation	160-350	Gray, greenish-gray, and brown carbonaceous clay. Locally contains fine-sand lenses and some lignite. Serves as lower confining bed for Memphis Sand and upper confining bed for Fort Pillow Sand.
	Paleocene		Fort Pillow Sand ("1,400-foot" sand)	210-280	Fine- to medium-grained sand; minor amounts of lignite and some clay lenses. Second most important aquifer but supplies less than 2 percent of water used in Memphis area.
			Old Breastworks Formation	200-250	Gray, greenish-gray, and brown carbonaceous clay. Contains some lignite and is sandy near top. Lower con-fining bed for water in Fort Pillow Sand.

Sand which comprises the Memphis aquifer. This aquifer is sometimes referred to as the "500-foot sand," an informal name that describes its mean depth (Parks and Carmichael, 1990b). Approximately 190 million gallons of water per day are withdrawn from the Memphis aquifer to provide essentially the entire water supply for the city. This pumpage represents nearly half of the ground-water use in Tennessee (Hutson and others, 1990). When wells were first drilled into the Memphis aquifer in the nineteenth century, water flowed at land surface, but cones of depression as deep as 100 feet have developed near pumping centers at Memphis (Brahana, 1982).

The water table in the unconfined aquifer is essentially a subdued replica of the topography, and ground water generally discharges to local surface drainage. As noted above, intensive pumping from the Memphis aquifer has lowered its potentiometric surface to such an extent that hydraulic gradients now favor flow from the unconfined aquifer downward to the Memphis aquifer. Graham and Parks (1986) estimated that heads in the unconfined aquifer near the North Hollywood Dump are about 60 feet higher than those in the Memphis aquifer. Ground-water flow modeling by Brahana (1982, and U.S. Geological Survey, written commun., 1990) indicates that over 30 percent of the pumpage from the Memphis aquifer may be derived by leakage from the unconfined aquifer. Little or no leakage was attributed to the area around the North Hollywood Dump (J.V. Brahana, U.S. Geological Survey, written commun., 1990), where the effective thickness of the intervening clay has been estimated to be at least 100 feet (Graham and Parks, 1986).

The North Hollywood Dump lies on the alluvial plain and old meanders of the Wolf River in north-central Memphis. The dump occupies approximately 70 acres immediately south of an area of bottomland hardwood forest along the Wolf River (fig. 2). The site is flanked to the southeast and west by sand and gravel dredge ponds and to the south by a mixed residential and commercial neighborhood. Begun as a neighborhood dumping ground in the 1930's, the site became a major municipal and industrial landfill by the mid-1950's (E.C. Jordan Co., 1984). Although records of waste emplacement are poor to nonexistent, chemical and industrial wastes are known to have been deposited at the site (Conestoga-Rovers and Associates, 1989). The dump officially closed in 1967, but unauthorized disposal occurred sporadically in subsequent years. Remedial action has included the maintenance of a vegetated cap over the surface of the dump since 1981.

Previous Investigations

The North Hollywood Dump has been the object of a series of investigations by local, state, and Federal groups. In 1979, a task force from the USEPA detected organochlorine pesticides in sediments and soils collected at the site (U.S Environmental Protection Agency, 1979, 1980b). These results motivated the formation of the Technical Action Group (TAG), a voluntary consortium including the City of Memphis, Velsicol Chemical Corporation, Memphis and Shelby County Health Department, the State of Tennessee, and the USEPA. TAG-sponsored teams confirmed the presence of substantial concentrations (greater than 100 milligrams per kilogram) of pesticides in surficial soils and sediments but detected only trace levels [generally less than 1 microgram per liter (μg/L)] in ground and surface water (E.C. Jordan Co., 1984). In 1980, the USGS and the Tennessee Valley Authority installed five observation wells at the margins of the dump. Organochlorine pesticides were detected in samples from all five wells; the highest concentration of a single compound was 5.6 μg/L of chlordane (Parks and others, 1982). Five more observation wells were installed in 1982, and ground-water samples were collected from all wells during the following year. In this sampling round, the highest concentration of a single compound was 2.4 μg/L of chlordane (Graham, 1985).

Figure 2.——Locations of well sites and surface—water
gaging stations at the North Hollywood Dump.

HYDROGEOLOGY

The ground-water flow system near the North Hollywood Dump consists of an unconfined aquifer underlain in sequence by a confining unit and a confined aquifer. Local flow in the unconfined aquifer is generally northward with discharge to the Wolf River.

Hydrogeologic Units

The hydrogeologic units of primary concern at the North Hollywood Dump are the alluvial and fluvial deposits of the unconfined aquifer, the confining unit of the Jackson Formation and the upper part of the Claiborne Group, and the underlying Memphis aquifer (fig. 3). The alluvial deposits occur at land surface in the Wolf River flood plain and below the fill material of the dump. The alluvium is 25 to 40 feet thick in this area and consists primarily of fine- to medium-grained sand with some coarse-grained sand and gravel and interbedded layers of silt, silty sand, and clay. Consistent with deposition in an alluvial environment, the stratigraphy features repeating sequences of gravel, sand, silt, and clay with traces of lignite. These deposits are in direct hydraulic connection with fluvial deposits.

The fluvial deposits in this area occur below the alluvium in the flood plain and also underlie the neighborhood south of the dump. These deposits consist primarily of medium- to coarse-grained sand with some gravel and minor amounts of silt and clay. The fluvial deposits are thickest south of the dump, where they range from 40 to 60 feet in thickness. Approximately 20 to 30 feet of loess overlies the fluvial deposits. The water table generally coincides with the base of the loess. Because of the direct hydraulic connection, ground water moves from the fluvial deposits northward into the alluvium.

Throughout the study area south of the Wolf River, the Jackson Formation and the upper part of the Claiborne Group are areally extensive. In this area they consist of relatively dense gray clay and silty clay with thin interbedded silt and fine-grained sand. The apparent continuity of confining clay in the area is consistent with the results of an independent study by Graham and Parks (1986), who characterized this layer throughout the Memphis area. The confining unit was less discrete at a borehole on the north side of the Wolf River (fig. 2, site 16). At this location, a borehole was drilled to a depth of 147 feet below land surface. Clay and silty clay horizons were encountered at 40 to 60 feet, 80 to 92 feet, and from 133 feet to the bottom of the borehole (Broshears and Bradley, 1990).

The waste material of the North Hollywood Dump lies directly on the alluvial deposits. At seven drilling locations, the fill ranged from 26 to 35 feet thick, with an average thickness of 28.5 feet. The waste material was generally above the water table. Saturation occurred locally only near the base of the waste. Broshears and Bradley (1990) described drilling methods and well construction. Three types of wells were drilled (fig.2). Observation wells were screened in the unconfined aquifer or at the top of the confining unit; these wells were used for the collection of ground-water samples and the periodic measurement of water levels. Leachate wells were screened within the waste material and were used for the collection of leachate samples. Water-level wells were equipped with recorders for the continuous measurement of water levels in the unconfined aquifer.

Bradley (1986) reported the results of a surface geophysical survey of the area. The lithology at selected sites and available geophysical logs were reported by Broshears and Bradley (1990).

Hydraulic Conditions and Properties

Hydraulic conditions and properties control ground-water flow within the unconfined aquifer. These properties include hydraulic conductivity,
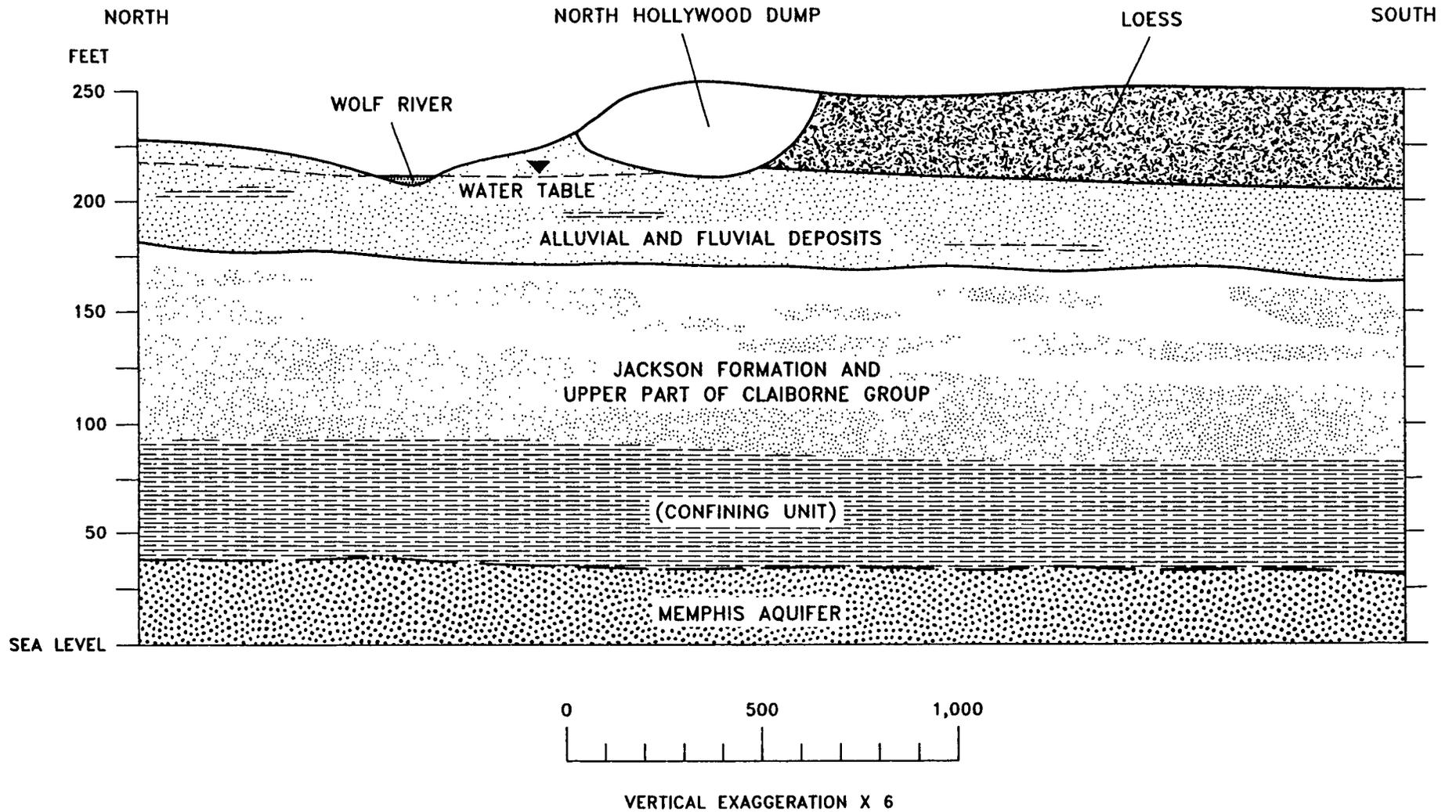
Figure 3.—Generalized hydrogeologic section at the North Hollywood Dump.

hydraulic gradient, and porosity. The hydraulic conductivities of similar alluvial materials in the Mississippi River valley have been estimated by several authors. Boswell and others (1968) cited 48 aquifer tests in alluvial materials from Missouri to Louisiana; based on these tests, mean hydraulic conductivity within the deposits was estimated at 220 ft/d. Newcombe (1976) reported a range of hydraulic conductivity of 90 to 400 ft/d (mean = 200 ft/d) for alluvial aquifers in Mississippi. Brahana and Mesko (1988) cited data from Broom and Lyford (1981) and Albin and Hines (1967) in reporting hydraulic conductivities ranging from 100 to 390 ft/d for alluvium in eastern Arkansas and southeastern Missouri. Based on aquifer tests and successful flow-model simulations, Hoos and others (1989) estimated an average hydraulic conductivity of 170 ft/d for the alluvial aquifer surrounding Reelfoot Lake in northwestern Tennessee.

Thirty-three slug tests were conducted by Conestoga-Rovers and Associates (written commun., 1985) at wells screened in the unconfined aquifer at the North Hollywood Dump. Based on these data and analyses using the method of Hvorslev (1951), estimates of hydraulic conductivity ranged from 1 to 180 ft/d. The short duration of these tests—most were completed within a few tens of seconds—compromises confidence in the estimates. Moreover, estimates of hydraulic conductivity from slug tests performed on partially penetrating wells in unconfined aquifers are always subject to error (Lohman, 1972).

The USGS measured specific capacity at 10 wells at the North Hollywood Dump. Values ranged from 0.9 to almost 8 gallons per minute per foot of drawdown (Broshears and Bradley, 1990). Using these 10 specific-capacity values, the mean value for hydraulic conductivity estimated from a modified Theis equation (Cooper and Jacob, 1946; Lohman, 1972) was 70 ft/d; from Boulton's equation (Boulton, 1954), 50 ft/d.

Considerable uncertainty is always involved in estimating hydraulic conductivity from specific-capacity data. This uncertainty relates to the effects of well losses, partial penetration, well development, and nonhorizontal flow.

Finally, an estimate of hydraulic conductivity was derived from the response of the unconfined aquifer to a flood wave in the Wolf River (fig. 4). Aquifer diffusivity (the ratio of the transmissivity to the storage coefficient) was estimated using the method of Pinder and others (1969), as described by Grubb and Zehner (1973). The sequence of observed heads at two wells equipped with continuous water-level recorders was compared with type curves representing a range of values for aquifer diffusivity (fig. 5). Type curves for diffusivities of 10,000 and 15,000 ft²/d bounded the observed data. Given a mean saturated thickness of 30 feet, and assuming a storage coefficient (approximated by the porosity) of 0.2, the bounding diffusivities are consistent with a hydraulic conductivity of 70 to 100 ft/d.

Based on values from the literature for similar formations, estimates from local specific-capacity tests, and the results of the diffusivity analysis, the horizontal hydraulic conductivity for the unconfined aquifer at the North Hollywood Dump is estimated to be 80 ft/d. Uncertainty in this estimate is about a factor of two.

Potentiometric contours portray a water table sloping from south to north with steepening gradients and some northeasterly and northwesterly flow near the discharge zone at the Wolf River (fig. 6). Little or no differences in hydraulic head were observed at sites with wells screened at different depths. The contours are based on water-level measurements made during March 1986. During this month, water levels were stable and were representative of average values during the 1985-87 monitoring period. The average hydraulic gradient computed from these data at the dump is approximately 0.007.
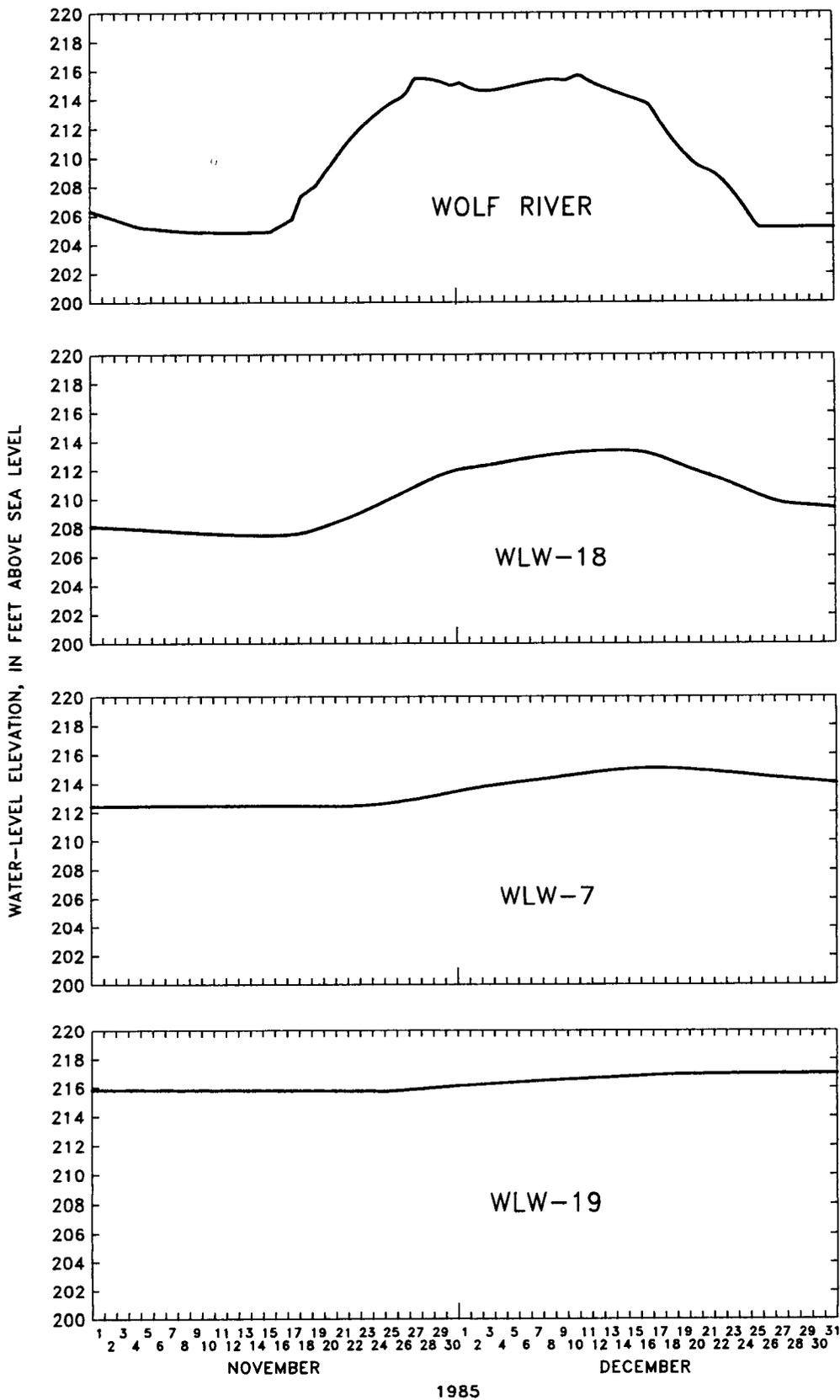
Figure 4.——Hydrographs of the Wolf River and water—level wells WLW—18, WLW—7, and WLW—19 at the North Hollywood Dump, November and December 1985.
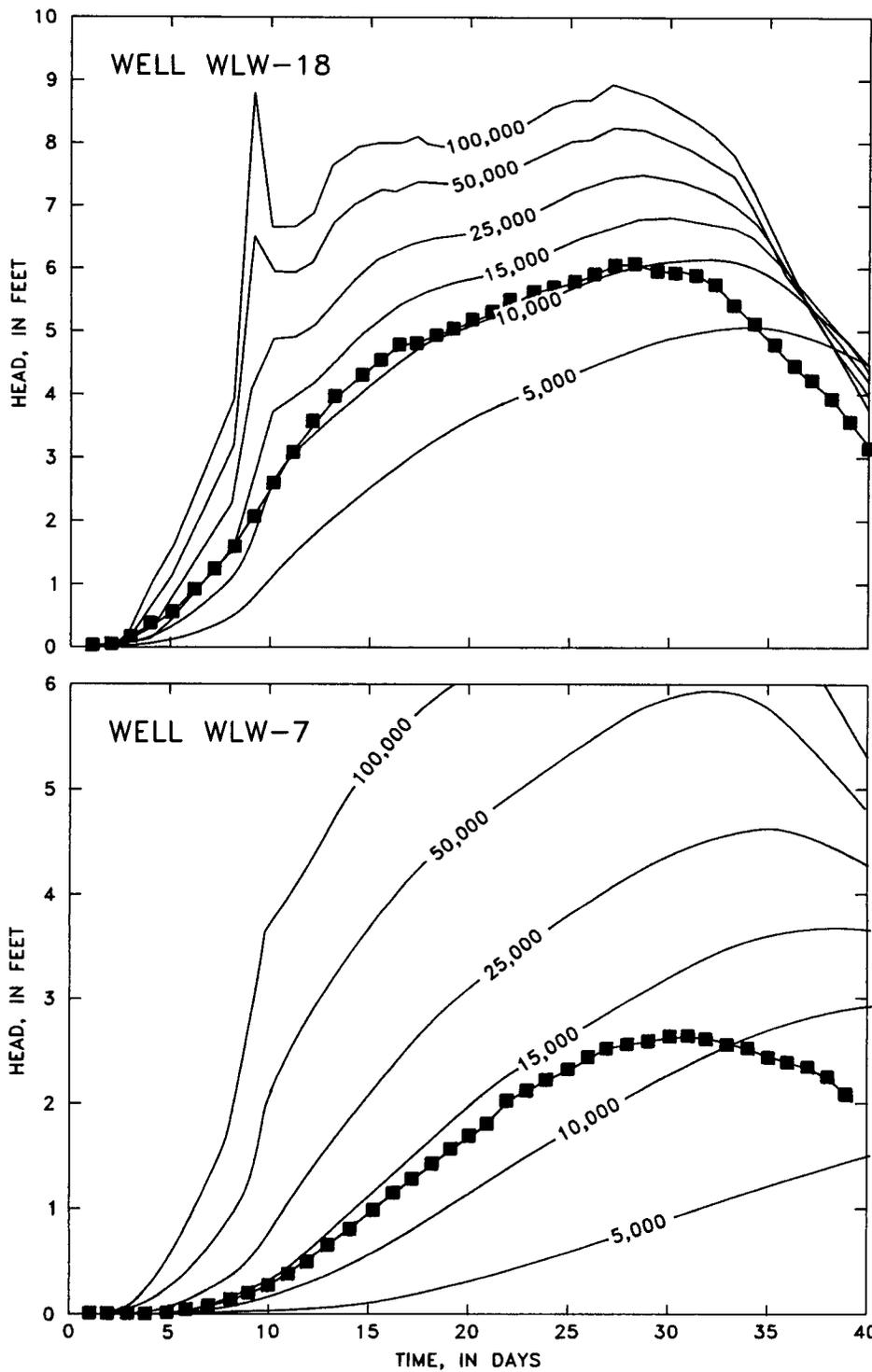
10

Figure 5.—Type curves for diffusivity and observed water levels at two wells at the North Hollywood Dump, November and December 1985.

35°11' 30"

I-240

212

Wolf   River

Oxbow
Lake

206
207   207
208   208   208
208
211   209   210   209   212
209   212   214   212   212   211
212   215   212
Active dredge
pond   216   214   213
214   217   216
Abandoned
dredge
pond
218   220
219   BELMONT CIRCLE   STREET   219
HEARD AVENUE   219   HOLLYWOOD
220
221   PEREZ AVENUE

0   1,000   2,000 FEET
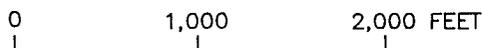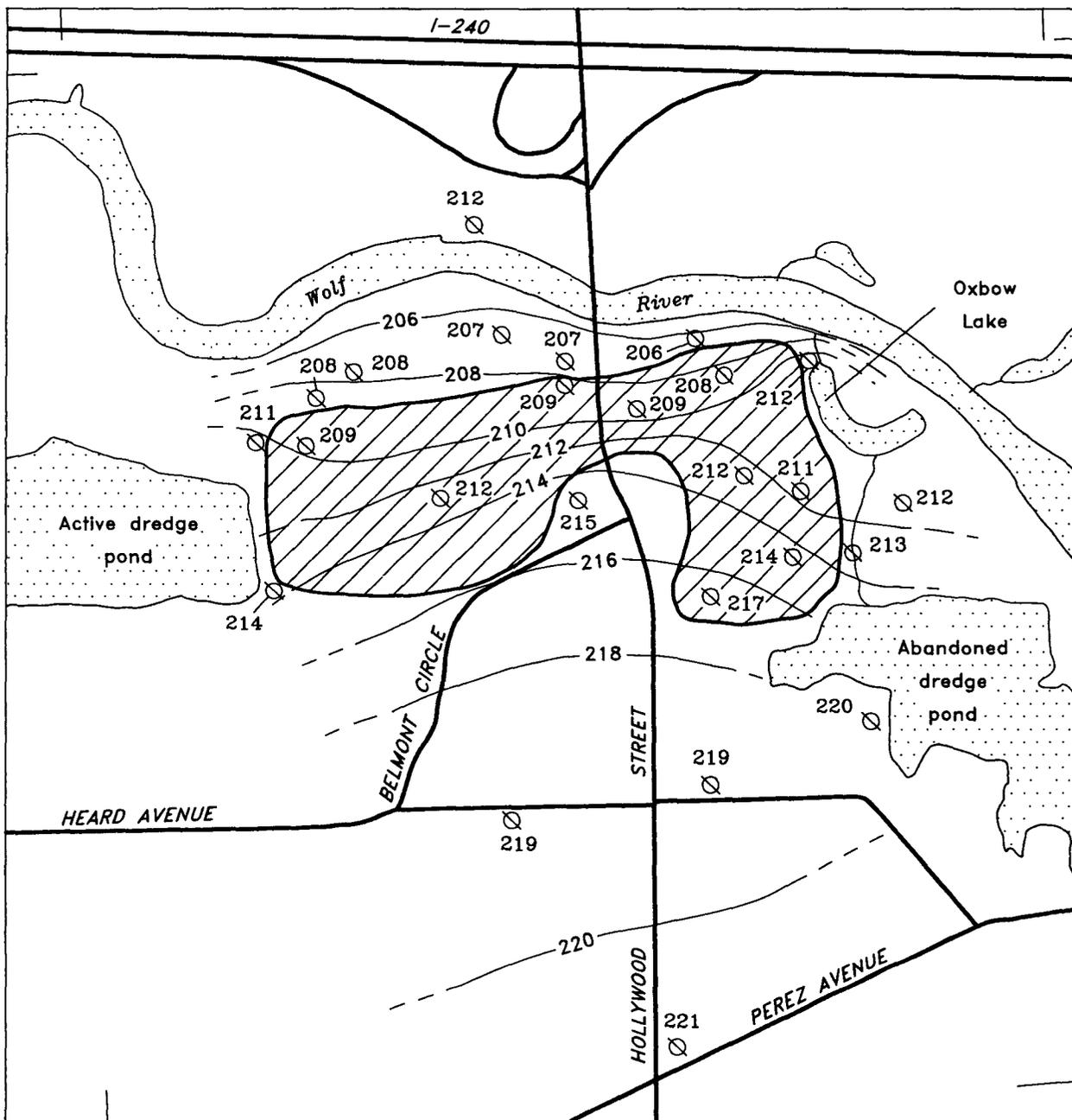
EXPLANATION

AREAL EXTENT OF NORTH HOLLYWOOD DUMP

—220—— WATER—TABLE CONTOUR——Shows altitude
of water table, March 1986. Dashed
where approximately located. Contour
interval 2 feet. Datum is sea level

221   OBSERVATION WELL AND WATER—TABLE
Q   ALTITUDE IN MARCH 1986

Figure 6.——Water—table contours at the North Hollywood Dump.

Porosity is the ratio of openings (voids) to the total volume of aquifer material. The porosity of unconfined aquifers in West Tennessee has been estimated at 0.1 to 0.3 (Moore, 1965). For the purposes of this investigation, a porosity of 0.2 was assumed for the study area. Uncertainty in this estimate is about a factor of two.

The vertical hydraulic conductivity of the clays beneath the unconfined aquifer was estimated by falling-head permeameter tests conducted on samples collected with Shelby tubes. Values for seven samples ranged from 6.9×10^5 to 5.6×10^{-3} ft/d; the median value was 3.2×10^{-4} ft/d (Construction Quality Consultants, written commun., 1986). These hydraulic conductivities are several orders of magnitude less than those estimated for the unconfined aquifer and indicate that an effective hydraulic barrier exists beneath the unconfined aquifer.

Ground-Water Flow

Based on Darcy's Law (Darcy, 1856), mean linear ground-water flow velocity is:

$$v = K / \Theta * dh/dl \qquad (1)$$

where

v = mean linear ground-water flow velocity,
K = hydraulic conductivity,
Θ = porosity, and
dh/dl = hydraulic gradient.

Any consistent dimensions are acceptable in equation 1.

Given estimates for the hydraulic conductivity, hydraulic gradient, and porosity of the unconfined aquifer at the North Hollywood Dump, ground water is estimated to flow from south to north beneath the dump at a mean linear velocity of 3 ft/d. Uncertainty in this estimate is about a factor of four.

Ground-water flow through the unconfined aquifer is anisotropic. Depositional processes in fluvial environments promote more continuity in grain size along horizontal dimensions than along vertical profiles. Because the hydraulic conductivity of sandy deposits greatly exceeds that of clay lenses, flow along horizontally continuous sandy deposits is favored over vertical flow, which is more likely to intersect a clay lens. Thus, ground-water flow near the North Hollywood Dump is predominantly horizontal over most of the area with some upward flow near the Wolf River.

Recharge to the unconfined aquifer in the study area occurs from precipitation and by ground water flowing into the area from the south. In the unsaturated zone, water moves downward through the loess and through the waste material of the dump. Water percolating through the waste is not expected to continue deeply into the aquifer but rather to flow more or less horizontally along shallow sandy horizons toward the discharge zone at the Wolf River.

Interactions between Ground Water and Surface Water

The distribution of hydraulic head in March 1986 (fig. 6) is representative of the ground-water flow system during most periods. Fluxes of water through the waste material at the North Hollywood Dump may increase substantially during times when the water table rises in response to intense rainfall. Stream and ground-water hydrographs for a major storm period at the dump during November and December 1985 are shown in figure 4. During this storm period, water levels increased steeply in the Wolf River and in the unconfined aquifer, and a significant volume of waste material became saturated. Based on the observed distribution of elevated water levels, an estimated 7 million cubic feet of waste material were saturated at the peak of this period. Assuming a porosity of 0.2, this volume is equal to more than 5 inches of water distributed equally

on the surface area of the dump. Annual mean infiltration of precipitation in the area has been estimated at 11 inches (E.C. Jordan Co., 1984). This estimate is consistent with a regional average for ground-water recharge of 11.5 inches per year made by Zurawski (1978) for areas of the Coastal Plain in West Tennessee. Thus, the November-December 1985 storm period produced a volume of water contacting the dump material that was a significant fraction of estimated annual infiltration.

WATER QUALITY

Ground-water samples were collected from observation and leachate wells near the North Hollywood Dump quarterly in 1986 and intermittently in 1987. Analytical results for samples collected during 1986 were reported by Broshears and Bradley (1990). Wells may be placed in two categories (table 2): those intercepting ground water hydraulically upgradient or lateral to the dump and those intercepting ground water hydrau-

Table 2.--*Designation of well location with respect to the potentiometric surface near the North Hollywood Dump*

Well location	Well name
Hydraulically upgradient or lateral to the dump	OW1A, OW1B, OW1C OW2A, OW2B, OW2C OW3A, OW3B, OW3C OW4A, OW4B OW12A, OW12B OW16A, OW16B TAG1, TAG2, TAG8 TAG9, TAG10
Hydraulically downgradient from the dump	LW5, OW5A, OW5B LW6, OW6A, OW6B LW7, OW7 LW8, OW8 LW9, OW9A, OW9B LW10, OW10 LW11, OW11 OW13 OW14A, OW14B OW17 TAG3, TAG4, TAG5 TAG6, TAG7

lically downgradient from the dump. Samples from wells in the latter category represent ground water that may be affected by leachate from the waste material.

Inorganic Constituents and Properties

Shallow ground water throughout the Memphis area is of the calcium bicarbonate type and tends to be very hard (Brahana and others, 1987); median hardness is about 200 milligrams per liter (mg/L) as calcium carbonate (Broshears, 1988). The median dissolved-solids concentration is near 300 mg/L, and the median specific conductance is about 500 microsiemens per centimeter (μS/cm). Iron concentration frequently exceeds 1,000 μg/L. In West Tennessee, use of the unconfined aquifer is limited because water of more favorable quality is available from deeper, confined aquifers (Hutson and others, 1990.)

Analyses of ground-water samples collected from wells upgradient from the North Hollywood Dump are consistent with regional values for the aquifer. For example, in samples taken from 14 wells at and near the site, specific conductance ranged from 313 to 642 μS/cm; the median value was 488 μS/cm (Broshears and Bradley, 1990).

The effects of the waste material are seen in samples from wells downgradient from the dump. For example, for samples collected in January 1986 from 19 downgradient wells, specific conductance ranged from 488 to 1,920 μS/cm (Broshears and Bradley, 1990). The median value was 1,270 μS/cm.

A rigorous characterization of the water quality of landfill leachate was beyond the scope of this investigation. However, a limited number of field measurements and laboratory analyses were conducted on leachate samples from two wells (LW5 and LW7). The specific conductance of samples taken from these wells commonly exceeded 2,000 μS/cm (U.S. Geological Survey, unpublished data). Analysis of a single sample

from LW5 for all compounds on the Appendix VIII list (U.S. Environmental Protection Agency, 1981) resulted in the detection of three polynuclear aromatic hydrocarbons and three organochlorine pesticides, as well as elevated concentrations of several trace elements (Environmental Testing and Certification, written commun., 1985).

The distribution of specific conductance is consistent with the flow of ground water in a northerly direction beneath the dump, recharge by landfill leachate with elevated concentrations of dissolved ionic constituents, and discharge of ground water at the Wolf River (fig. 7). The mean value of specific conductance for samples collected in June 1987 at two wells north of the Wolf River was 615 μS/cm. This value is similar to that for wells south of the dump and suggests that there may be little or no effect by the dump on water quality north of the Wolf River.

To further confirm the conceptual model that leachate from the North Hollywood Dump travels northward along shallow horizons in the unconfined aquifer, a drive-point sampling campaign was conducted in June and September 1987. Drive-point wells are an economical and efficient means of extracting a single ground-water sample with minimal disturbance to the subsurface environment (Kerfoot, 1984). At 24 sites along the Wolf River, a hollow, stainless-steel shaft 0.5 inch in diameter was inserted to depths of 5 to 12 feet. The shaft had slotted perforations at its lower end and a steel nipple at its upper end. A peristaltic pump withdrew ground water through tubing connected to the nipple. Samples were analyzed for three indicator variables: specific conductance (23 sites), chloride (21 sites), and total organic carbon (22 sites). Concurrently, these variables were also measured in samples from the network of observation wells. Distributions of chloride and total organic carbon are shown in figures 8 and 9. These distributions are consistent with the flow of ground water northward beneath the dump and the discharge of shallow, leachate-impacted water to the Wolf River. Particularly striking is

the limited lateral (east-west) extent of water-quality degradation. Values for all three variables declined sharply at the lateral margins of the discharge zone. At the eastern margin, for example, specific conductance declined from 1,550 to 144 μS/cm within a linear increment of 90 feet (fig. 7).

Increases in specific conductance in ground-water samples collected downgradient from the North Hollywood Dump are attributable to elevated concentrations of common ionic constituents. In comparisons between samples from wells upgradient and downgradient from the dump, most common ions increased in median concentration by 50 percent or more (fig. 10). Based on a ranked-sum test, statistically significant (alpha less than 0.01) increases (downgradient concentrations greater than upgradient or lateral concentrations) were observed for dissolved solids, specific conductance, sodium, potassium, and chloride. The latter constituents are highly soluble and are among the most abundant elements in municipal wastes (Tchobanoglous and others, 1977).

The distribution of sulfate ion exhibited a striking pattern. Upgradient from the dump increasing specific conductance was generally associated with increasing sulfate concentration, whereas downgradient from the dump, the reversed pattern was evident (fig. 11). Sulfate was the only major constituent that decreased in median concentration downgradient from the dump. This decrease is attributed to the reduction of sulfate under anaerobic conditions and is consistent with sulfide odors that were common during sampling from downgradient wells.

Samples from seven wells downgradient from the dump were analyzed for minor inorganic constituents, including most of the priority-pollutant trace metals (Broshears and Bradley, 1990). The only constituent present in a concentration exceeding the health-related maximum contaminant level established by the State of Tennessee for public water supply was barium at well OW14B. This well is screened 14 to 19 feet
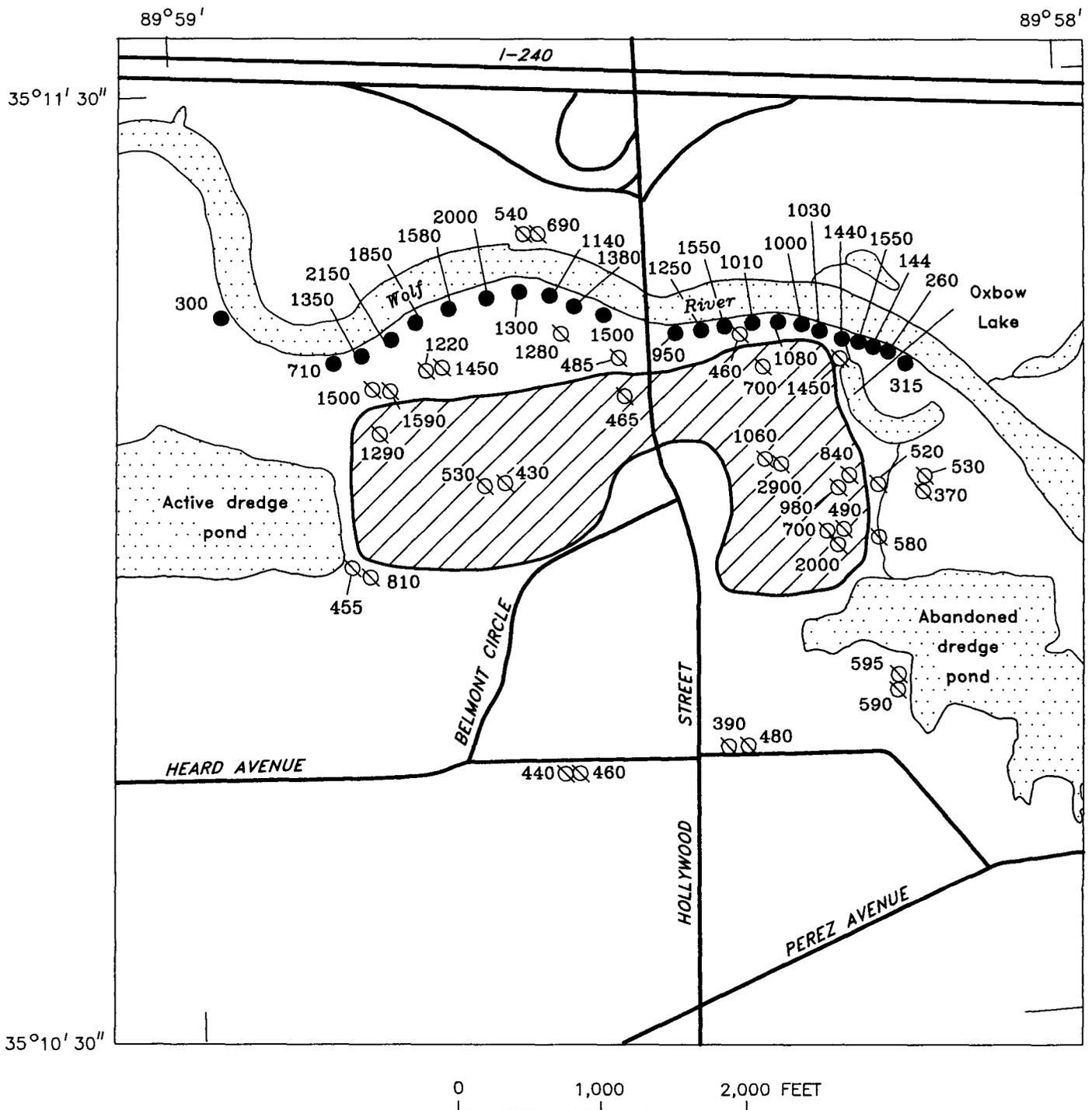
Figure 7.——Distribution of specific conductance in shallow ground water near the North Hollywood Dump, June and September 1987.

Figure 8.——Distribution of chloride concentration in shallow ground water near the North Hollywood Dump, June and September 1987.
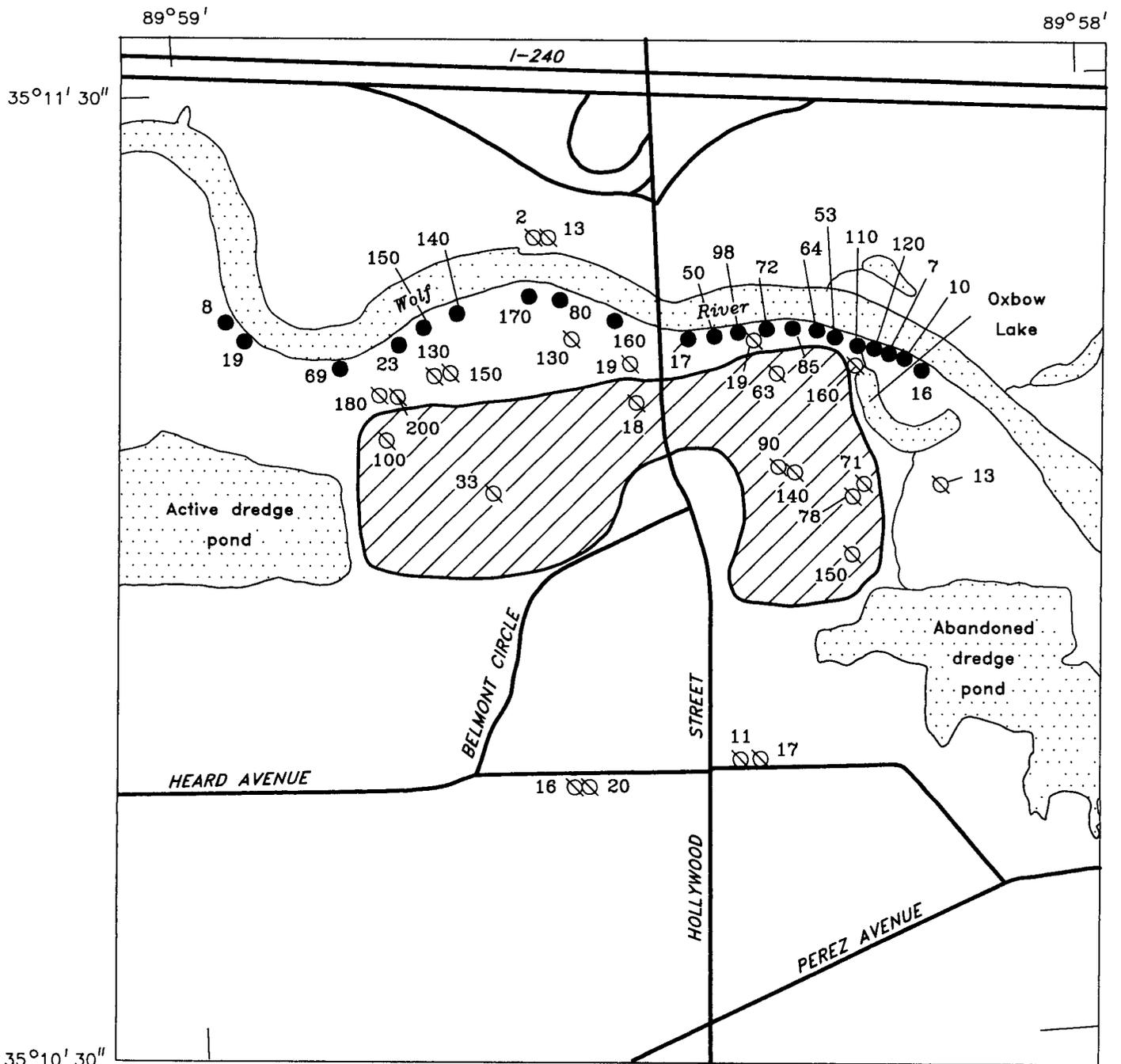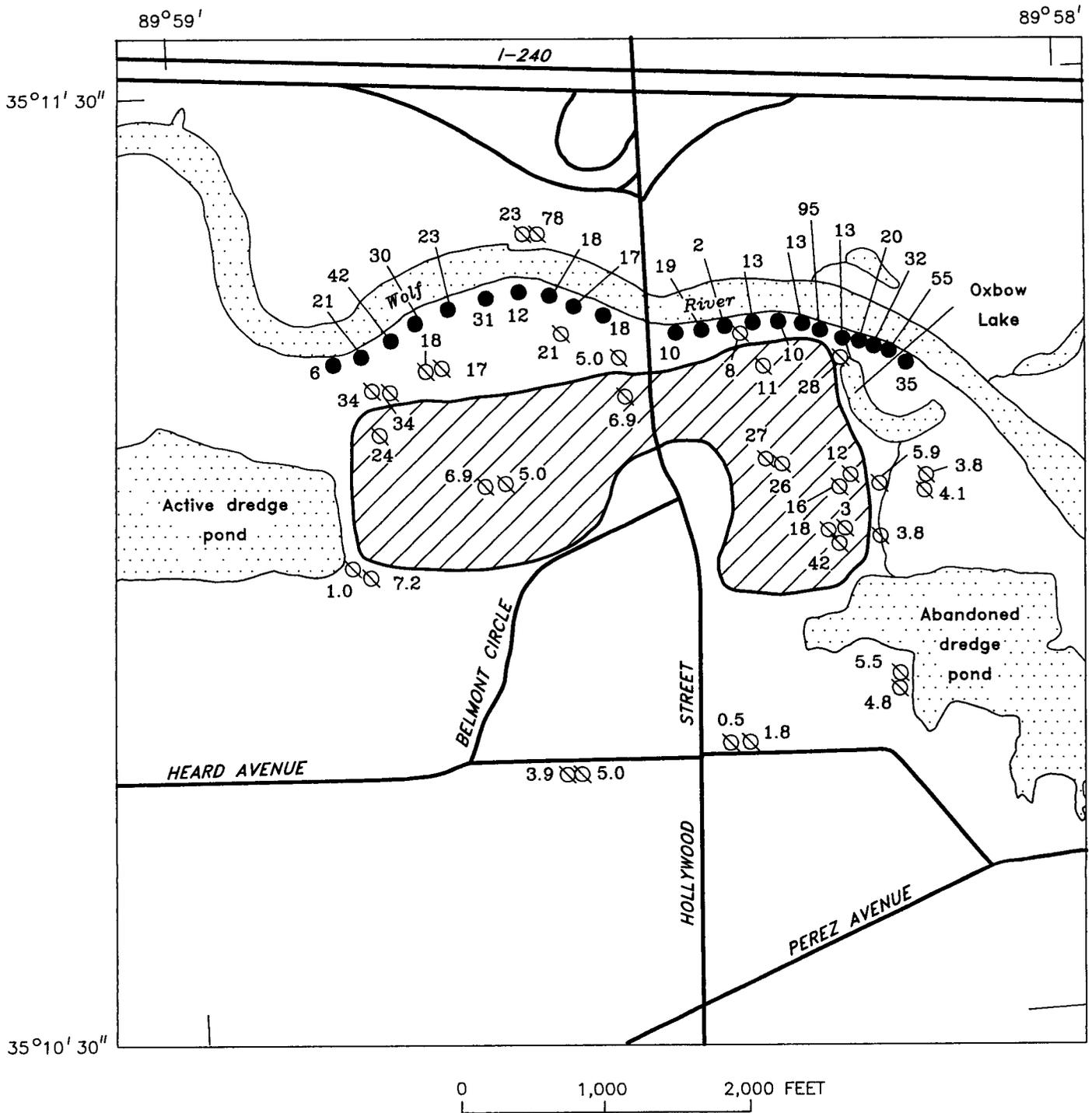
EXPLANATION

	AREAL EXTENT OF NORTH HOLLYWOOD DUMP
16 ⊘	OBSERVATION WELL AND CHLORIDE CONCENTRATION OF GROUND WATER, IN MILLIGRAMS PER LITER
69 ●	DRIVE—POINT WELL AND CHLORIDE CONCENTRATION OF GROUND WATER, IN MILLIGRAMS PER LITER

Figure 9.——Distribution of total organic carbon in shallow ground water near the North Hollywood Dump, June and September 1987.
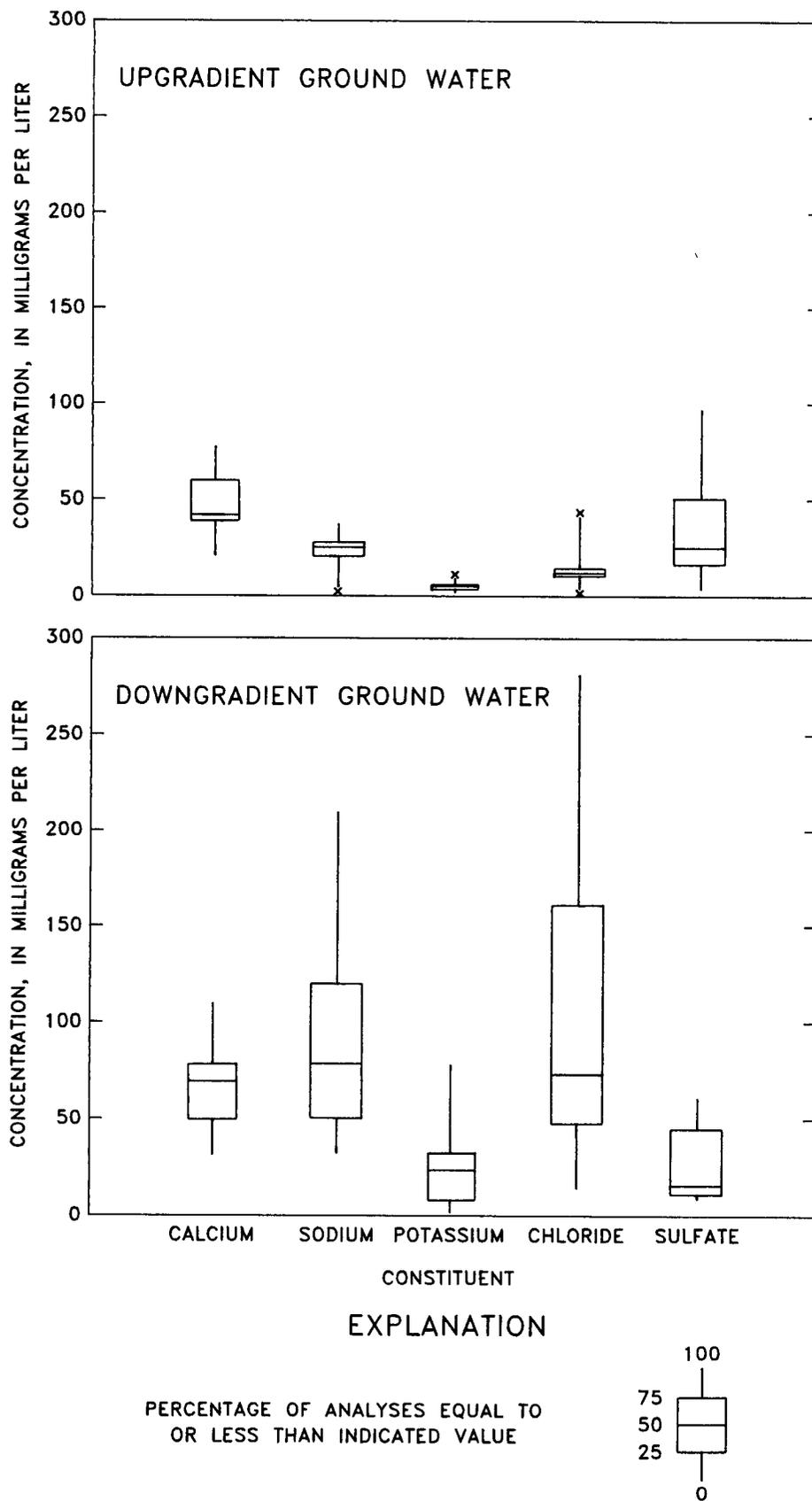
Figure 10.——Boxplots of selected major ionic constituents in ground water upgradient and downgradient from the North Hollywood Dump, January 1986.
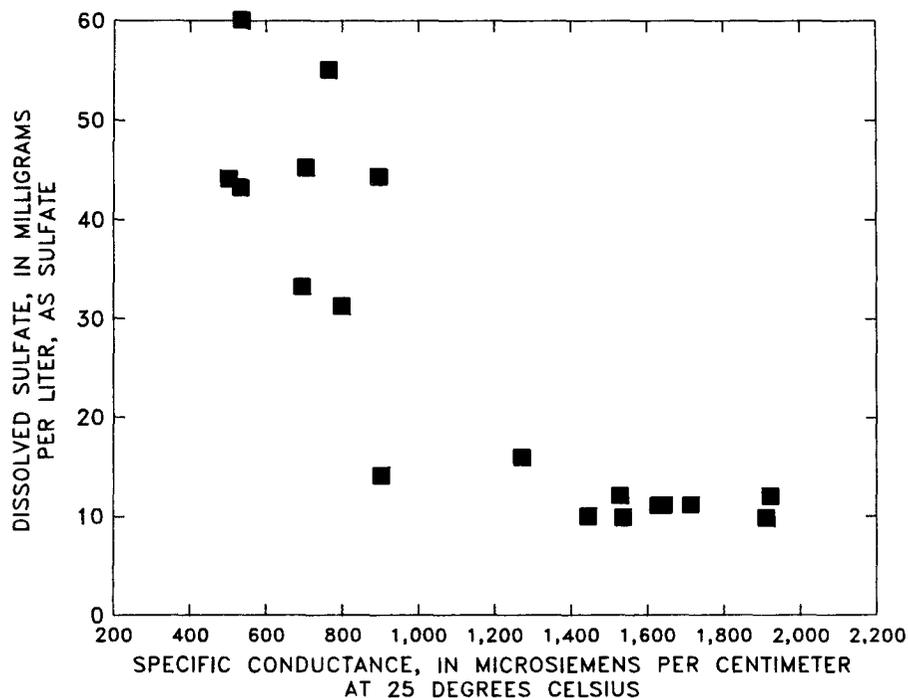
Figure 11.——Relation between sulfate concentration and specific conductance in ground water downgradient from the North Hollywood Dump, January 1986.

below land surface immediately north of the western side of the dump (fig. 2). The dissolved barium concentration at this well was 1,700 μg/L; the maximum contaminant level is 1,000 μg/L (Tennessee Department of Health and Environment, 1988). The concentration of dissolved arsenic was 44 μg/L in a sample from well TAG4; this value is near the maximum contaminant level of 50 μg/L.

Samples from six of the seven wells had concentrations of dissolved iron and manganese that exceeded secondary drinking-water regulations of the State of Tennessee (Tennessee Department of Health and Environment, 1988). The concentration of iron was more than 10,000 μg/L in samples from wells TAG4, OW14B, and OW10; manganese concentration exceeded 1,000 μg/L in samples from six of seven wells. These values suggest a reducing environment in the ground water, as is common beneath landfills. Reddish-

brown stains on the sands of the ground-water discharge zone along the Wolf River indicate iron oxidation and subsequent precipitation of iron oxides and oxyhydroxides.

Of particular interest are analyses of water from well TAG3, which is screened 24 to 29 feet below land surface between the northeast side of the dump and the Wolf River (fig. 2). Concentrations of major constituents in water sampled from this well did not differ markedly from concentrations observed in ground water upgradient from the dump (Broshears and Bradley, 1990). For example, the January 1986 value for specific conductance was 488 μS/cm. These observations probably reflect that the screened interval in TAG3 intersects a slightly deeper flow path that is not affected by leachate from the dump. Leachate from the dump apparently is limited to shallow horizons within the unconfined aquifer, and ground water that has been affected by leachate

flows more or less horizontally before discharging to the Wolf River.

Delineation of Water-Quality Degradation with a Bioassay

The response of bacteria to toxic constituents in water collected from drive-point wells and permanent wells was used as an indicator of water-quality degradation. The bioassay exploited the negative chemotaxis of fluorescent bacteria. By definition, bioassays are responsive to the biologically available fraction of toxic constituents. These tests are potentially powerful integrators of water-quality information, and they permit screening for the potential presence of toxic constituents at trace levels prior to committing resources to expensive laboratory analyses. Bacteria can be especially useful organisms for bioassays because of their ease of culture and rapid response times.

A full discussion of procedures for the bioassay is available in Booth (1990). Briefly, water samples were loaded into a series of microcapillary tubes. These tubes, together with appropriate control waters, were inserted into a dish containing an aqueous culture of *Pseudomonas aeruginosa*, a fluorescent bacterium common in soil and water. After an incubation of 10 to 30 minutes, concentrations of bacteria inside the microcapillary tubes were measured, either with a fluorometer or by standard plate count. A comparison of population density inside the sample and control tubes provided an index of negative chemotaxis by the bacteria, and thus an index of the apparent toxicity of the sample. Data were expressed by assigning a value of 100 to the control tubes. Values less than 100 in sample tubes indicated the relative avoidance of the sample by the bacteria.

All assays were conducted in duplicate on single samples from each well; mean values are displayed in figure 12. Samples taken from upgradient wells exhibited only mild avoidance by the bacteria (index greater than or equal to 90), and only slightly greater avoidance was evident in samples collected laterally from the dump. The response was most marked in a leachate sample in the southeastern part of the dump (index = 46) and in drive-point samples from the middle of the discharge zone along the Wolf River. These results provide additional support for the conceptual model of shallow, leachate-impacted ground water at the North Hollywood Dump.

Organochlorine Pesticides

Ground-water samples were collected quarterly in 1986 from the network of monitoring wells. Analyses for organochlorine pesticides and other priority pollutants were performed by the Velsicol Chemical Corporation. These analyses have been reported by Conestoga-Rovers and Associates (1989) and will not be discussed in this report. In October 1986, samples from 35 wells were analyzed for organochlorine pesticides by a USGS laboratory. Results of these analyses were reported by Broshears and Bradley (1990). Detectable total-recoverable concentrations of organochlorine pesticides were identified in 17 of the 35 samples. Seven compounds were detected; in order of frequency of detection, these compounds were chlordane, aldrin, heptachlor epoxide, dieldrin, DDD, heptachlor, and perthane. The highest concentration of a single compound was 11 μg/L of chlordane in the sample from well TAG5.

Distributions of total-recoverable pesticide concentration detected in these samples do not form a distinct pattern (fig. 13). In general, however, frequency of detection and summed concentrations of all detected pesticides increased from south to north across the study area. Pesticides were detected in water from 5 of 13 wells located upgradient or lateral to the dump, and in this group the maximum summed concentration was 0.43 μg/L at well TAG1 (see fig. 2). The occurrence of pesticides in surficial soils in the
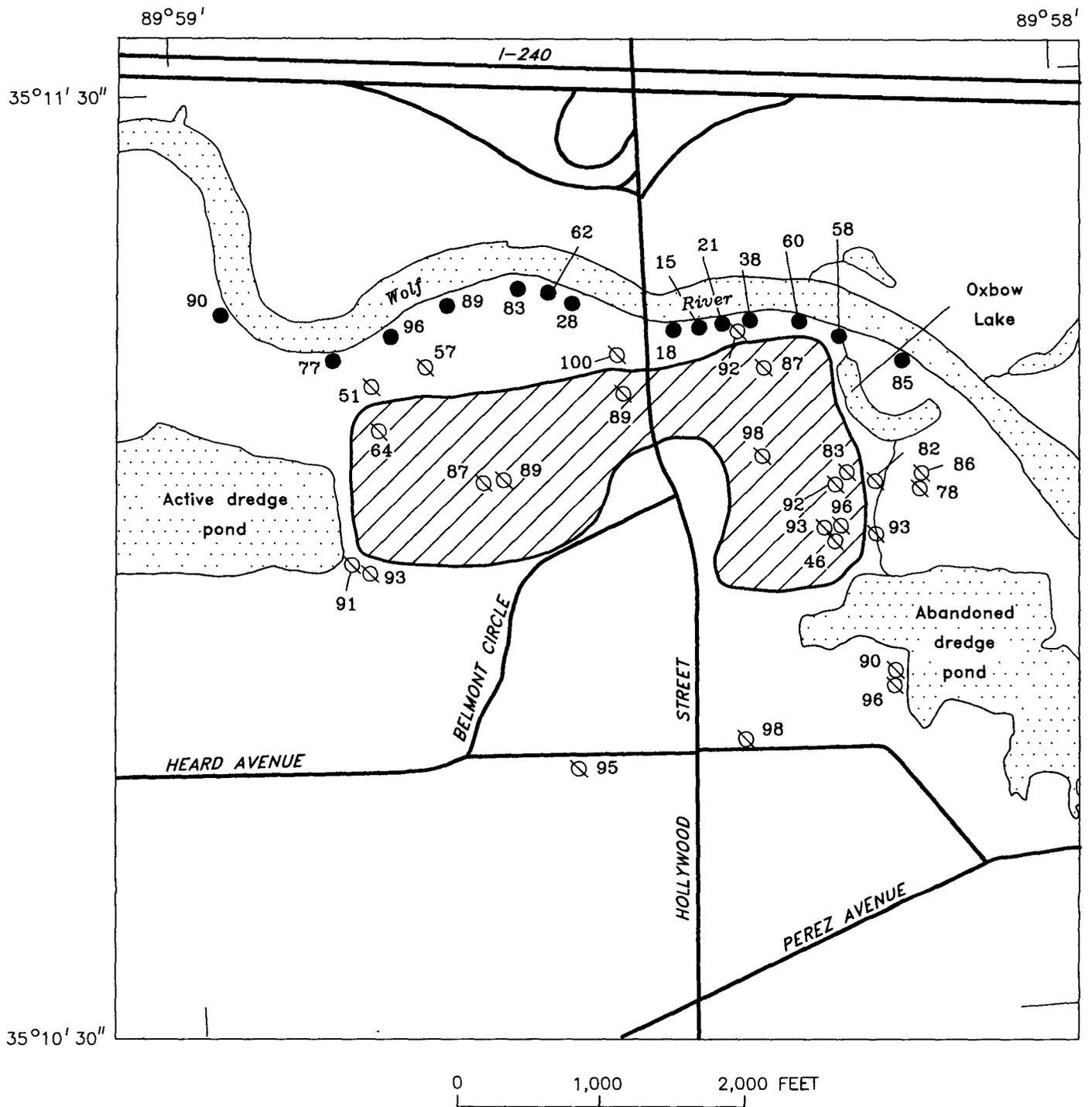
Figure 12.—Distribution of the toxicity index in shallow ground water near the North Hollywood Dump, June and September 1987.
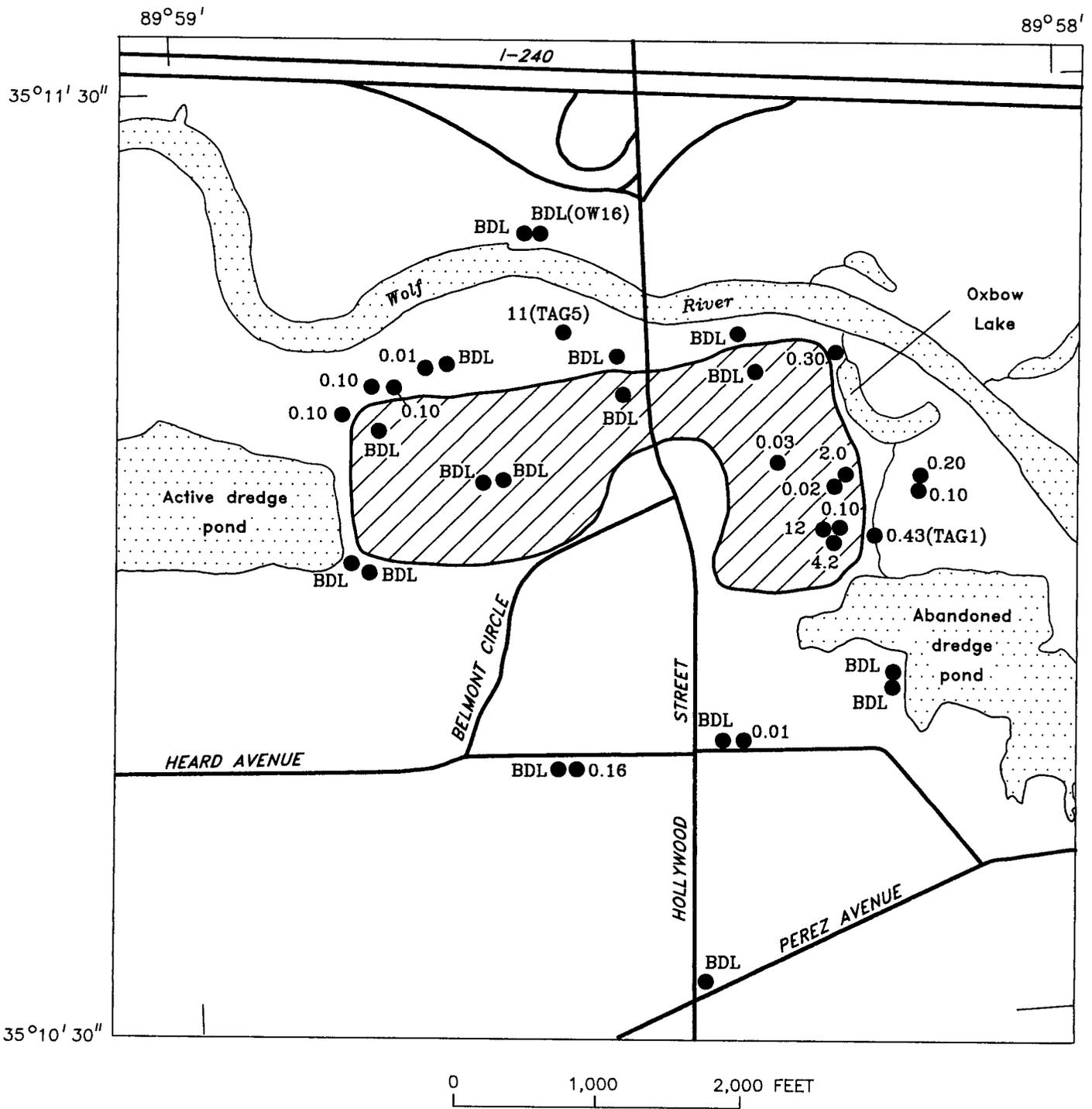
Figure 13.——Distribution of concentrations of total—recoverable organochlorine pesticides in ground water near the North Hollywood Dump, October 1986.

neighborhood south of the dump has been documented by the USEPA (1980b). The source of these compounds may be long-term residential or commercial application for termite control. Downgradient from the dump, the frequency of detection increased to 12 of 20 wells; the maximum concentration of summed pesticides also increased to over 11 μg/L in water from well TAG5. Pesticides were not detected north of the Wolf River in samples taken from wells OW16A and OW16B.

POTENTIAL FOR PESTICIDE TRANSPORT

The diffuse pattern of pesticide concentrations in the water of the unconfined aquifer near the North Hollywood Dump reflects the chemical behavior of these compounds. Organochlorine pesticides are sparingly soluble and have a strong tendency to associate with the surfaces of sediments and aquifer solids (Moore and Ramamoorthy, 1984). Such properties greatly restrict the mobility of the compounds in ground water, and they are seldom present in ground water at concentrations that are high relative to their analytical detection limits. Because many years have elapsed since the disposal of pesticides at the dump, the fact that these compounds are still present at detectable concentrations testifies to an apparent low rate of degradation.

The physical and chemical properties of chlordane are representative of organochlorine pesticides, and chlordane is the pesticide detected most frequently and in the highest concentrations in ground water near the North Hollywood Dump. Combined with the preceding account of hydrogeology and water quality at the North Hollywood Dump, an understanding of chlordane behavior in ground water permits development of a conceptual model for the transport of organochlorine pesticides in ground water near the site.

Properties of Organochlorine Pesticides

Organochlorine pesticides include DDT and related compounds, lindane, and cyclodienes (chlordane, heptachlor, aldrin, dieldrin, and endrin). These pesticides derive their physical and chemical properties from their relatively high molecular weights, low polarities, and high degree of chlorination (Callahan, 1979). Such characteristics are associated with low solubilities in water, high octanol-water partitioning coefficients, and resistance to degradation by hydrolysis or microbially mediated oxidation-reduction reactions.

Values for physical, chemical, and biological properties of selected organochlorine pesticides are shown in table 3. For some characteristics, wide ranges in reported values reflect the inherent difficulty in measuring aqueous concentrations of compounds whose solubility is near analytical detection limits. These difficulties are particularly grave for compounds such as organochlorine pesticides that sorb strongly to the surfaces of laboratory vessels.

Values for variously defined thresholds of toxicity for chlordane are shown in table 4. These thresholds address both acute and chronic effects, and target organisms include humans, other mammals, and aquatic life. The lowest thresholds relate to the potential carcinogenicity of chlordane. Uncertainty in establishing a dose-effect relation for carcinogens at very low doses is a central problem in toxicology. Typically, the effects of high doses over relatively short time periods on laboratory animals (often inbred mice) must be extrapolated to low, chronic doses to human beings (Crouch and Wilson, 1982). Variations in the resulting thresholds (table 4) reflect differences in the mathematical form of the extrapolation. Evaluation of the thresholds in table 4 is beyond the scope of this investigation. It is simply noted that no threshold in the table is less than 0.00048 μg/L.

Table 3.—*Properties of selected organochlorine pesticides*

[µg/L, micrograms per liter; atm, atmospheres;
log K_{ow}, log of octanol-water partitioning coefficient]

Compound	Water solubility, in µg/L	Vapor pressure, in atm	Log K_{ow}	Bioconcentration factor	Half-life, in years
Chlordane	9[a] - 56[b]	$1.3 \times 10^{-6a,c}$	2.78[d] - 5.58[e]	14,100[c]	0.7[f] - 9.6[e]
Endrin	249[e] - 26[d]	2.6×10^{-10a}	3.21[g] - 5.60[h]	3,970[j]	0.4[g] - 11.8[k]
Heptachlor	30[e] - 180[b]	$3.9 \times 10^{-7a,d}$	3.87[g] - 5.73[e]	11,200[l]	0.8[m] - 9.5[n]
Lindane	150[e] - 17,000[e]	1.2×10^{-8p}	2.81[g] - 3.89[q]	35 - 617[r]	0.03[e] - 4.2[k]

[a]Verscheueren (1983)
[b]Callahan (1979)
[c]USEPA (1980c)
[d]USEPA (1980d)
[e]Conway (1982)
[f]Eichelberger and Lichtenberg (1971)

[g]Rao and Davidson (1980)
[h]Neely and others (1974)
[j]USEPA (1980e)
[k]Nash and Harris (1973)
[l]USEPA (1980f)
[m]Brown (1978)

[n]Khan (1980)
[p]Mackay and Leinonen (1975)
[q]Veith and others (1979)
[r]USEPA (1980g)
[s]MacRae and others (1967)

Factors Influencing Pesticide Transport

Consideration of the transport potential of a constituent in an aquifer must include a minimum of four factors: the source term, the ground-water velocity, any differential mobility of the constituent relative to ground-water velocity, and the persistence of the constituent in the system.

The source term may be limited by solubility. If only dissolved constituents are mobile, then the maximum possible loading to the ground-water flow system is equal to the product of the solubility of the constituent and the total flux of water through the source material. For example, as described in a previous section, recharge by precipitation at the North Hollywood Dump has been estimated at 11 in/yr. This rate applied over the 70 acres of dump area would produce an annual recharge volume of nearly 3 million cubic feet. If this volume of water were saturated with chlordane at a concentration of 10 to 100 µg/L, the annual loading to the aquifer due to recharge by precipitation would be about 800 to

8,000 grams. By the same rationale, the loading due to the November-December 1985 storm period would be about 400 to 4,000 grams.

Dissolved constituents that do not interact with aquifer solids move at the same velocity as the ground water. Tritium and conservative anionic tracers like bromide behave in this manner. Compounds like organochlorine pesticides, however, move much slower than the mean ground-water velocity. As these compounds move in an aquifer, they partition themselves between the mobile (aqueous) phase and the immobile (aquifer-solid) phase; their net mobility is a function of the relative time spent in each phase. (The same principle applies to window-shopping while walking: If a person walks at 3 miles per hour but spends one-third of his time window-shopping, his net velocity is only 2 miles per hour.) The degree to which mobility is retarded is a function of the compound's relative affinity for mobile and immobile phases and the relative availability of each phase.

25

Table 4.--*Thresholds of toxicity for chlordane*

[All values are in micrograms per liter, except as noted; LC-50, concentration lethal to 50 percent of test organisms; mg/kg/day, milligrams per kilogram per day]

Variable	Value	Reference
96-hour LC-50		
Penaeus duorum	0.4	Parrish and others (1976)
Pimephales promelas	36.9	Cardwell and others (1977)
120-hour LC-50		
Palaemonetes kadiakensis	2.5	Sanders (1972)
Lowest observed effect level, brook trout	.32	Cardwell and others (1977)
Lowest observed adverse effect level, liver necrosis in rats	.045 mg/kg/day	Yonemura and others (1983)
Drinking water equivalent level, based on lowest observed adverse effect level in rats and safety factor	2	U.S. Environmental Protection Agency (1987).
Maximum contaminant level, public-water supply	2	U.S. Environmental Protection Agency (1989).
Carcinogenic potential, 10^{-6} risk based on drinking water and extrapolation by:		U.S. Environmental Protection Agency (1987).
Multi-stage model	.027	
Weibull model	.03	
Logit model	2	
Probit model	50	
Protection of fresh-water aquatic life, 24-hr average	.0043	U.S. Environmental Protection Agency (1980a).
Carcinogenic potential, 10^{-6} risk based on consumption of aquatic organisms	.00048	U.S. Environmental Protection Agency (1980a).

Estimates of partitioning affinity can be made from the results of batch sorption experiments (Lyman and others, 1982). In these experiments the constituent is placed in water, and aquifer solids are added. Following equilibration, the relative amount of compound in each phase is measured. The experiment is usually repeated over a range of initial concentrations, and the results are plotted as equilibrium isotherms. The partitioning behavior is often linear for sparingly soluble compounds. A linear distribution coefficient implies that, within a range of relevant aqueous concentrations, there is no upper bound on the capacity of the sorbent to bind the compound. In this case, a constant distribution coefficient can be calculated. The linear distribution coefficient is defined as:

$$K_d = C_s / C_w \qquad (2)$$

where

K_d = distribution coefficient,

C_s = concentration in solid phase at equilibrium, and

C_w = concentration in water at equilibrium

When the solid-phase concentration is expressed in micrograms per kilogram and the aqueous concentration is expressed in micrograms per liter, the distribution coefficient has units of liters per kilogram, or equivalently, milliliters per gram.

The relative availability of each phase in a ground-water flow system is simply the ratio of the bulk density of solids to the porosity. From this ratio and the distribution coefficient, a retardation factor can be calculated. The retardation factor is the ratio between the mean velocity of ground water and the mean velocity of the compound. For a compound exhibiting linear, reversible sorption, the retardation factor is defined as:

$$R = 1 + (\frac{\rho_d}{\Theta})K_d \qquad (3)$$

where
R = retardation factor,
ρ_d = bulk density of solids,
Θ = porosity, and
K_d = distribution coefficient

Note that for a compound that does not sorb to solids (that is, K_d = 0), the retardation factor is unity, and the compound moves at the ground-water velocity. For compounds with large distribution coefficients, the retardation factor can be quite large, and the mobility of the compound may be several orders of magnitude slower than the mean ground-water velocity. For example, assuming a bulk density of 2.0 grams per cubic centimeter (g/cm^3), a porosity of 0.2 milliliters per cubic centimeter (mL/cm^3), and a K_d value of 100 mL/g, the retardation factor is 1001.

Organochlorine pesticides in both dissolved and sorbed phases may be subject to degradation by physical, chemical, and (or) microbially mediated mechanisms. The results of several degradation studies on chlordane are shown in table 5. The reader is cautioned that most of these investigations relate to shallow soil environments where degradation mechanisms are substantially different from those operating in an aquifer. For example, removal by volatilization or by surface runoff in dissolved or sorbed phases contributes to the decline of concentrations of pesticides applied to surface soils, but these mechanisms are inhibited or inoperative in a ground-water system. Accordingly, degradation rates of chlordane in an aquifer may be much

different, and potentially much slower, than the values shown in table 5. On the other hand, anaerobic conditions in leachate-affected ground water may favor microbially mediated reductive dehalogenation, and the resulting dehalogenated ring structures might be more susceptible to further degradation if transported into an aerobic environment (Fogel and others, 1982). The values (table 5) are reported in half-lives, which are appropriate only for first-order degradation kinetics. In general, these values testify to the persistence of organochlorine pesticides in field situations and to probable persistence in ground water.

Inherent problems are encountered in conducting degradation experiments on compounds like organochlorine pesticides. In addition to the analytical problems cited above, the persistence of these chemicals makes lengthy experiments necessary. For example, to demonstrate a 10-percent decrease in a compound with a half-life of 10 years, an experimental incubation of 18 months is required. A 20-percent decrease would require incubation for 39 months. Because analytical uncertainties for organic compounds routinely exceed 10 to 20 percent (Wershaw and others, 1987), lengthy experiments are needed if statistically significant losses of the parent compound are to be observed.

Table 5.--*Reported environmental half-lives for chlordane*

Half-life, in years	Reference
0.7	Eichelberger and Lichtenberg (1971).
1.0	Brown (1978)
2-4	Khan (1980)
3.3	Rao and Davidson (1980)
7.9	Nash and Woolson (1967)
9.6	Conway (1982)

Experiments with Chlordane

To gain a greater understanding of general and site-specific processes controlling the fate of organochlorine pesticides in ground water at the North Hollywood Dump, a series of experiments was conducted. The experiments focused on the solubility, sorption, and degradation of chlordane.

Solubility experiments addressed the effects of dissolved organic matter and co-solvents on the solubility of chlordane in synthetic ground water and in ground water collected upgradient and downgradient from the North Hollywood Dump (L.R. Logan-Johnson, Memphis State University, written commun., 1989). Solutions were shaken with excess chlordane labeled with carbon-14; aliquots were extracted with hexane and analyzed by liquid scintillation. The solubility of chlordane in de-ionized water was 32 μg/L, a concentration within the range of reported values in the litera-

ture (table 3). The addition of humic substances to the water increased the observed solubility of chlordane (fig. 14); for example, at a dissolved organic carbon concentration of 66 mg/L, the observed solubility of chlordane was 470 μg/L. This influence was also observed in solubility experiments conducted with ground water collected near the North Hollywood Dump. Ground water collected upgradient from the dump had a total organic carbon concentration of 2 mg/L; the observed solubility of chlordane in this water was 52 μg/L. The observed solubility of chlordane increased to 150 μg/L in a downgradient groundwater sample with a total organic carbon concentration of 34 mg/L. Enhancement of the solubility of hydrophobic organic compounds by humic substances in solution has been documented by other authors (Chiou and others, 1987) and is attributable to the association of the compound with nonpolar domains within the aqueous organic material.
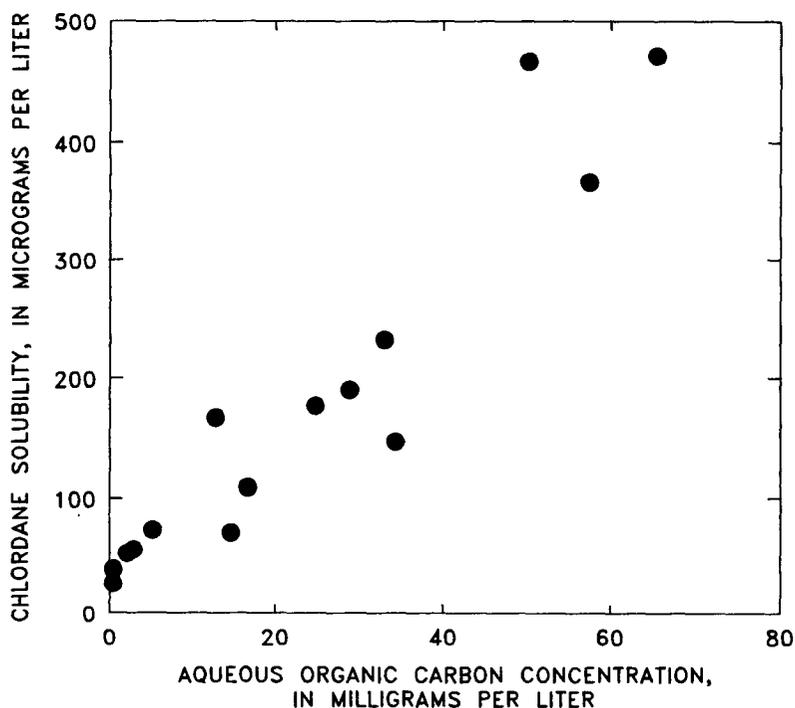
Figure 14.——Effect of aqueous organic carbon on the solubility of chlordane in water.

Batch sorption experiments quantified the partitioning behavior of chlordane in a water-sediment system (L.R. Logan-Johnson, Memphis State University, written commun., 1989). Geologic materials from the unconfined aquifer were collected at seven sites near the North Hollywood Dump, and the organic carbon fraction of each sample was determined by oxidation in an induction furnace. Organic carbon fractions ranged from 0.04 to 0.62 percent. Ground water was collected from wells both upgradient and downgradient from the North Hollywood Dump. Ground water, geologic materials, and chlordane labeled with carbon-14 were shaken for 24 hours at 24 °C in teflon centrifuge tubes. The slurry was centrifuged, and the activity of the aqueous phase was determined in a scintillation counter. After drying, solids were combusted at 900 °C and the activity of the evolved carbon dioxide was measured. All experiments were conducted in triplicate. Chlordane was found to sorb strongly to glass, teflon, and aquifer solids. Values of K_d (the ratio of the carbon-14 concentration in the aquifer solids to the carbon-14 concentration in the aqueous phase) ranged from 28 to 310 mL/g. Values of K_d were generally higher in systems containing solid phases with higher fractions of organic carbon (fig. 15). For example, the K_d for sandy materials collected near the Wolf River was approximately 30 mL/g; this material had an organic carbon fraction of 0.04 percent. K_d values ranged from 112 to 310 mL/g for more fine-grained material with organic carbon fractions of 0.25 to 0.62 percent. These values are consistent with the common observation that sorption of hydrophobic organic compounds is correlated with the organic fraction of the sorbent (Choi and Chen, 1976; Rao and Davidson, 1980; Sharom and others, 1980; Chiou and others, 1983).

Degradation studies were conducted for 6 months in microcosms maintained under both aerobic and anaerobic conditions (L.M. Mallory, Memphis State University, written commun., 1988). Chlordane labeled with carbon-14 was added to saturated aquifer materials at an initial
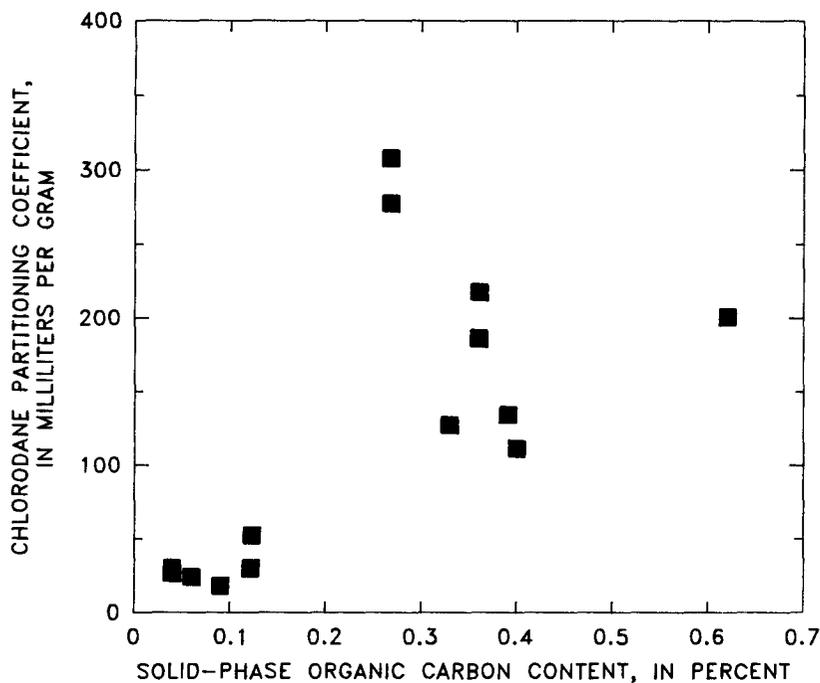
Figure 15.——Effect of solid—phase organic carbon content on the partitioning of chlordane between solid and liquid phases.

concentration of 2.4 μg/kg of wet weight. Replicate microcosms were sampled at log-linear time intervals and were analyzed for C-14 derivatives in solid, liquid-extractable, and gas phases. No appearance of C-14 derivatives in the gas phase or water-extractable phase was observed in the experiments. No solid-phase daughter products were identified, and at the conclusion of the experiments, chlordane concentrations were essentially unchanged from initial values.

Estimated Chlordane Loading to the Wolf River

Pesticide loading to the Wolf River due to the discharge of ground water contaminated by leachate from the North Hollywood Dump may be estimated by a consideration of source strength and the potential for transport and persistence in the unconfined aquifer. The source strength may be estimated from calculation of the total mass of waste material and chlordane concentration in this mass. Chlordane mobility in the unconfined aquifer may be estimated from knowledge of ground-water velocity and the distribution of chlordane between mobile and immobile phases. Chlordane persistence is poorly defined, and a wide range of degradation rates must be considered.

Source Strength

Thirty-seven samples of waste material were collected in July 1985 at 4 to 7 vertical increments at 7 sites at the North Hollywood Dump (Conestoga-Rovers and Associates, 1989). In 24 of these samples, the chlordane concentration did not exceed the reporting level of 80 μg/kg. In the other 13 samples, the concentration of chlordane ranged from 247 μg/kg to 16,070 μg/kg. By arbitrarily assigning the reporting level to samples in which chlordane concentration was not quantified, a mean concentration of 1,200 μg/kg was estimated for the waste material.

The total mass of chlordane at the North Hollywood Dump is estimated at 6,000 kg. This estimate was calculated by multiplying the total mass of waste material by a mean chlordane concentration of 1,200 μg/kg. The total mass of waste material was estimated by multiplying the total waste volume (70 acres of dump area with a mean thickness of 28.5 feet) by an estimated waste density of 2,000 kg/m^3.

Pesticide loading to the unconfined aquifer near the North Hollywood Dump can occur by infiltration of precipitation into the waste material and the solubilization of pesticides as the water percolates to the water table. Additionally, pesticides may be mobilized if ground water enters the waste material through its southern boundary or during periods of increased water-table elevation associated with high stages of the Wolf River. These latter processes, which are immune to manipulations of the hydraulic properties of the surface of the dump, may produce a substantial volume of water in direct contact with the waste.

Annual variations occur in the fluxes of leachate from the North Hollywood Dump. During years when the water table remains below the base of the waste material, the leachate flux will be limited to recharge from precipitation. Infiltration of 11 in/yr through an area of 70 acres results in a flux of 0.09 ft^3/s. This flux represents a lower bound on the mean loading of contaminated water from the dump. A reasonable upper bound on mean annual leachate flux is more difficult to estimate. A very conservative approach is to assume that all the water in the unconfined aquifer beneath the dump is contaminated. Mean discharge of ground water beneath the North Hollywood Dump is estimated at 0.6 ft^3/s. This estimate is based on a mean ground-water velocity of 3 ft/d passing through a porous cross-sectional area 30 feet deep (the entire saturated thickness) and 3,000 feet wide (the perimeter of the dump along the Wolf River). A porosity of 20 percent is assumed.

Regardless of the source or quantity of water contacting the waste, mobilization of a particular pesticide from the dump is limited by the effective solubility of that compound. The solubility of chlordane is enhanced by higher concentrations of organic matter in the aqueous phase. In a ground-water sample with a concentration of 34 mg/L as total organic carbon, which is representative of ground water downgradient from the North Hollywood Dump (fig. 9), chlordane solubility was observed to be 150 μg/L. If this concentration is applied to the range of leachate-flux values estimated above (0.09 to 0.6 ft^3/s), the mean chlordane flux from the North Hollywood Dump ranges from 12 to 80 kg/yr. If these rates were sustained, and chlordane within the dump did not degrade, the estimated mass of chlordane within the waste material (6,000 kg) would be exhausted in 75 to 500 years.

Mobility and Persistence

Chlordane transport in the unconfined aquifer is determined by the ground-water velocity and sorption onto aquifer solids. In experiments using ground water collected near the dump and a sandy material from the unconfined aquifer, the observed distribution coefficient was 30 mL/g (see section on *Experiments with Chlordane*). Because ground-water flow in the unconfined aquifer occurs primarily along sandy pathways, this value is appropriate for estimating chlordane mobility. Using equation 3, and assuming a bulk density of 2.0 g/cm^3 (a typical value for a sandy aquifer) and a porosity of 0.2, the mobility of chlordane would be retarded by a factor of approximately 300 relative to the mean ground-water velocity. Thus, if ground water has a mean velocity of 3 ft/d, chlordane would migrate at a mean rate of 0.01 ft/d. Because travel distances from the base of the dump to the discharge area range from 200 to 2,000 feet, potential mean travel times for chlordane leached from the waste material and migrating to the Wolf River are estimated to be of the order of 50 to 500 years. Due to the uncertainty in the estimate of mean ground-water velocity, mean travel-time estimates for ground

water may be in error by a factor of four. In addition, because of the inherent hydrologic heterogeneity of the system, ground water in some areas of the dump will flow at velocities faster or slower than the mean rate. Because of geo-chemical heterogeneity, including spatial differences in the distribution coefficient, estimates of mean travel times for chlordane are uncertain by at least an order of magnitude.

Chlordane and other organochlorine pesticides are expected to degrade very slowly in the aquifer. Even very slow degradation rates, however, can reduce concentrations significantly during a period of decades or centuries. For the purpose of simulating chlordane transport in the unconfined aquifer, half-lifes ranging from 10 to 100 years were considered. The possibility of no degradation was also addressed.

Contaminant Transport Simulations

Chlordane behavior in the unconfined aquifer was simulated using a three-dimensional contaminant transport model developed by Galya (1987). This model depicts the behavior of a reactive, non-conservative solute as the solute originates from a horizontal plane source and moves through a unidirectional flow field. Changes in contaminant concentration due to transport and degradation are described by the equation:

$$\frac{dC}{dt} = \frac{D_x}{R}\frac{d^2C}{dx^2} + \frac{D_y}{R}\frac{d^2C}{dy^2} + \frac{D_z}{R}\frac{d^2C}{dz^2} - \frac{v}{R}\frac{dC}{dx} - \lambda C + \frac{M_s}{\Theta} \quad (4)$$

where
C = concentration;
t = time;
D_x, D_y, D_z = dispersion coefficients in the x, y, and z directions;
R = retardation factor;
v = ground-water velocity (advection in x-direction only);
λ = first-order degradation coefficient = ln 2 / half-life;
M_s = mass flux of contaminant source; and
Θ = porosity

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Any consistent units are acceptable. Equation 4 can be solved analytically through the application of Green's functions (Carslaw and Jaeger, 1959). The initial condition is:

$$C = 0 \quad \text{at} \quad t = 0$$

and boundary conditions include:

$$C = 0 \quad \text{at} \quad y = \pm \infty$$

$$D_z \frac{dC}{dz} = 0 \quad \text{at} \quad z = 0 \text{ and } z = H$$

where
H = thickness of aquifer

The North Hollywood Dump was depicted as the composite of three rectangular, horizontal plane sources (fig. 16). The flux of chlordane from the source area to the flow field was simulated at a constant rate of 26 kg/yr. The simulated flux represents 140 acre-feet of water saturated with chlordane at a concentration of 150 μg/L. This volume of water is more than twice the estimate of mean annual recharge from precipitation and accounts for additional contact between waste material and ground water during periods of elevation in the water table. A flux of 26 kg/yr would exhaust a chlordane mass of 6,000 kg in 230 years. In reality, the flux could not be sustained at this level until the source was exhausted but would be diminished in the waning years. Simulation of a constant source strength for 230 years will result in an overestimation of chlordane loading to the aquifer and thus an overestimation of chlordane discharge to the Wolf River.

Values for transport variables for the simulations were estimated from the available data or from relations documented in the literature (table 6). Estimates of ground-water velocity, chlordane retardation, and aquifer porosity have been described in previous sections of this report. Dispersion coefficients depend on velocity, scale of measurement, and direction (Sauty, 1980;

Waldrop and others, 1985). The velocity dependency of the longitudinal (x-direction) dispersion coefficient was assumed to be linear; thus, the dispersion coefficient was assigned a value that was a multiple of the ground-water velocity. This velocity multiplier, commonly called the dispersivity, has units of length and is of the same spatial scale as the scale of heterogeneity in aquifer hydraulic properties, here considered to be 10 feet. Directional dependency was addressed by discounting transverse (y- and z-direction) dispersion coefficients by an order of magnitude relative to the longitudinal dispersion coefficient (table 6).

Concentrations of chlordane were simulated at 69 uniformly distributed points along the zone of ground-water discharge to the Wolf River (fig. 16). These 69 points were located at 23 sites in plan view, with 3 points in the vertical at each site. The mean concentration for all points was calculated at 5-year intervals for a period as long as 800 years (fig. 17). Because no reliable estimate for the persistence of chlordane could be

Table 6.--*Input values for simulation of chlordane transport and fate in the unconfined aquifer near the North Hollywood Dump*

Variable	Value
Ground-water velocity	3 feet per day
Retardation factor	300
Porosity	0.2
Dispersion coefficients	
x-direction (toward river).	30 feet squared per day.
y-direction	3 feet squared per day
z-direction	3 feet squared per day
Dispersivity	
x-direction	10 feet
y-direction	1 foot
z-direction	1 foot
First-order degradation coefficient.	0 to 0.0693 years^{-1}
Half-life	10 years to infinity
Mass flux at source	26 kilograms per year

I-240

35°11' 30"

Wolf

River

Oxbow
Lake

Active dredge
pond

Abandoned
dredge
pond

BELMONT CIRCLE

HOLLYWOOD STREET

HEARD AVENUE

PEREZ AVENUE

35°10' 30"

0   1,000   2,000 FEET
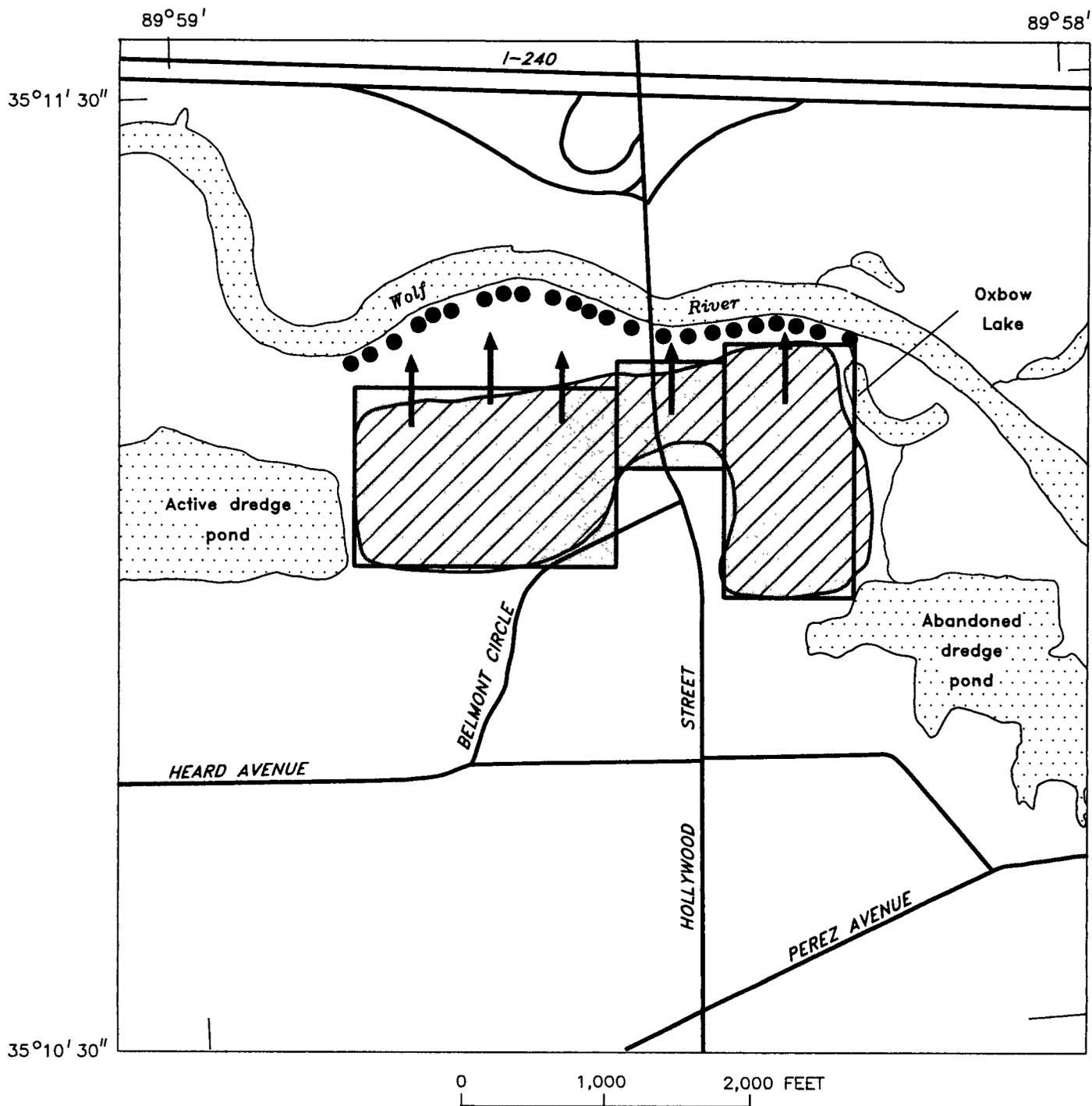
EXPLANATION

AREAL EXTENT OF NORTH HOLLYWOOD DUMP

AREA OF SIMULATED CHLORDANE LOADING

DIRECTION OF SIMULATED GROUND-WATER FLOW

SITE OF SIMULATED VERTICALLY AVERAGED CHLORDANE
CONCENTRATION IN GROUND WATER DISCHARGING TO
THE WOLF RIVER--Chlordane concentration simulated
at three equally spaced vertical increments at
each site

Figure 16.--Conceptual representation of chlordane loading from the North
Hollywood Dump and transport within the adjacent unconfined aquifer. 33
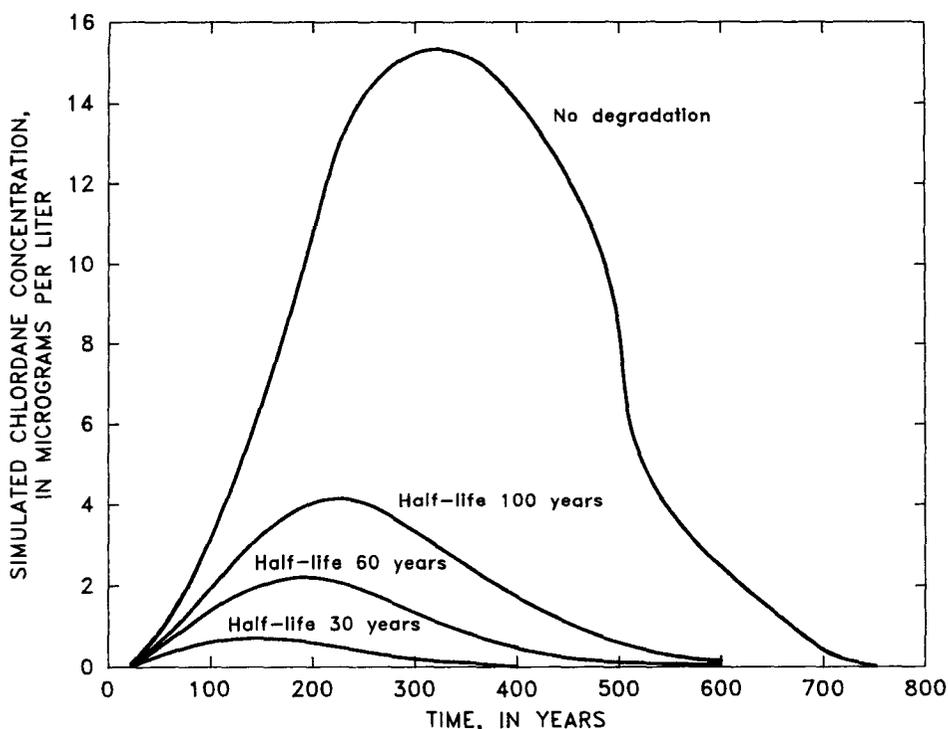
Figure 17.--Effect of half-life on simulations of the mean concentration of chlordane along the discharge zone between the unconfined aquifer and the Wolf River near the North Hollywood Dump.

determined, calculations were made for several potential values of the half-life of the compound. In simulations depicting no degradation of chlordane, the mean concentration in ground water discharging to the river reached a maximum of about 15 μg/L after 320 years. For smaller values of the half-life, simulated peak concentrations were reduced, and the peak occurred earlier in time (table 7). For example, the peak concentration was 0.75 μg/L at 135 years for the simulation depicting a chlordane half-life of 30 years.

Chlordane in ground water that discharges to the Wolf River is subject to dilution. The mean flow of the Wolf River at Walnut Road at Memphis, Tennessee, for 18 years of record is 990 ft³/s (Lowery and others, 1988). This gaging station is approximately 10 river miles upstream from the North Hollywood Dump, but flow does not change markedly through intervening reaches. The mean flow of the river is 1,650 times greater

than the mean estimated discharge of the segment of the unconfined aquifer underlying the dump. If there are no upstream sources of the pesticide, the concentration of chlordane following complete mixing in the Wolf River at average flow conditions would be 1,650 times less than in ground water affected by the dump. Based on this dilution factor, the concentration of chlordane in locally discharging ground water could reach 0.79 μg/L without exceeding the most stringent criterion for water quality in the river (0.00048 μg/L, table 4).

Simulated concentrations of chlordane in discharging ground water did not exceed 0.79 μg/L when the half-life of chlordane was assumed to be 30 years or less (table 7). When chlordane was assumed to undergo no degradation at all, the simulated concentration in the discharging ground water exceeded 0.79 μg/L for 600 years.

Table 7.--*Effect of half-life on simulated mean concentration of chlordane along the discharge zone between the unconfined aquifer and the Wolf River near the North Hollywood Dump*

[µg/L, micrograms per liter]

Half-life (years)	Peak concentration (µg/L)	Year of occurrence of peak	Years concentration exceeded 0.79 µg/L
Infinite	15.41	320	55 - 655
100	4.18	225	60 - 470
80	3.25	210	65 - 420
60	2.24	190	70 - 350
50	1.72	175	75 - 305
40	1.21	160	85 - 250
30	.75	135	none
20	.34	110	none
10	.07	75	none

Actual concentrations of chlordane in ground water discharging to the Wolf River near the North Hollywood Dump may differ considerably from the simulated concentrations. Simulations of contaminant transport in ground water always require simplifying assumptions regarding hydrologic and geochemical processes. The simulations described above are based on best estimates of the mass of chlordane in the dump, the amount of water contacting the waste material, ground-water velocity, chlordane sorption, and possible ranges of chlordane persistence. Actual concentrations could be somewhat higher if conditions more favorable to chlordane mobilization occurred. Such conditions include greater volumes of water moving through the waste material and less sorption of chlordane in the aquifer.

Actual concentrations of chlordane in the discharging ground water will likely be lower than the simulated concentrations. In the simulations, the leachate flux is assumed to be saturated with chlordane. Because every parcel of water moving through the waste material is not likely to become saturated, this assumption overestimates the source strength and thus overestimates chlordane loading

to the Wolf River. The simulations depict chlordane moving through the entire thickness of the aquifer at a retarded velocity based on chlordane sorption to sandy material. While most of the ground-water flux will occur through sandy zones, to the extent that chlordane encounters silty and more organically rich aquifer solids, retardation may be increased. Increased retardation would provide more time for degradation, thus reducing simulated concentrations in transport scenarios where degradation occurs.

SUMMARY AND CONCLUSIONS

The potential for pesticide transport in ground water near the North Hollywood Dump is very limited. Discharge of ground water degraded by leachate from the landfill occurs to the Wolf River. Organochlorine pesticides sorb strongly to aquifer solids, greatly restricting their mobility in the ground-water flow system.

The North Hollywood Dump is underlain by approximately 30 feet of saturated unconsolidated sands, silts, and clays. Beneath this unconfined aquifer is a relatively continuous confining unit. Under average hydrologic conditions, the water table slopes with a mean gradient of 0.007 from south to north where it intersects the Wolf River. Flow in the unconfined aquifer is primarily in the coarser grained material, where longitudinal hydraulic conductivity is approximately 80 ft/d. The porosity of the aquifer was assumed to be 0.2. Using these hydraulic properties, mean ground-water velocity is estimated to be 3 ft/d. Flow is very nearly horizontal beneath the dump. Evidence for horizontal flow patterns includes stratigraphy, absence of vertical hydraulic gradients, and the shallow distribution of water-quality degradation in the system.

The effects of the dump on ground-water quality are evident in physical and chemical analyses and the results of a bioassay. The median value of specific conductance in samples from 14 wells screened upgradient or lateral to the

dump was 488 μS/cm; in samples from 19 down-gradient wells, the median value of specific conductance was 1,270 μS/cm. Organochlorine pesticides were detected in samples from 17 of 35 wells. The frequency of detection was higher in samples from downgradient wells (12 of 20) than in upgradient or laterally located wells (5 of 13). Samples from wells located north of the Wolf River did not contain detectable concentrations of these pesticides. A bioassay featuring negative chemotaxis by fluorescent bacteria demonstrated that bacteria tended to avoid ground water collected downgradient from the dump.

Potential pesticide loading to the unconfined aquifer is limited by solubility. Pesticide solubility is enhanced by dissolved organic matter. For chlordane, observed solubility was 32 μg/L in de-ionized water and 150 μg/L in water with a total organic carbon concentration of 34 mg/L. Because of pesticide sorption onto the organic fraction of aquifer solids, pesticide mobility is

retarded with respect to ground-water velocity. Chlordane may degrade while moving through the aquifer, but potential rates of degradation cannot be quantified with the available data. Upon discharge to the Wolf River, chlordane concentration is diluted at average flow conditions by a factor of about 1,650.

The simulated concentration of chlordane in the Wolf River resulting from the discharge of ground water affected by the North Hollywood Dump is sensitive to assumptions regarding chlordane persistence in the aquifer. If the chlordane half-life in the aquifer is 30 years or less, simulated concentrations in the river at average flow conditions do not exceed the most stringent water-quality criterion. Because the source strength of chlordane likely is over-estimated in the simulations, actual concentrations of chlordane in ground water are expected to be less than the simulated values.

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