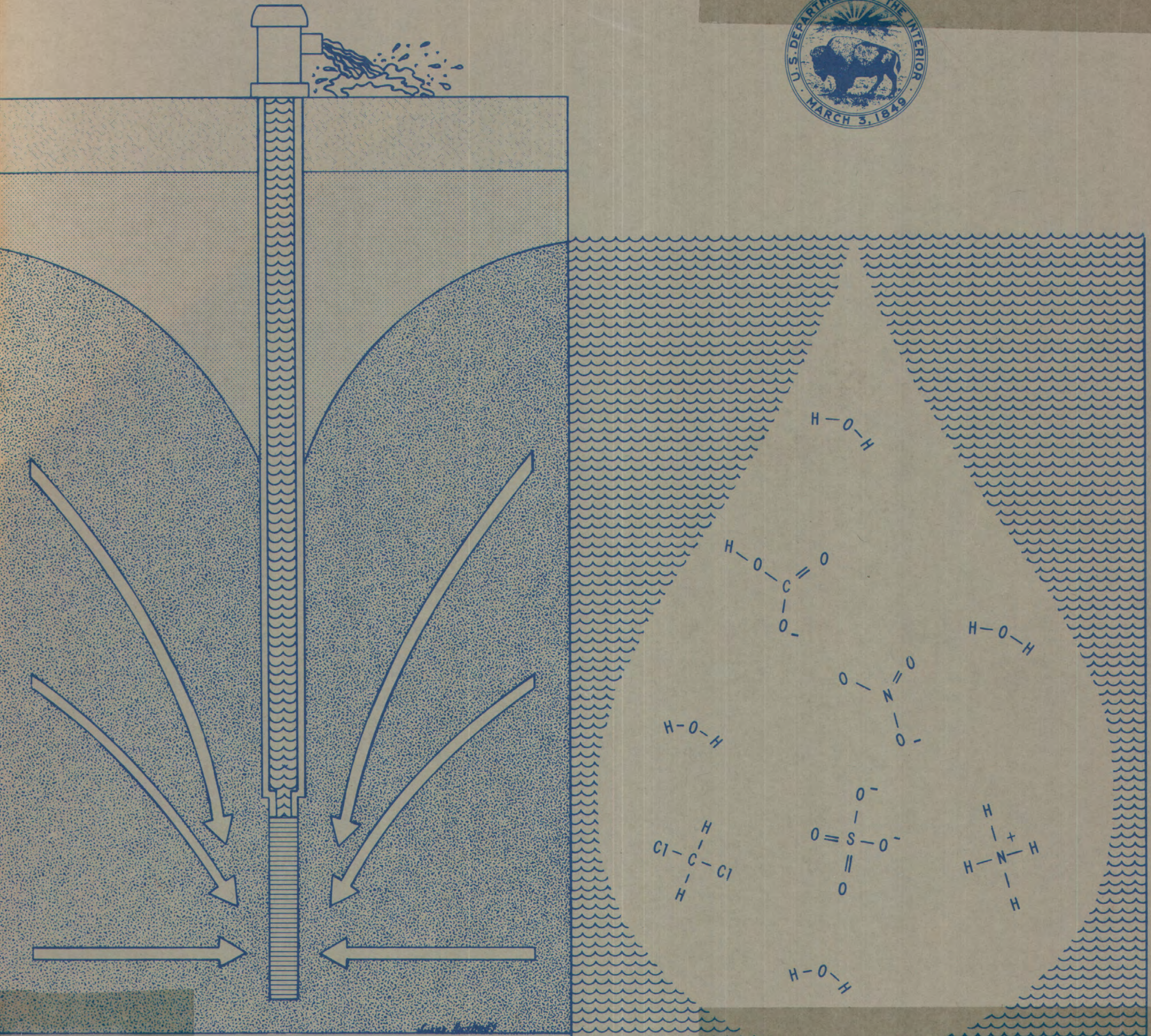
Hydrologic and Chemical Evaluation of the Ground-Water Resources of Northwest Elkhart County, Indiana

GEOLOGICAL SURVEY WATER-RESOURCES INVESTIGATIONS 81-53

U.S. DEPARTMENT OF THE INTERIOR
MARCH 3, 1849



epared in cooperation with the
Department of Natural Resources
and the
Elkhart Water Works

50272-101

REPORT DOCUMENTATION PAGE	**1. REPORT NO.**	**2.**	**3. Recipient's Accession No.**

4. Title and Subtitle

Hydrologic and Chemical Evaluation of the Ground-Water Resources of Northwest Elkhart County, Indiana

5. Report Date

October 1981

6.

7. Author(s)

Thomas E. Imbrigiotta and Angel Martin, Jr.

8. Performing Organization Rept. No.

USGS/WRI/NTIS 81-53

9. Performing Organization Name and Address

U.S. Geological Survey, Water Resources Division
6023 Guion Road, Suite 201
Indianapolis, Indiana 46254

10. Project/Task/Work Unit No.

11. Contract(C) or Grant(G) No.

(C)

(G)

12. Sponsoring Organization Name and Address

U.S. Geological Survey, Water Resources Division
6023 Guion Road, Suite 201
Indianapolis, Indiana 46254

13. Type of Report & Period Covered

Final

14.

15. Supplementary Notes

Prepared in cooperation with the Indiana Department of Natural Resources and Elkhart Water Works

16. Abstract (Limit: 200 words)

A study of the aquifer system underlying northwest Elkhart County, Indiana, has defined the general flow and quality of the area's ground water. The regional ground-water flow is toward the St. Joseph River. A ground-water-flow model was used to simulate pumpings of 7.5-, 10-, and 20-million gallons per day at the airport. The model-generated water-level maps indicated that only for the 20-million-gallons-per-day simulated pumping could a small part of the total flow into the pumping center come from the Himco landfill area.

The general ground-water quality was characterized by slightly basic pH (7-8), average hardness greater than 200 milligrams per liter as calcium carbonate, average alkalinity greater than 150 milligrams per liter as calcium carbonate, and a calcium bicarbonate water type. Eight volatile organic compounds were detected in samples from an industrial park area in east Elkhart. Concentrations of dissolved solids, bicarbonate, bromide, chloride, sulfate, ammonia, calcium, magnesium, manganese, potassium, sodium, and dissolved organic carbon in the leachate were at least five times their background concentrations. The leachate plume was primarily in the shallow aquifer and extended between 3,100 and 5,600 feet downgradient from the Himco landfill.

17. Document Analysis a. Descriptors

Aquifer characteristics, Aquifer testing, Computer models, Drawdown, Flow system, Glacial drift, Groundwater availability, Landfills, Leachate, Observation wells, Organic compounds, Plumes, Solid wastes, Statistical methods, Surface-groundwater relations, Water chemistry

b. Identifiers/Open-Ended Terms

Attenuation, Christiana Creek, Elkhart, Elkhart County, Elkhart River, Historical pumpage, Indiana, Leachate indicators, Potentiometric surface, St. Joseph River, Saturated thickness, Streamflow reduction, Volatile organic coumpounds

c. COSATI Field/Group

18. Availability Statement No restriction on distribution	**19. Security Class (This Report)** UNCLASSIFIED	**21. No. of Pages** 149
	20. Security Class (This Page) UNCLASSIFIED	**22. Price**

(See ANSI-Z39.18) *See Instructions on Reverse* **OPTIONAL FORM 272 (4-77)**
(Formerly NTIS-35)
Department of Commerce

HYDROLOGIC AND CHEMICAL EVALUATION OF THE GROUND-WATER RESOURCES OF NORTHWEST ELKHART COUNTY, INDIANA

By Thomas E. Imbrigiotta and Angel Martin, Jr.

U.S. GEOLOGICAL SURVEY

Water-Resources Investigation 81-53

Prepared in cooperation with the
Indiana Department of Natural Resources and the
Elkhart Water Works

October 1981

UNITED STATES DEPARTMENT OF THE INTERIOR

JAMES G. WATT, Secretary

GEOLOGICAL SURVEY

Dallas L. Peck, Director

For additional information write to:

U.S. Geological Survey
6023 Guion Road, Suite 201
Indianapolis, Indiana 46254

CONTENTS

Page

CONTENTS--Continued

ILLUSTRATIONS

ILLUSTRATIONS--Continued

ILLUSTRATIONS--Continued

TABLES

FACTORS FOR CONVERTING INCH-POUND UNITS TO INTERNATIONAL SYSTEM (SI) OF METRIC UNITS

Multiply inch-pound unit	By	To obtain SI unit
	Length	
inch (in.)	25.40	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow	
cubic foot per second (ft^3/s)	28.32	liter per second (L/s)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
million gallons per day (Mgal/d)	43.81	liter per second (L/s)
	Hydraulic Units	
transmissivity, square foot per day (ft^2/d)	0.0929	square meter per day (m^2/d)
hydraulic conductivity, foot per day (ft/d)	0.3048	meter per day (m/d)
foot per mile (ft/mi)	0.1894	meter per kilometer (m/km)
inch per year (in./yr)	25.40	millimeter per year (mm/yr)

NATIONAL GEODETIC VERTICAL DATUM OF 1929 (NGVD)

Formerly called SEA LEVEL DATUM 1929. A geodetic datum derived from a general adjustment of the first order level nets of both the United States and Canada. In the adjustment, sea levels from selected TIDE stations in both countries were held as fixed. The year indicates the time of the last general adjustment. This datum should not be confused with MEAN SEA LEVEL.

BRAND NAMES

The use of brand names in this report is for identificaion purposes only and does not imply endorsement by the U.S. Geological Survey.

ABBREVIATIONS

Abbreviation	Description
ANOVA	Analysis of variance
x	by
ft^3/s	cubic foot per second
°C	degree Celsius
diss.	dissolved
DOC	dissolved organic carbon
ft	foot
ft/d	foot per day
>	greater than
≥	greater than or equal to
in./yr	inch per year
ISBH	Indiana State Board of Health
<	less than
≤	less than or equal to
µg/L	microgram per liter
µmho/cm	micromho per centimeter
µ	micron
mi	mile
mg/L	milligram per liter
Mg/L as $CaCO_3$	milligram per liter as calcium carbonate
Mgal	million gallons
Mgal/d	million gallons per day
mv	millivolt
min	minute
NGVD of 1929	National Geodetic Vertical Datum of 1929
NIPDWR	National Interim Primary Drinking Water Regulations
NSDWR	National Secondary Drinking Water Regulations
ln	natural logarithm
pH	negative logarithm of the hydrogen ion concentration
±	plus or minus
PVC	polyvinyl chloride
ft^2/d	square foot per day
mi^2	square mile
St.	street
tot.	total
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
yr	year

HYDROLOGIC AND CHEMICAL EVALUATION OF THE GROUND-WATER RESOURCES OF NORTHWEST ELKHART COUNTY, INDIANA

By Thomas E. Imbrigiotta and Angel Martin, Jr.

ABSTRACT

A 3-year study in northwest Elkhart County, Indiana, was done to (1) define the general flow and quality of water in the outwash aquifer system, (2) determine if a well field proposed for a site at the Elkhart Municipal Airport would draw leachate from the Himco landfill, and (3) define the areal extent of the ground water affected by the landfill and an east-side industrial-park area.

The outwash aquifers are mainly composed of sand and gravel and are separated by a silt and clay bed. The saturated thickness of these deposits averages 175 feet and ranges from 85 to 500 feet. Where present, the silt and clay bed confines the underlying aquifer. The confining bed is absent in the area underlying the landfill and part of the airport. Average hydraulic conductivities of the sand, and the sand and gravel, are 80 and 400 feet per day, respectively. Regional ground-water flow is toward the St. Joseph River.

A quasi-three-dimensional ground-water flow model, calibrated to match measured water levels and river seepages, was used to simulate proposed pumpings in the airport area. Drawdowns for simulated pumpings of 7.5-, 10-, and 20-million gallons per day were small percentages of the saturated thicknesses of the unconfined aquifer underlying the airport area. Model-simulated streamflow reductions in Christiana Creek for these pumpings were 9, 12, and 23 percent of base flow, respectively. The model-simulated, steady-state water-level maps for the 7.5- and 10-million-gallons-per-day pumping experiments indicated that these pumpings would not draw water from the landfill area to the pumping center at the airport. The simulated map for the 20-million-gallons-per-day pumping indicated that only a small part of the flow into the pumping center could come from the landfill area.

The general quality of ground water in the study area, defined by analyses of 68 water samples from 35 observation wells, included slightly basic pH (7-8), average hardness greater than 200 milligrams per liter as calcium carbonate, average alkalinity greater than 150 milligrams per liter as calcium carbonate, and, in more than 90 percent of the analyses, a calcium bicarbonate water type.

Eight volatile organic compounds were detected in water from 8 of 19 wells sampled in the industrial-park area. The health-effects advisory limit

for chronic exposure to trichloroethylene was exceeded in samples from six of these wells. Areally, the results did not define a plume or a source of the volatile organic compounds.

Concentrations of dissolved solids, bicarbonate, bromide, chloride, sulfate, ammonia, calcium, magnesium, manganese, potassium, sodium, and dissolved organic carbon in the leachate were at least five times their background concentrations. Concentrations of bromide and dissolved solids were the best indicators of leachate. Bromide concentrations plotted areally indicated that the landfill was the source of the leachate, the leachate plume extended between 3,100 and 5,600 feet downgradient from the landfill, and the plume had not spread laterally much farther than the original width of the landfill. Bromide and dissolved-solids concentrations plotted in cross section indicated that the leachate plume was sinking as it spread downgradient in the shallow unconfined aquifer and the leachate was present in deep wells under and immediately downgradient from the landfill. All chemical constituents were attenuated within approximately 1 mile downgradient from the landfill.

INTRODUCTION

Problem Definition

Elkhart, Ind., obtains its water supply from three well fields screened in a thick outwash aquifer. In 1980, daily pumpage from the three fields averaged 7 Mgal. Increases in population and industrial activity projected for the next 20 years will increase the demand for water in the Elkhart area. A new well field to be located at the Elkhart Municipal Airport has been proposed to meet the demand.

Ground-water quality has been degraded in some areas near Elkhart by liquid and solid-waste-disposal practices. Volatile organic compounds have been detected in residential wells near an industrial area in east Elkhart. A landfill on the northwest side of the city has been closed because its leachate has penetrated the shallow unconfined aquifer. Because the landfill is approximately 1 mi from the site proposed for the new well field, the quality of ground water at the site could be affected.

To plan the expansion of the water supply, the Elkhart Water Works needed to determine (1) the hydrologic suitability of the proposed well-field site, (2) the areal extent of the effects of the landfill and the industrial area on the local ground-water systems, (3) the effect of the landfill on ground water at the proposed well-field site, and (4) the locations of other sites near Elkhart that could provide an adequate ground-water supply.

Purpose and Scope

To assess the problems in Elkhart, the U.S. Geological Survey, in cooperation with the Indiana Department of Natural Resources and the Elkhart Water Works, began a study whose objectives were to:

(1) Define the general ground-water hydrology of northwest Elkhart County.

(2) Define the general ground-water quality of the Elkhart area.

(3) Determine the areal extent of the effects of the Himco landfill and the east-side industrial park on the local ground-water system.

(4) Determine the hydrologic effects (extent of cone of depression, streamflow reduction) of the proposed ground-water pumping at the Elkhart Municipal Airport on the local ground-water system.

(5) Determine whether leachate from the Himco landfill would affect the quality of water at the proposed new well-field site.

Objectives 1 and 4 are discussed in Part 1 of this report, "Ground-Water Investigation," and objectives 2, 3, and 5 are discussed in Part 2, "Water-Quality Investigation."

Acknowledgments

The authors acknowledge the assistance and the cooperation of the many organizations and people who made this report possible: The 14 industries and 2 municipalities who helped reconstruct historical pumpage data; Indiana and Michigan Electric Co., VanPLY, Inc., Morse Industries, Inc., Wayne R. Howard, and George M. Cooper, for permission to install observation wells on their properties; Ortman Drilling Co., Kokomo, Ind., for their excellent workmanship in installing the deep observation wells; and Sam H. Lyle of the Elkhart Water Works, for his invaluable help and support in data collection.

BACKGROUND

Location and History

The study area is in the northwest quadrant of Elkhart County in north-central Indiana (fig. 1). The 120-mi^2 study area is bounded on the west by the Elkhart-St. Joseph County line, on the north by the Michigan State line, on the east by Indiana State Road 15, and on the south by a line 0.2 mi south of the towns of Jamestown and Dunlap. These boundaries, as well as the major rivers, creeks, roads, cities, and towns are shown in figure 1.

The city of Elkhart, the largest urban center in the study area, covers 16.8 mi^2. The population of Elkhart is 46,000, and the population in the fringe area around the city is 23,000 (Elkhart Chamber of Commerce, 1979). Elkhart is a diversified industrial community of more than 700 businesses. Major industrial activities are the manufacture of pharmaceuticals, recreational vehicles, mobile homes, and band instruments.

Agriculture is the major land use elsewhere in Elkhart County. Dairy and poultry farming predominate, but fruit farming (apples) also ranks high (Elkhart Chamber of Commerce, 1979).

Himco Landfill

The Himco landfill, referred to as the "landfill" throughout this report, is in northwest Elkhart at the intersection of County Road 10 and Nappanee St. Extension. (See figs. 1 and 41.) The Elkhart Municipal Airport is approximately 1 mi northeast of this intersection.

The Himco landfill was operated from 1960 to 1977 by Himco Waste Away Service, Inc. This company hauled and disposed of wastes from virtually all of Elkhart's major commercial and industrial businesses during this period. Paper refuse, calcium sulfate, and other pharmaceutical byproducts were the wastes deposited in the largest quantities (William Paynter, Indiana State Health Commission, written commun., 1977[1], and James King, Indiana State Board of Health, written commun., 1978[2]).

---

[1] Memorandum from William Paynter, State Health Commissioner, Indiana State Board of Health, April 5, 1977, to Otis Bowen, Governor, State of Indiana.

[2] Memorandum from James King, Geologist, Indiana State Board of Health, January 3, 1978, to David Lamm, Chief of Solid Waste Management Section, Indiana State Board of Health.

The ISBH (Indiana State Board of Health) identified the Himco site as an open dump in 1971. In early 1974, residents complained to the ISBH that shallow wells immediately adjacent to the site's south border had taste, odor, and color problems. The ISBH, under the authority of the 1974 Indiana State Law SPC-18 governing operation and licensing of landfills, forced the closure of this site by autumn 1977. Thus, from 1974 to 1977 it was operated as a landfill.

The approximately 30-acre landfill site was originally a swampy low-lying area. By the time of closure, the surface elevation of the fill had risen approximately 15 ft above the original ground level at the center and 5 ft above ground level on the perimeter. A 0.5-1 ft thick layer of sand and topsoil was used as a final cover.

Industrial Park

The east-side industrial park, which is referred to as the "industrial park" throughout this report, is bounded on the north by Conrail tracks, on the west by Riverview Avenue and Outer Drive, on the south by U.S. 20 and the city limits of Elkhart, and on the east by Middleton Run Road. (See figs. 1 and 42.) Within the 500-acre park are two industrial developments and some land that is principally industrial or commercial. Some of the more than 60 businesses in the park include plants that produce injection-molded plastic and foam parts, bottled aerosols, and mobile-home parts.

Taste and odor problems were first reported in shallow wells in the spring of 1975 by residents living in the area between the St. Joseph River and the industrial park. In January 1976, a fire and an explosion at a company in the park directly south of these homes caused storage tanks containing pesticides and organic solvents to rupture. At that time, some of the spilled chemicals infiltrated the subsurface, and some flowed overland into a nearby pond. Pesticides were not detected in water samples collected from residential wells shortly after the fire (Ralph C. Pickard, Indiana State Board of Health, written commun., 1977[1]). However, samples collected from these wells in May 1976 and analyzed by the USEPA (U.S. Environmental Protection Agency) were found to contain eight volatile organic compounds, including methylene chloride, 1,1-dichloroethane, and 1,1,1-trichloroethane (Billy Fairless, U.S. Environmental Protection Agency, written commun., 1976[2]). Because the complaints predated the fire and explosion, the source of these chemicals was presumed to be elsewhere in the industrial park. The ISBH investigated all industries in the park adjacent to the affected residential area but was unable to determine the source of these compounds.

---

[1] Letter from Ralph C. Pickard, Assistant Commissioner for Environmental Health, January 5, 1977, to Mrs. Thomas H. Hoetger, well owner, Elkhart, Ind.

[2] Letter from Billy Fairless, Acting Director, Central Regional Laboratory, U.S. Environmental Protection Agency, June 1, 1976, to Neil Ott, Chief, Water-Supply Section, Indiana State Board of Health.

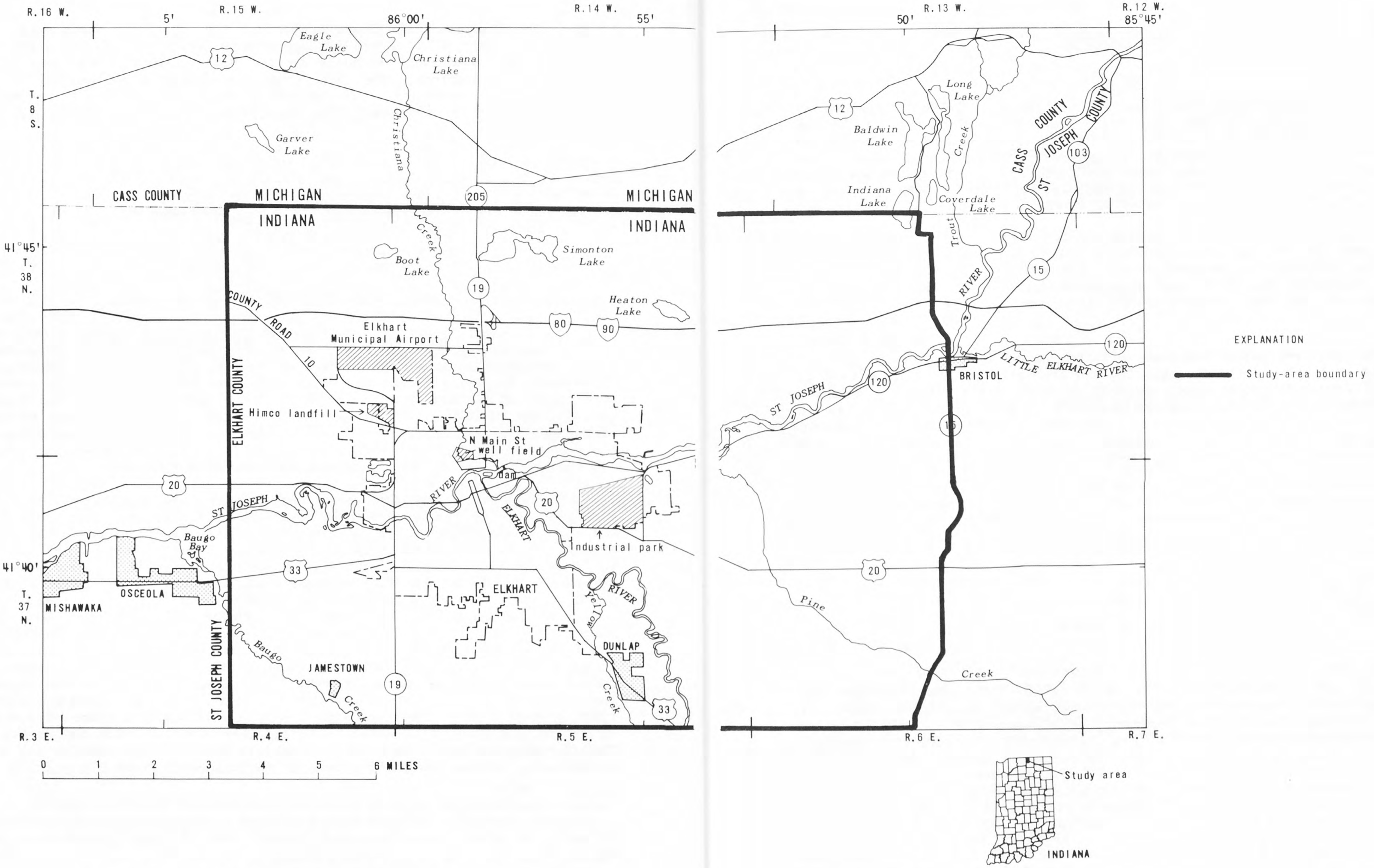


Figure 1.-- Study area in northwest Elkhart County, Ind.

Physiography

The study area is in the St. Joseph River basin. The St. Joseph River flows from east to west through this region, and all surface drainage in the study area flows into either the river or its tributaries. The physiography on both sides of the river consists of generally flat plains that grade to rolling and hilly terrain to the north and south. The land-surface elevation ranges from 950 ft in the hills in the southeast to 740 ft (NGVD of 1929) in the west at the St. Joseph River.

Climate

The Elkhart area has a temperate climate and an average yearly temperature of 10.1° C. Temperature ranges from an average of -4.2° C in January to 23.1° C in July. The mean annual rainfall is 34.5 inches. For the climatic period 1941-70, the average monthly rainfall ranged from 1.69 inches in February to 3.91 inches in July (National Oceanic and Atmospheric Administration, 1978). Snowfall averages 24.5 in./yr (inch per year) and usually occurs between November and March.

Geology

Surficial Geology

Glacial ice extended into the study area four times during the Pleistocene Epoch. Each period was followed by warmer periods of erosion by melt-water streams as the ice receded. As a result, a thick deposit of stratified and unstratified drift covers the bedrock. The surficial geology, shown in figure 2, is characterized by a valley train outwash deposit bounded on the north, south, and southeast by till plains that include ice-contact stratified drift and ground-moraine deposits (Johnson and Keller, 1972; Schneider and Keller, 1970). The deposits of till are mainly composed of silty sand interbedded with clay and silt units.

Bedrock Geology

Bedrock consists of the Coldwater Shale of Mississippian age and the Sunbury and the Ellsworth Shales of Devonian and Mississippian age (Johnson and Keller, 1972; Schneider and Keller, 1970). Drillers' logs, seismic logs, and test-drilling information were used to prepare the bedrock topographic map shown in figure 3. The bedrock dips northeast. In the west-central part of the study area is a preglacial bedrock valley whose axis trends northeast-southwest. Depth to shale ranges from 85 to 500 ft and averages 175 ft.

Drillers' logs and test-drilling information were also used to construct two geologic sections (figs. 4 and 5). Lines A-A' and B-B' for the two sections are shown in figures 2 and 3. These sections show vertical and lateral relationships in the unconsolidated deposits, which are discussed in more detail in the section "Hydraulic Characteristics of the Unconsolidated Deposits."

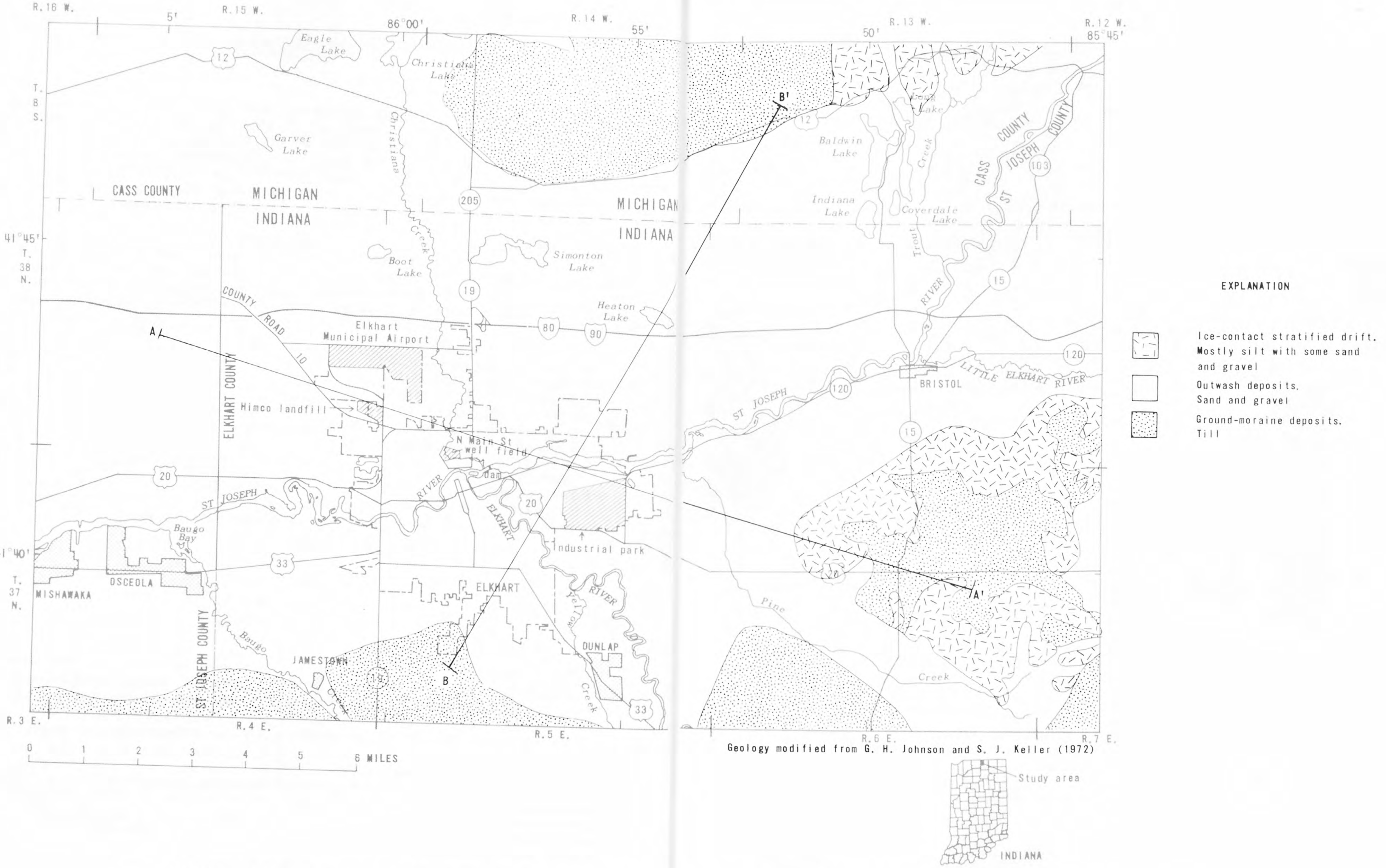
PART 1. GROUND-WATER INVESTIGATION

Part 1 of the report deals with ground-water availability in northwest Elkhart County, Ind. The objectives were to define the general ground-water hydrology and determine the hydrologic effects of the proposed pumping at the Elkhart Municipal Airport on the local ground-water system.

APPROACH

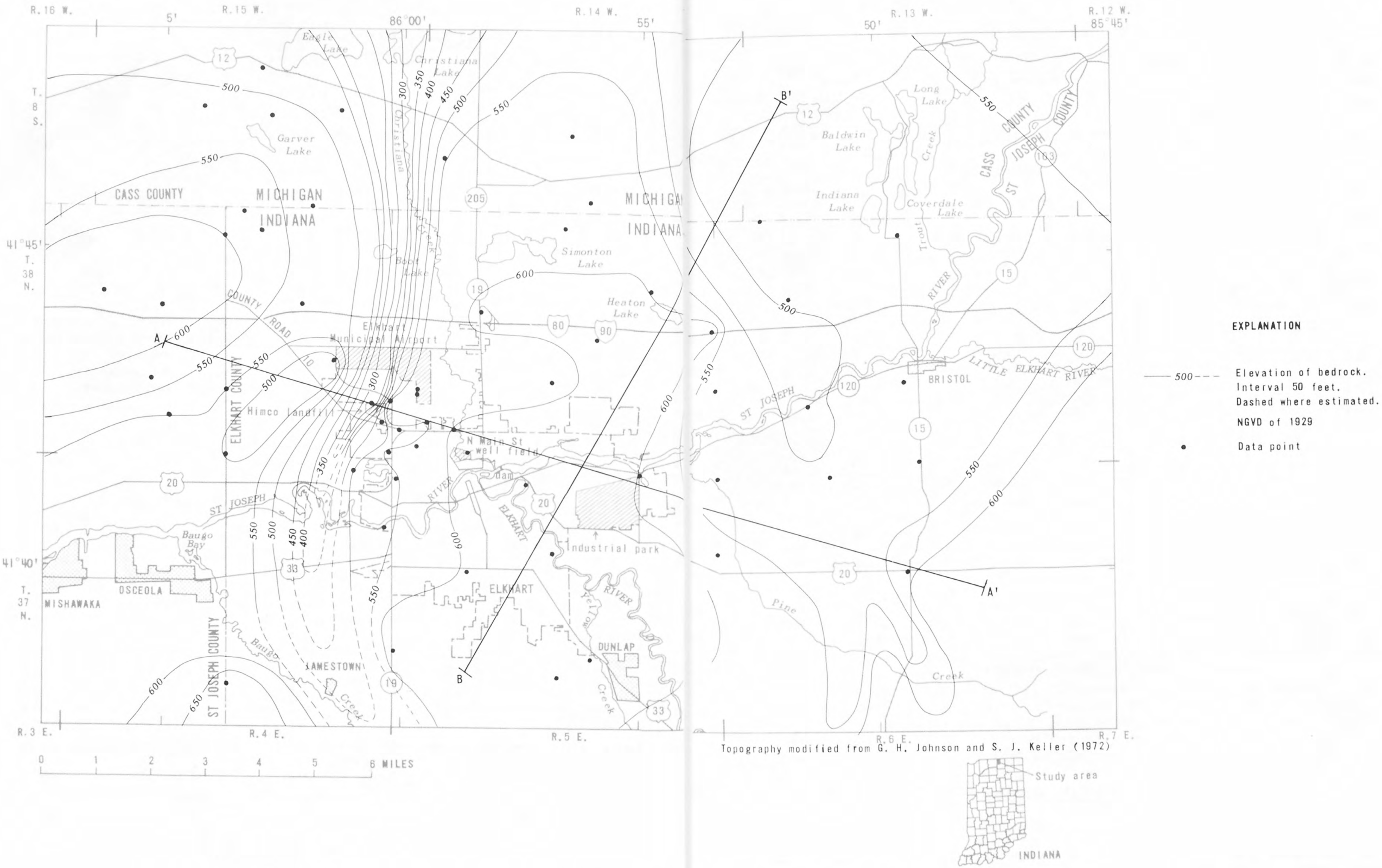
For the first objective, various data were collected on the unconsolidated deposits--thickness, areal extent, hydraulic conductivity, transmissivity, specific yield, and storage coefficient; the general ground-water flow pattern and the stream-aquifer connections were defined; water-level fluctuations were determined; and historical and present-day ground-water pumpages were tabulated.

For the second objective, digital-modeling techniques were applied. A quasi-three-dimensional ground-water flow model, constructed to simulate the ground-water system, was calibrated to steady-state conditions. Steady-state and transient pumping simulations were run to investigate the long- and short-term hydrologic effects of the proposed Elkhart Municipal Airport well field on the ground-water flow system. Results obtained from the simulations were also used to estimate the effect of the pumping on the flow system around the Himco landfill.



Geology modified from G. H. Johnson and S. J. Keller (1972)

Figure 2.-- Surficial geology in parts of Elkhart and St. Joseph Counties, Ind., and Cass and St. Joseph Counties, Mich.



Topography modified from G. H. Johnson and S. J. Keller (1972)

Figure 3.-- Bedrock topography in parts of Elkhart and St. Joseph Counties, Ind., Cass and St. Joseph Counties, Mich.

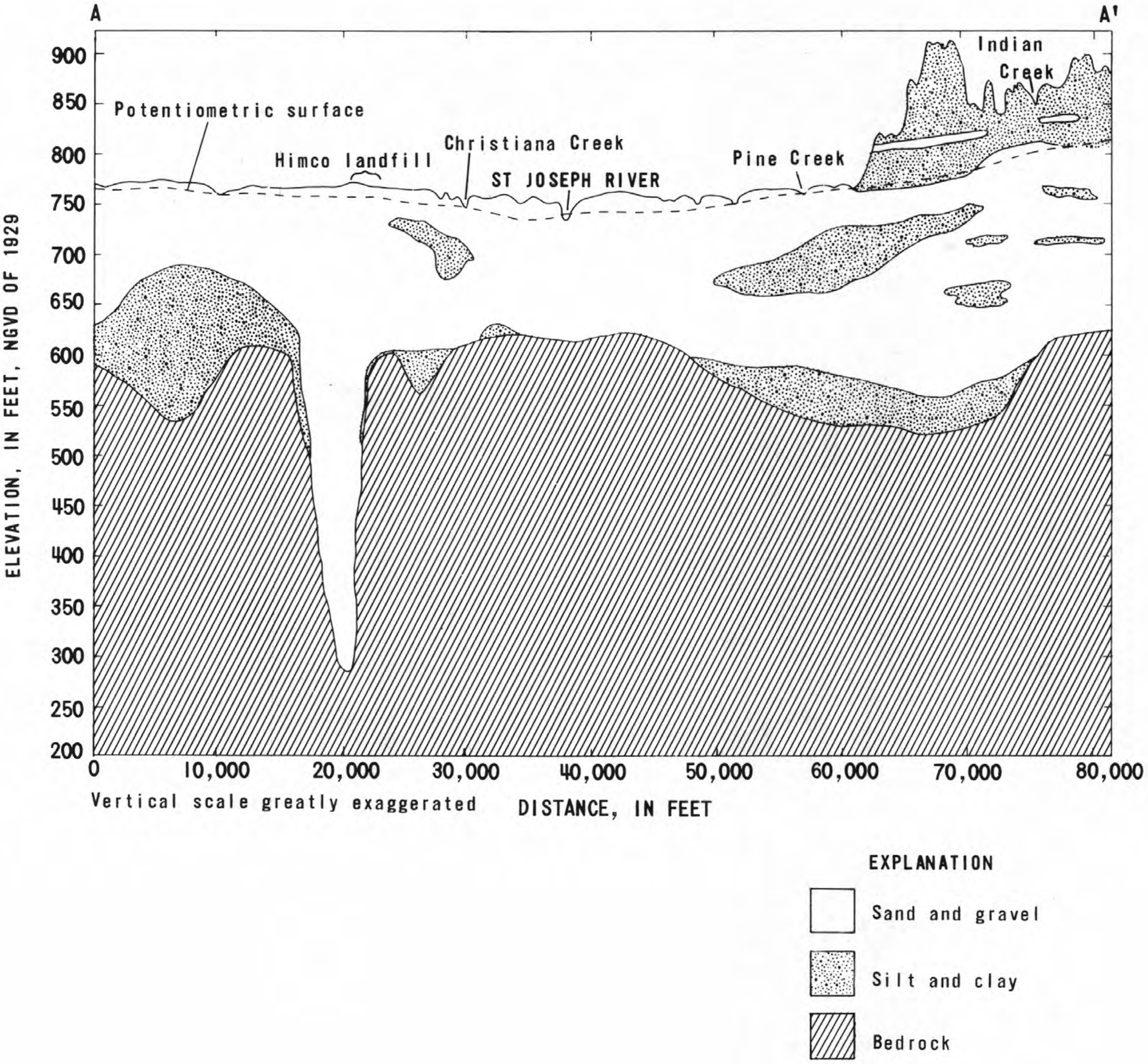


Figure 4.-- Geologic section A-A'.

HYDROLOGY

Hydraulic Characteristics of the Unconsolidated Deposits

The thickness and the areal extent of the layers in the unconsolidated deposits were determined from lithologic logs of previously installed wells, from logs of wells installed during this study (fig. 6), and from natural

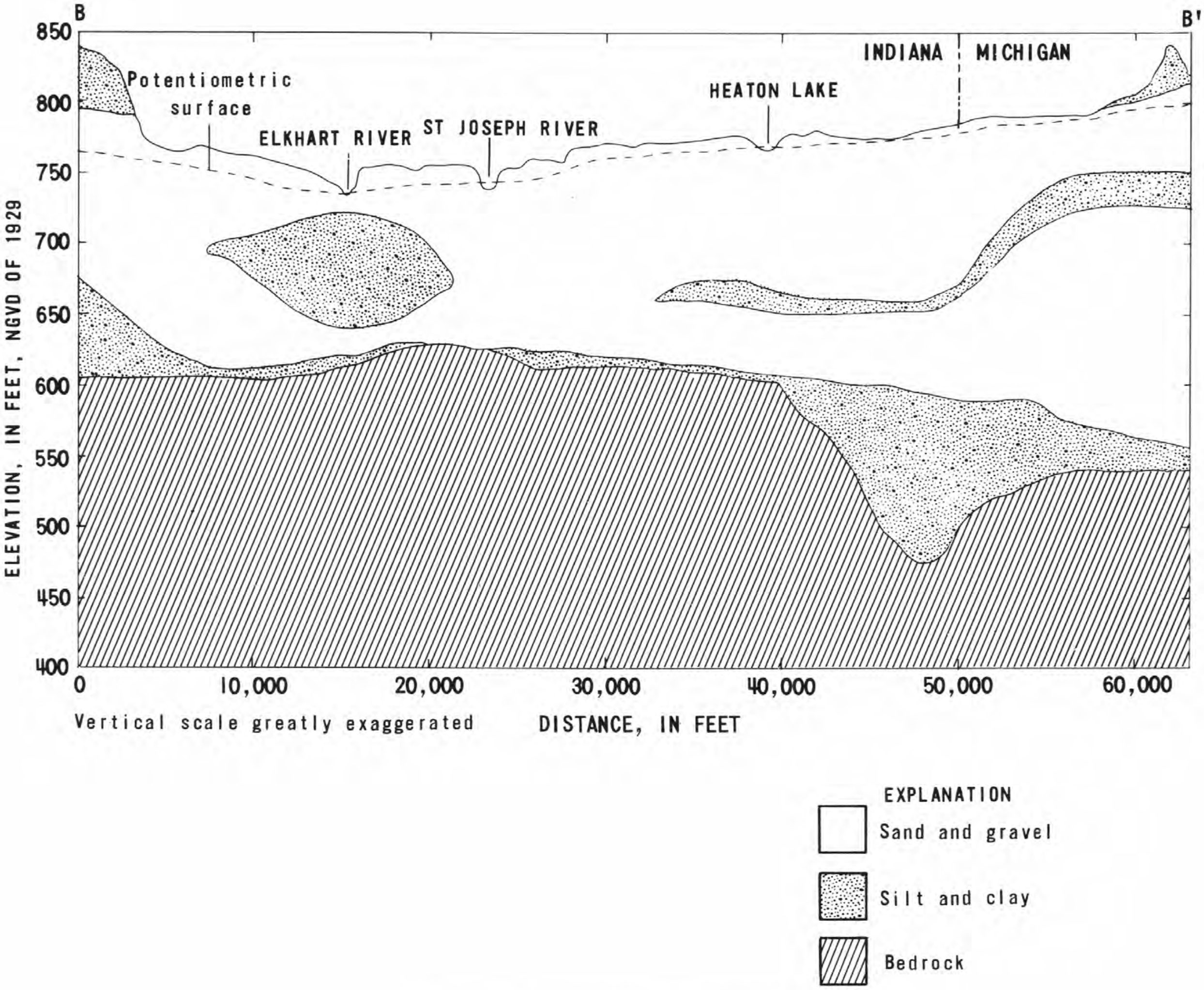


Figure 5.-- Geologic section B-B'.

gamma-radiation logs of 35 deep test holes. These data indicate that the sand and gravel associated with the valley-train deposits along the St. Joseph River valley form an extensive outwash aquifer. The outwash deposits, whose average depth is 175 ft, are overlain by a veneer of topsoil, whose average thickness is 10 inches. Interbedded within these deposits, in much of the study area, is a layer of silt and clay whose maximum thickness is 80 ft and average thickness is 20 ft. Where present, this layer divides the outwash deposits into two aquifers and confines the deeper aquifer. The thickness and the areal extent of the unconfined aquifer, where the confining bed exists, are shown in figure 7, and the thickness and the areal extent of the confined aquifer are shown in figure 8. The thickness and the areal extent of the confining bed are shown in figure 9.

In two areas, the confining silt and clay layer is absent (figs. 7-9). The area to the southeast coincides with the surficial deposits of till, which average 60 ft in thickness. This till unit is underlain by a layer of

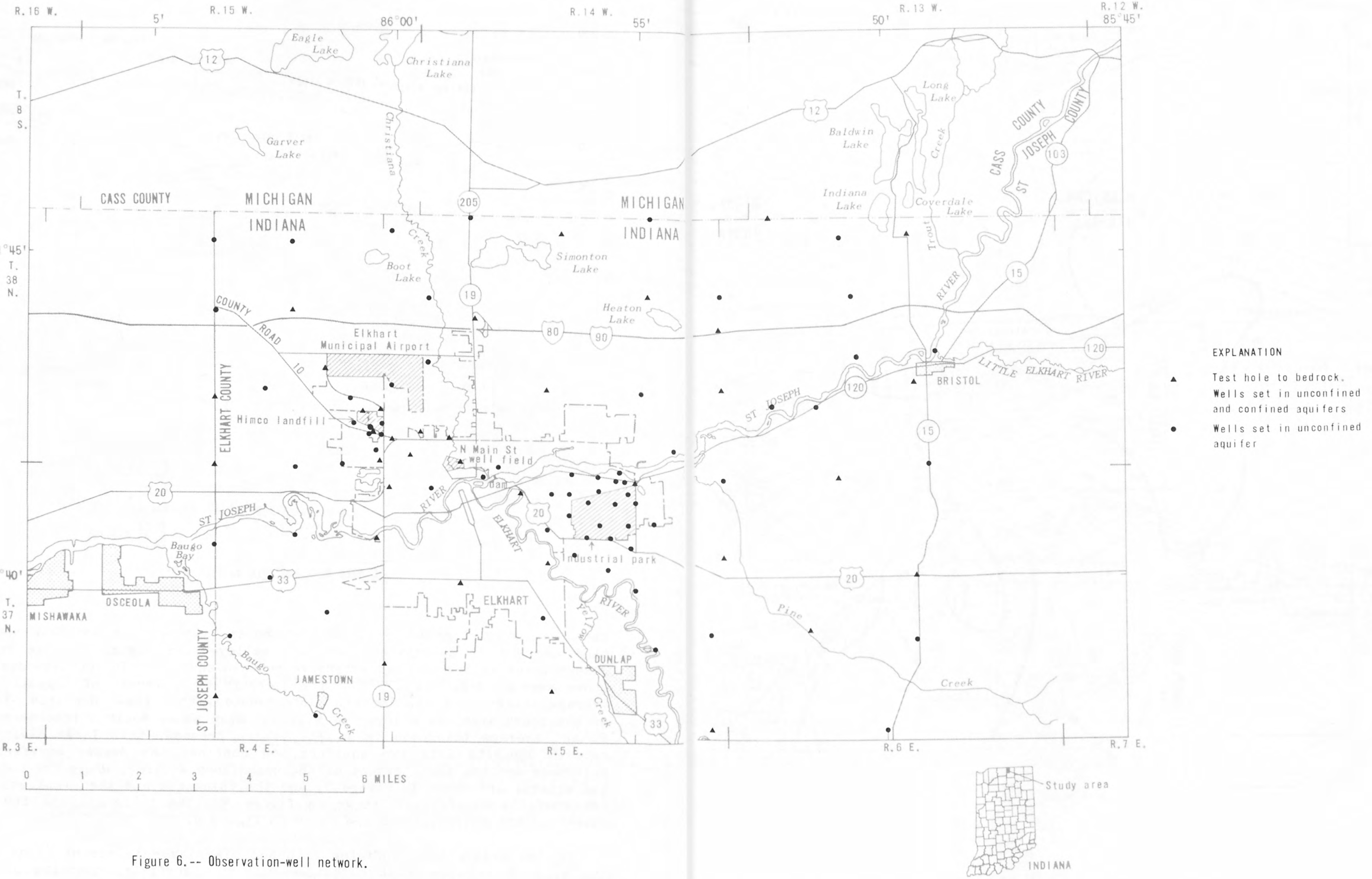


Figure 6.-- Observation-well network.

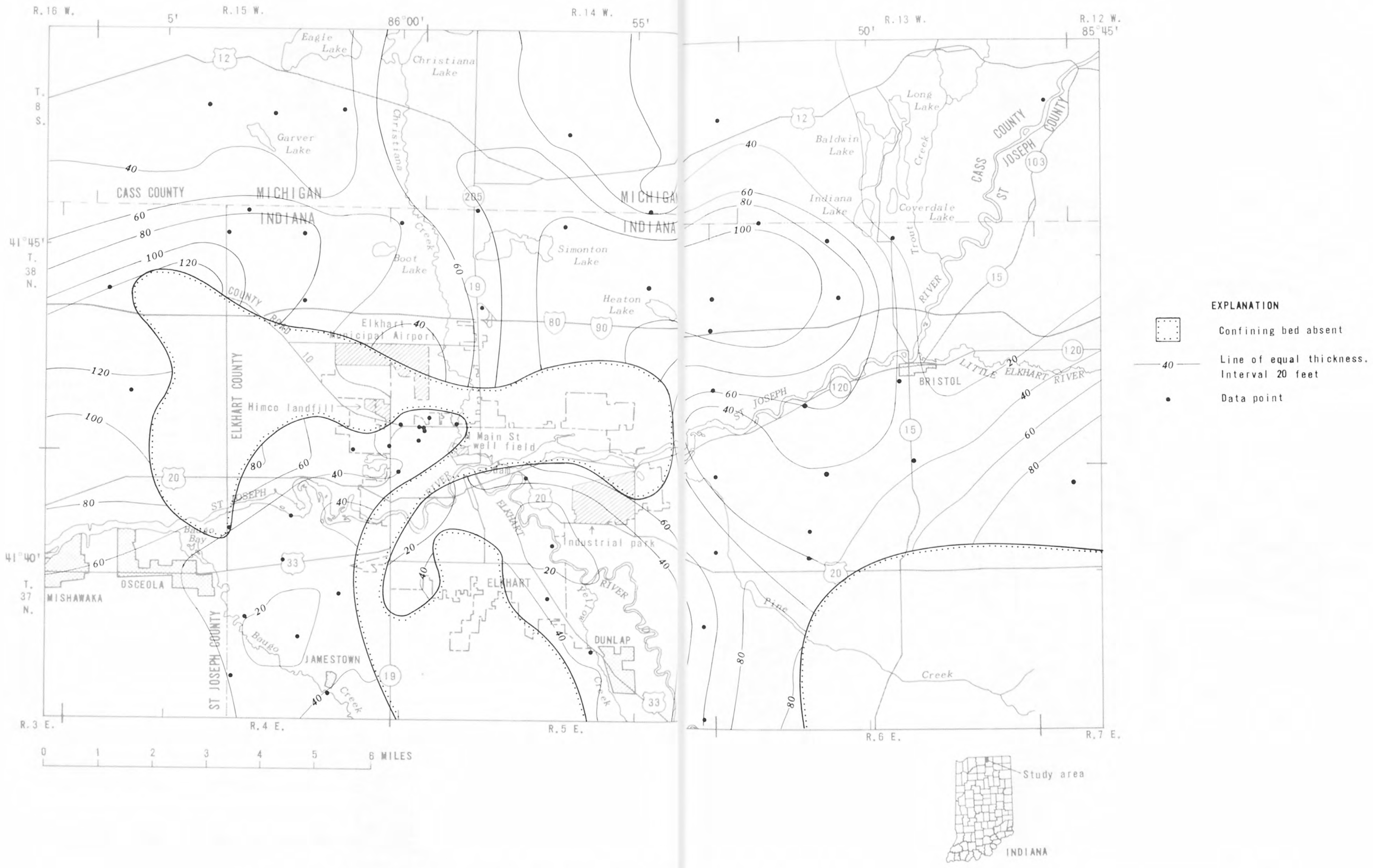


Figure 7.-- Thickness of the unconfined aquifer where the confining bed exists.

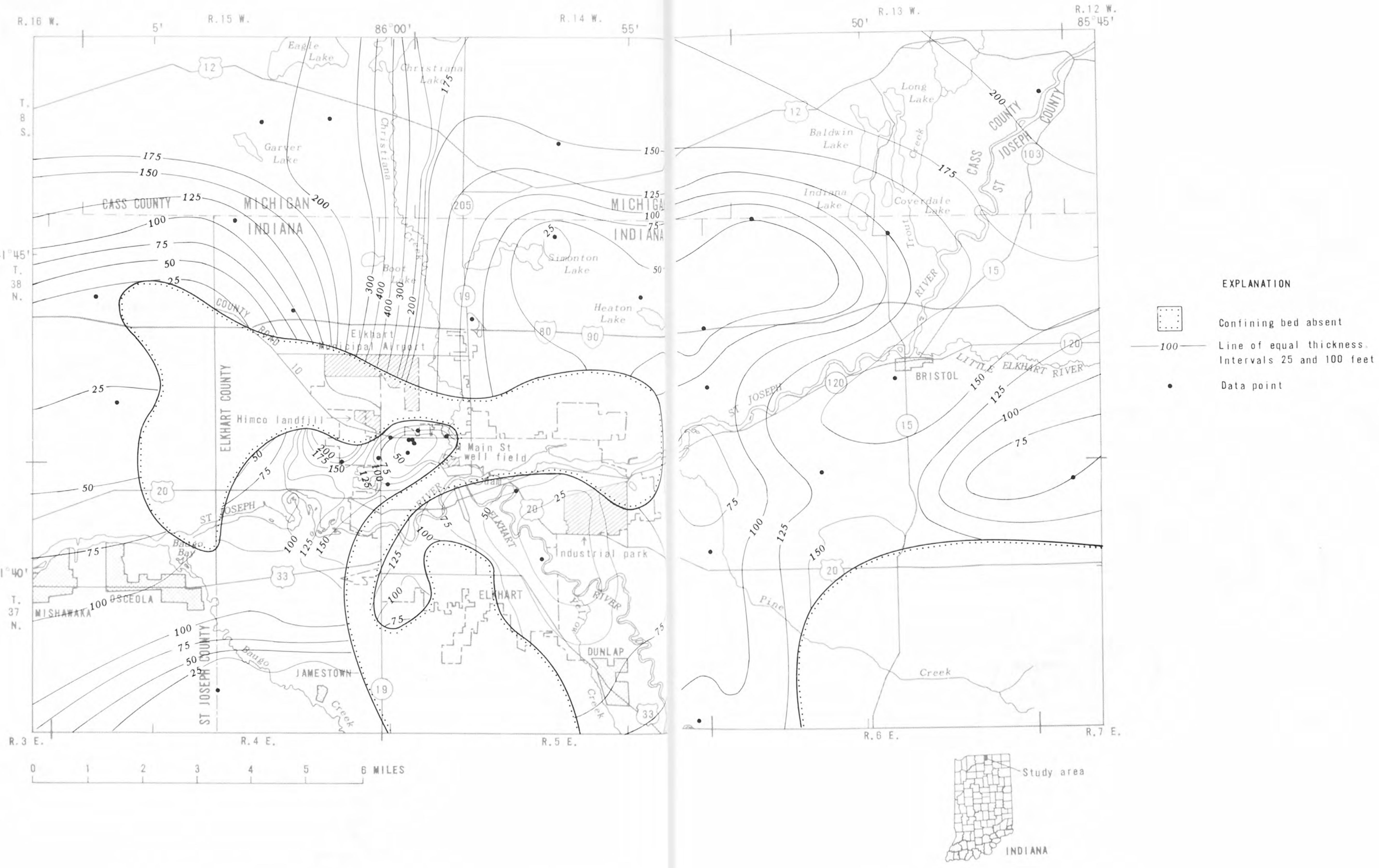


Figure 8.-- Thickness of the confined aquifer.

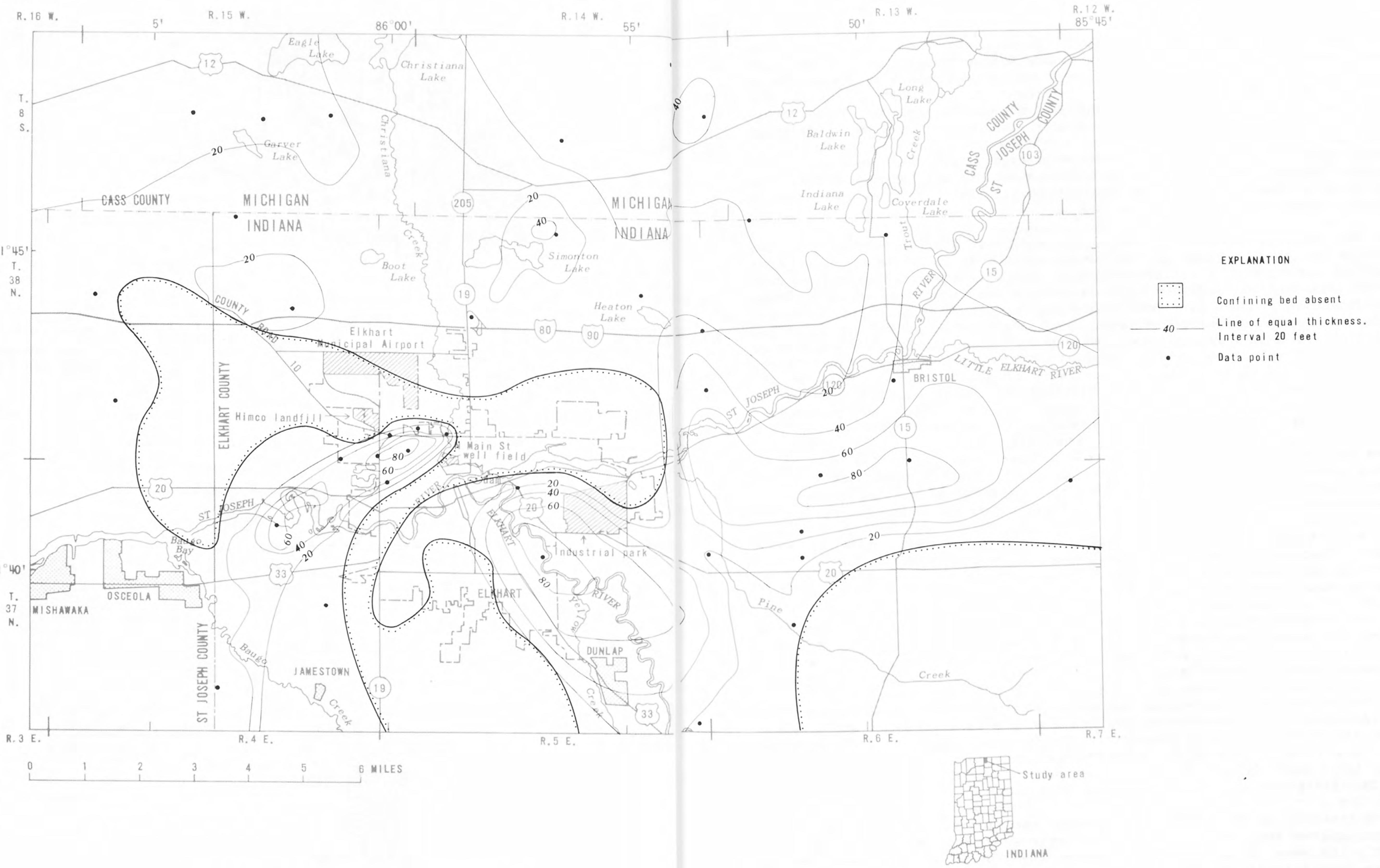


Figure 9.-- Thickness of the confining bed.

sand and gravel interbedded with localized silt and clay lenses. The combined deposit averages 200 ft in thickness and correlates in altitude with the outwash deposits in the valley. Some of the sand and gravel deposits in the southeast area are much siltier than those in the valley. The till and the silt and clay lenses at depth locally confine the sand and gravel aquifer. In the other more centrally located region, where there is no confining bed, the aquifer ranges from 40 ft in thickness in the area of the North Main Street well field, where a thick layer of clay and silt was deposited directly upon the bedrock surface, to more than 450 ft in the bedrock valley. The thickness of the aquifer in the areas of no confining silt and clay bed is presented in figure 10.

The hydraulic conductivities of the different aquifer materials in the outwash system are based on data collected in specific-capacity tests of observation wells in the landfill area. Specific-capacity data can be converted directly to transmissivity values with techniques developed by Theis (1963) and Brown (1963) for nonsteady-state conditions in unconfined and confined aquifers, respectively. Because of low pumping rates (averaging 15 gal/min) and short screens, the resultant transmissivity values were assumed to represent only the screened interval of the aquifer (Meyer and others, 1975, p. 18). The hydraulic conductivity of the aquifer material at the screened interval was calculated by dividing the transmissivity values by the screen length. The results were separated into two categories--one based on sand and the others on sand and gravel in the screened interval of each well. Average values of hydraulic conductivity calculated for sand, and for sand and gravel, were 80 and 400 ft/d, respectively. These values are in close agreement with other hydraulic conductivity values calculated for outwash systems in South Bend, Ind. (Marie, 1975), Marion County, Ind. (Meyer and others, 1975), and Carmel, Ind. (Gillies, 1976).

The lateral hydraulic conductivity of the silt and clay confining bed is small, about 0.1 ft/day (Freeze and Cherry, 1979, table 2.2), relative to the sand and gravel aquifer. Therefore, it was considered insignificant and was omitted in all subsequent calculations.

The transmissivities of the unconfined and the confined aquifers were calculated at points of known thickness by summing the products of the saturated thickness of each lithologic unit (sand or sand and gravel) and the hydraulic conductivity of each unit.

Transmissivities calculated for the unconfined aquifer ranged from 4,000 ft^2/d in several places to 175,000 ft^2/d in the area of the bedrock valley near the Himco landfill. Transmissivities calculated for the confined aquifer ranged from 5,000 ft^2/d, near the town of Jamestown, to 85,000 ft^2/d, near the Elkhart Municipal Airport and north of the Michigan State line west of Christiana Creek.

Collection of field data to determine the specific yield of the unconfined aquifer and the storage coefficient of the confined aquifer was not within the scope of this study. However, these properties were evaluated by Marie (1975) in three aquifer tests at the South Bend, Ind., waste-water-treatment facility. These tests were done approximately 25 mi west of the study area in an outwash deposit similar to that underlying northwest Elkhart County. Results obtained were 0.16 for the specific yield of the unconfined aquifer and 0.00006 for the storage coefficient of the confined aquifer. These values fall within the typical ranges of 0.01 to 0.35 for specific yield and 0.00001 to 0.001 for storage coefficient (Bennett, 1976).

No field data were collected for the direct determination of the vertical hydraulic conductivities of the aquifer, confining bed, or streambed materials. However, an aquifer test in Marion County, Ind. (Meyer and others, 1975), analyzed with type curves presented by Stallman (1965), in an outwash system similar to the one underlying northwest Elkhart County, yielded a ratio of vertical to lateral hydraulic conductivity of 1:10. On the basis of this ratio, the vertical hydraulic conductivities of the sand, and of the sand and gravel, were calculated to be 8 and 40 ft/d, respectively. Previous hydrologic studies in Indiana, in systems with materials similar to that in the study area, have determined that 0.007 ft/d and 0.67 ft/d are average values for the vertical hydraulic conductivity of confining bed and streambed materials, respectively (Marie, 1975; Meyer and others, 1975; Gillies, 1976).

Ground-Water Flow

The direction of water flow in the aquifers underlying northwest Elkhart County was determined by measuring and mapping ground-water levels in observation, domestic, and industrial wells throughout the study area. Stream stage was also measured on the major streams at points near observation wells. The average distance between stage measurements was 1.75 river miles. Water-level maps were drawn from data collected in the unconfined and the confined aquifers in May 1979. (See figs. 11 and 12.) The flow direction, which is perpendicular to the equipotential lines shown, is generally toward the St. Joseph River. This flow pattern is characteristic of a well-connected stream-aquifer system with a gaining stream. Vertical water-level differences between the aquifers are generally small (less than 2 ft in either direction) in areas away from the St. Joseph River and are difficult to perceive from figures 11 and 12, owing to the contour interval used. However, in areas near the river, upward gradients, some greater than 5 ft, indicate that ground water discharges to the river. There is an anomaly at and near the hydroelectric dam where the impounded water level is held unnaturally high. Here the river loses water to the aquifer.

Contours are shown in figure 12 in areas where the confining bed is absent. Data in these areas were collected from wells screened in both the upper and the lower parts of the unconfined aquifer. Upward vertical gradients of as much as 7 ft were observed in these areas near the St. Joseph River.

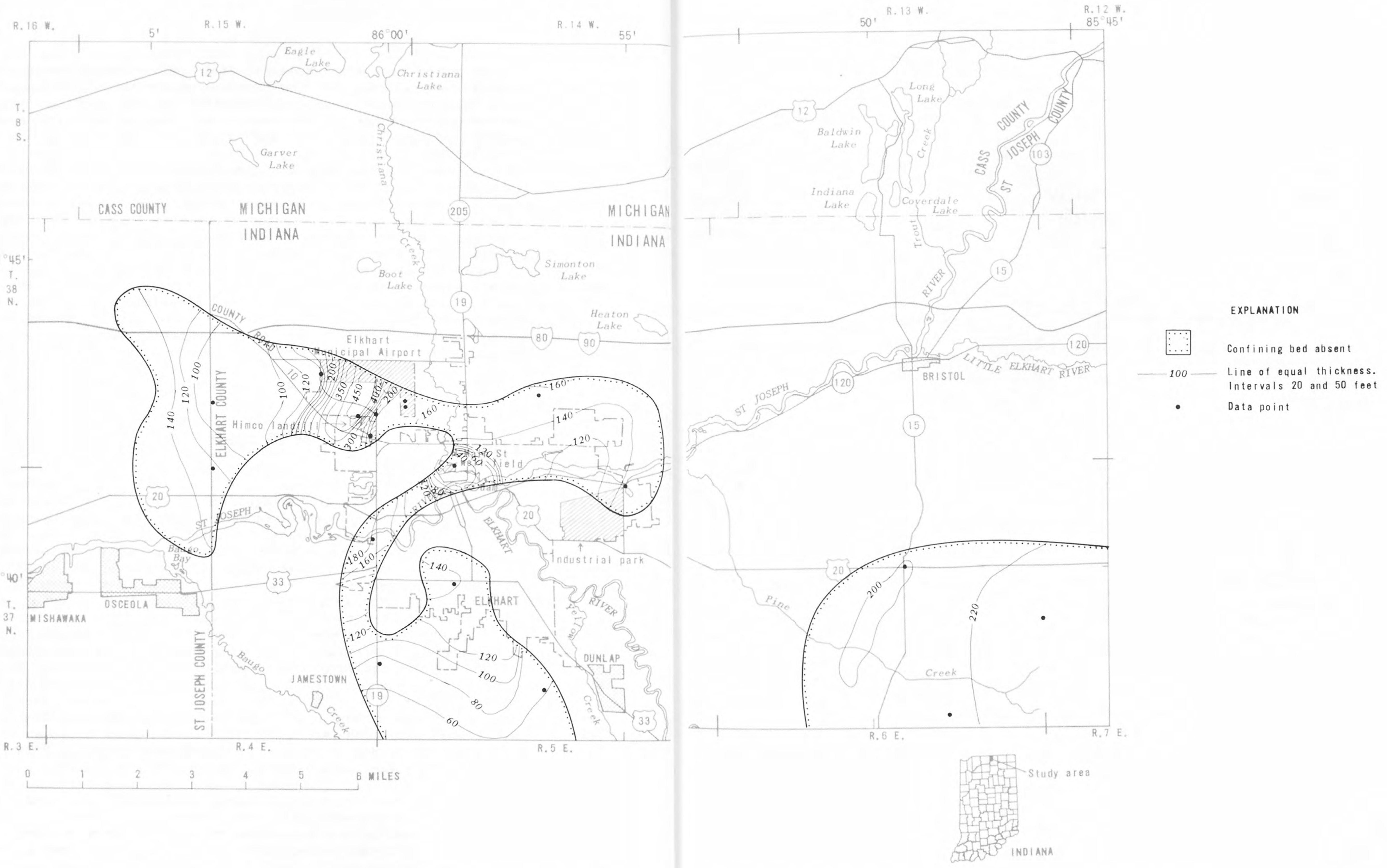


Figure 10.-- Thickness of the unconfined aquifer where the confining bed is absent.

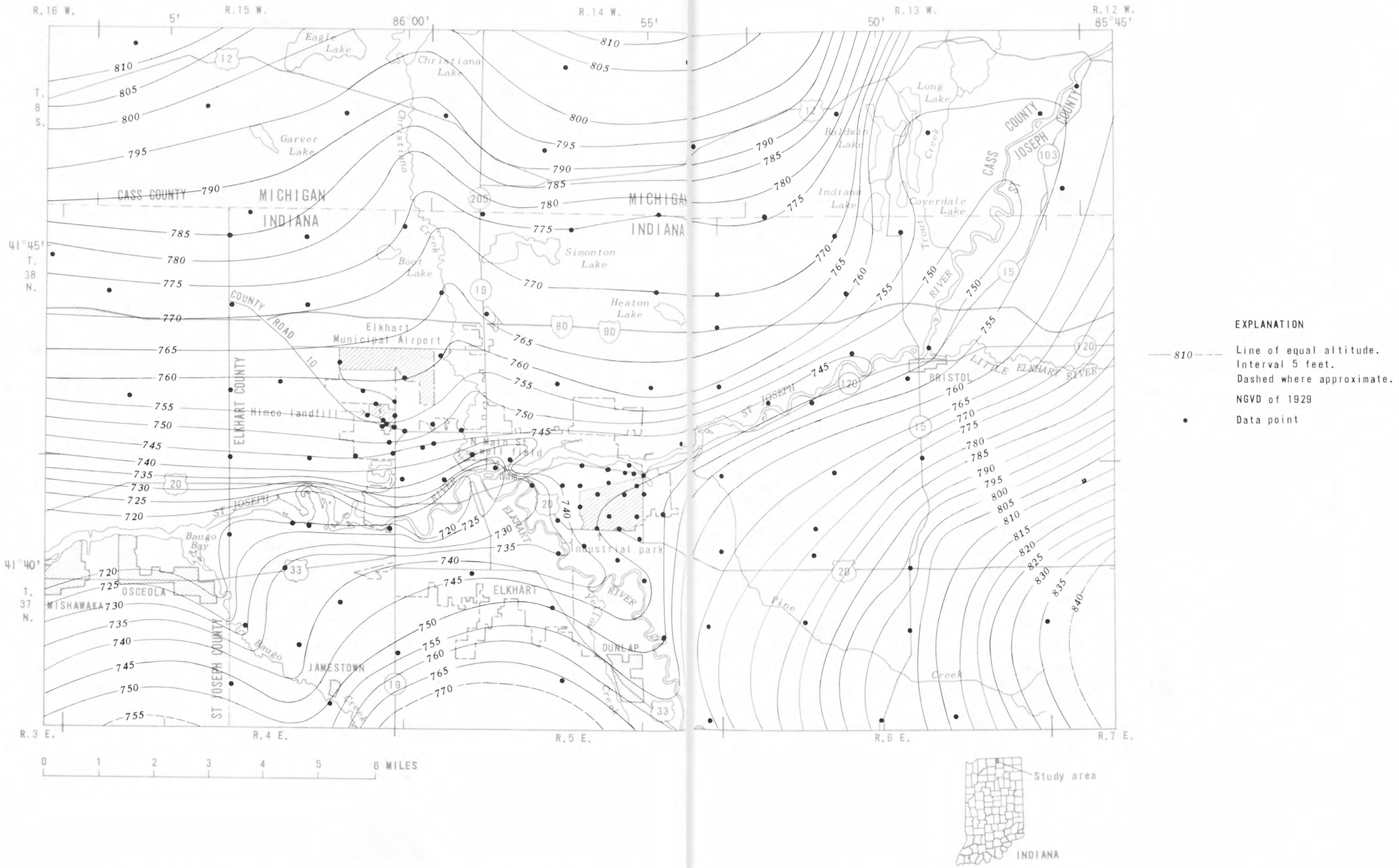


Figure 11.-- Water levels in the unconfined aquifer, May 1979.

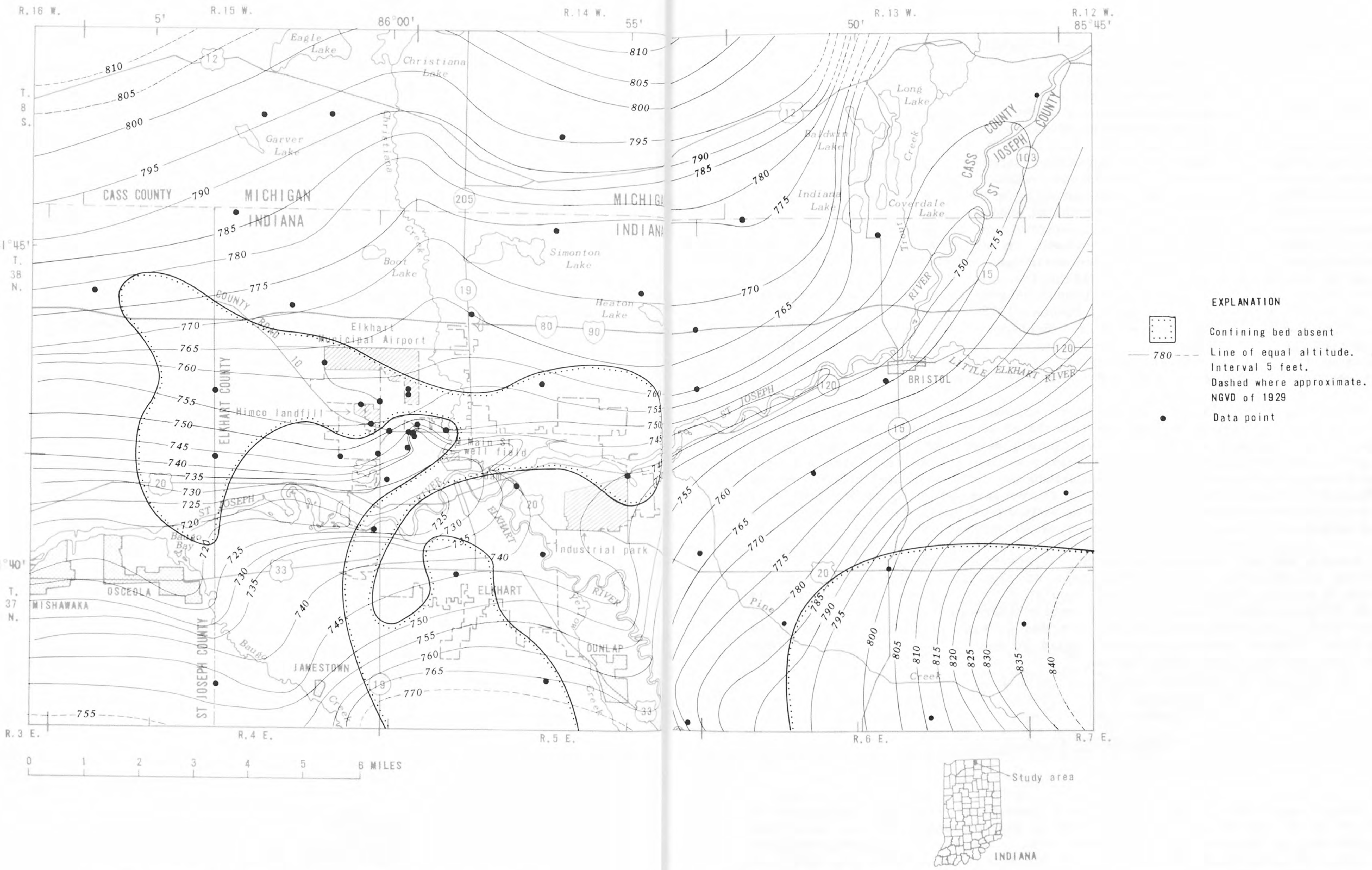


Figure 12.-- Water levels in the confined aquifer, May 1979.

Stream-Aquifer Connection

The hydraulic connection between a stream and an aquifer is a primary concern in determining the potential yield of the aquifer. Pumping large quantities of water from an aquifer can divert ground water from discharging into the rivers and creeks and can even change gaining streams to losing streams. Stream discharge, stream stage, and water levels near the streams in both aquifers were measured simultaneously to help define the stream-aquifer connection. Seepage (gain and loss) measurements were made on the Elkhart River, Christiana Creek, and Baugo Creek in October 1978, May-June 1979, and September 1979, and on Pine Creek in September 1979. (See reaches identified in fig. 13.) Flow-duration estimates for streams were based on 48 years of record at the USGS gage, Elkhart River at Goshen, 4 river miles south of the study area. This gage is the only nearby active one not affected by artificial controls. Flows in October 1978 were measured at a 68-percent flow duration and in May-June and September 1979 were measured at the 84-percent and the 78-percent flow durations, respectively (Horner, 1976). Values of the two sets of seepages measured in 1979 were similar, whereas the ones measured in 1978 were higher than those of 1979.

As a result of the ±5-percent error assumed for all discharge measurements, a range of seepage values was calculated on each reach for each set of measurements (table 1). These calculations showed that reaches 1, 4, 5, 7, 8, and 9 were gaining stream segments, whereas reach 3 was a losing segment. The seepage of reaches 2 and 6 could not be definitely defined as either gaining or losing. Reach 3 on Christiana Creek was a losing reach, primarily because the heavy pumping at the North Main Street well field reverses flow gradients in the area.

Seepage was not measured on the St. Joseph River, the principal discharge for the ground-water system. Discharge at the gaging station on the St. Joseph River at Elkhart (fig. 13), regulated by the reservoir upstream, is 1,600 ft^3/s at the 79-percent flow duration (Horner, 1976). Seepage to the river within the study area is probably small relative to the total flow in the river. Hence, seepage estimates would be masked by the inherent ±5-percent error in discharge measurements.

Water-Level Fluctuations

Annual water-level fluctuations were determined from the hydrograph of Elkhart 5, a continuous-record observation well screened in the unconfined aquifer and in operation since October 1976. (See fig. 13.) The hydrograph in figure 14 shows that water levels are highest in late March and April and lowest in September and October. Water levels fluctuate from 2 to 4 ft/yr. Water levels measured in the spring and the autumn of 1978 and 1979 fluctuated similarly.

Seasonal fluctuations in water levels are caused by differences in the amount of recharge reaching the aquifers during the year. Precipitation is virtually the only source of recharge for the aquifers. However, as noted under the section "Climate," the average monthly rainfall is evenly distributed throughout the year. Fluctuation in recharge to the aquifer, therefore, is not caused by the yearly precipitation distribution.

Evapotranspiration in the unsaturated zone is probably the primary cause of water-level fluctuation. Water levels decline during the warm summer months because shallow-rooted plants intercept water in the unsaturated zone and prevent it from reaching the water table and recharging the aquifers. The decrease in evapotranspiration at the end of the growing season during the cooler months corresponds to a water-level rise. Because depth to the water table averages 10 ft, direct evaporation from the water table is unlikely. The effect of evapotranspiration on the ground-water flow system is basically a reduction in recharge to the aquifers.

Ground-Water Pumpage

The ground-water pumpage in the Elkhart area was determined in a survey of the major ground-water users. Average daily pumpage from the outwash aquifers by 14 industries and 2 municipalities is 13.39 Mgal. Two users, the city of Elkhart and a major industry, pump 85 percent of this total. The locations from which these two major ground-water users pumped are shown in figure 13. The 1974-78 average daily pumpage from and return to the designated aquifers are given in the table that follows.

	Average daily pumpage, 1974-78		Average daily return, 1974-78
Pumping center	From unconfined aquifer (Mgal)	From confined aquifer (Mgal)	To unconfined aquifer (Mgal)
City of Elkhart			
N. Main St. well field	5.75	----	----
Bower St. well field	----	0.69	----
South well field	----	1.15	----
Major industry	.63	3.16	1.64

Elkhart Water Works data were obtained from Sam Lyle, Elkhart Water Works (written commun., 1979).

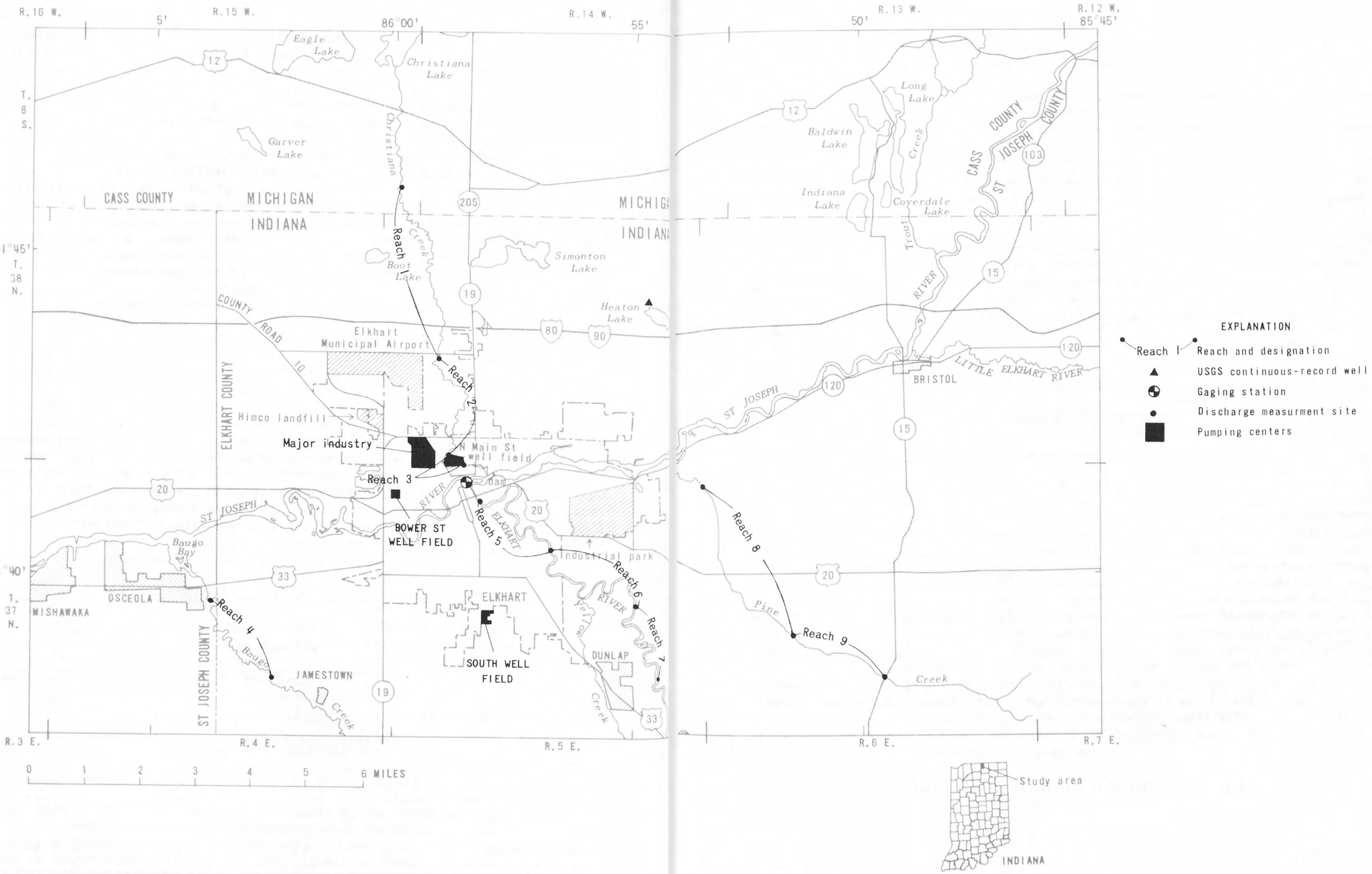


Figure 13.-- Seepage reaches and pumping centers.

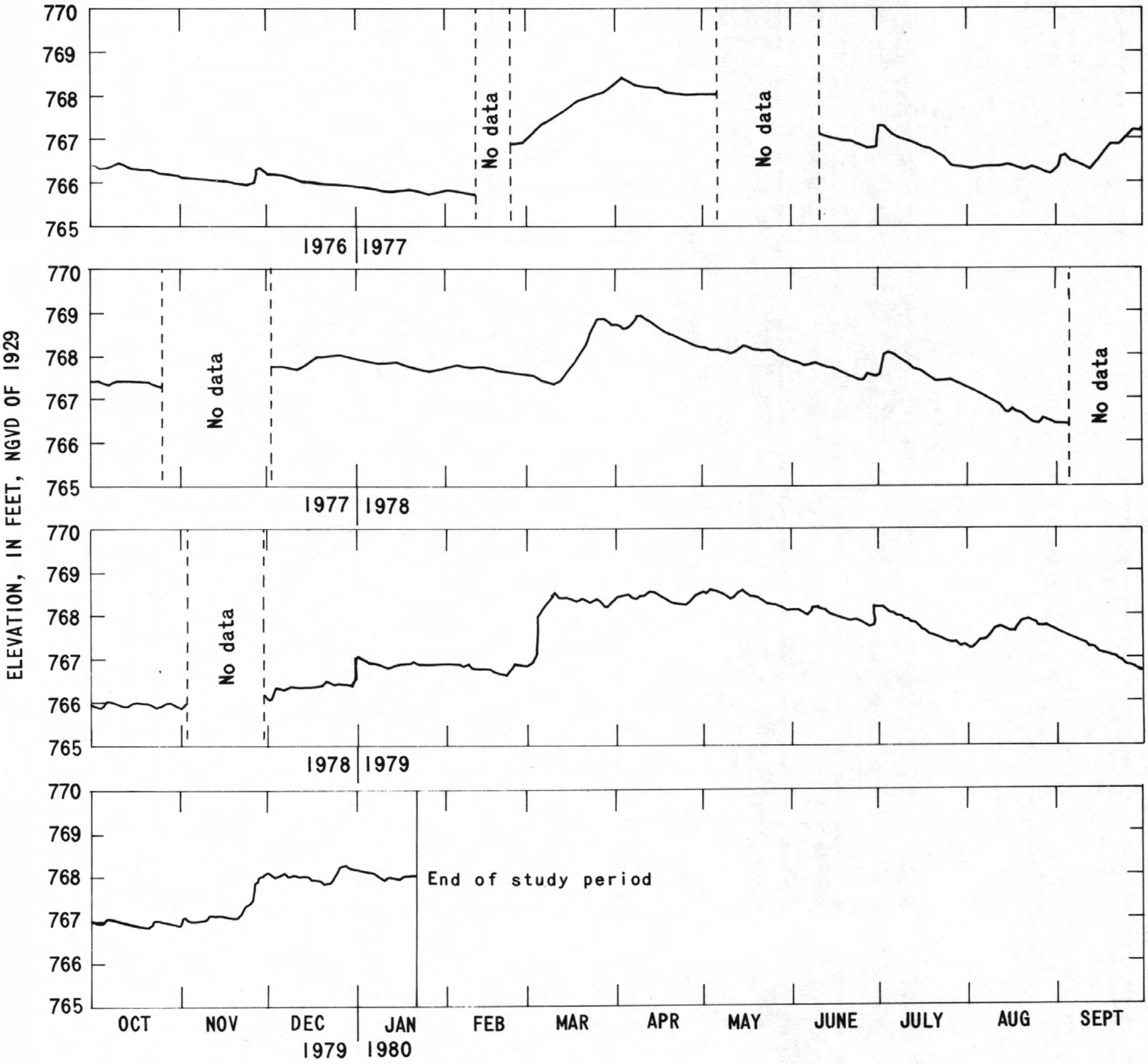


Figure 14.-- Water-level fluctuations in well Elkhart 5.

The three Elkhart well fields have a daily capacity of 22 Mgal. The highest daily recorded pumpage is 18 Mgal. Most of the pumping is done at the North Main Street well field, owing to its central location and its proximity to Christiana Creek. A dam on the creek at the well field creates a series of interconnected recharge ponds. The recharge from these ponds to the ground-water system keeps water levels high despite the heavy pumping. No major change in pumpage is likely within the next 5 years.

The major industry withdraws water from the unconfined and the confined aquifers. The industry returns some water to the unconfined aquifer by tile fields and recharge ponds.

Ground water is pumped for irrigation in the northeast section of the study area as well as in southern Michigan. Irrigation pumping, seasonal and highly variable from year to year, depends primarily on weather conditions. For these reasons and because the data would have been of little use in accomplishing the objectives of the study, a detailed evaluation of irrigation pumping was not done.

Summary of Hydrology

The Elkhart area is underlain by areally extensive, thick outwash aquifers composed mainly of sand and gravel. In much of the area, a confining silt and clay bed separates the outwash deposit into two aquifers, an upper unconfined aquifer and a lower confined aquifer. Near the landfill and the Elkhart Municipal Airport, this bed is absent. Estimates of the hydraulic conductivities of the aquifer materials are typical for outwash deposits. Data collected indicate that the aquifers are highly transmissive in most areas. The specific yield and the storage coefficient estimated for the confined and the unconfined aquifers, respectively, are within the ranges typical for outwash systems. Ground-water flow is generally horizontal and toward the St. Joseph River. However, in some areas near the river, ground water flows upward into the river. Most of the creeks and rivers are gaining streams that are apparently well connected with the aquifers. Water levels fluctuate from 2 to 4 ft/yr. The fluctuation is primarily due to variations in recharge to the aquifers, which is probably caused by evapotranspiration in the unsaturated zone. The Elkhart Water Works and a major industry pump 85 percent of the ground-water withdrawn.

MODELING OF GROUND-WATER FLOW

Generalized Concept of Model

Digital-modeling techniques were used to simulate the movement of water in the aquifers and estimate the hydrologic effects of the proposed ground-water pumpage at the Elkhart Municipal Airport. The construction of the ground-water flow model was based on the authors' visualization of the ground-water flow system. Because the unconsolidated deposits cannot be modeled precisely, assumptions were made that reduced the complexity of the model design but did not significantly affect the accuracy of the results.

Modeling Assumptions

Assumptions made for the model simulating the ground-water system include:

1. The flow system is three dimensional.
   a. Flow in the aquifers is horizontal. There is no horizontal flow in the confining bed.
   b. Flow through the confining beds between the aquifers is vertical.
2. The aquifers are homogeneous and isotropic.
3. The bedrock is impermeable.
4. The ground-water flow system is at steady state.
5. The specific yield and the storage coefficient of the unconfined and confined aquifers, respectively, are uniform throughout the modeled area. Storage in the confining bed is negligible.

Assumption 1, part a, was made even though some of the flow in the confining bed is horizontal. This flow is probably insignificant compared to the horizontal flow through the aquifers.

Assumption 3 was made on the basis that the hydraulic conductivity of the shale is several orders of magnitude less than that of the aquifers. Hydraulic conductivity of the shale ranges from 1.3×10^{-7} to 1.3×10^{-3} ft/day (Freeze and Cherry, 1979, table 2.2). The contribution to the total flow of the system by the bedrock is therefore negligible.

The potentiometric surfaces of May 1979 and the seepage runs of September 1979 for Pine Creek and May-June 1979 for all other streams were assumed to represent the ground-water flow system at steady state (assumption 4).

Model Selection and Design

The ground-water system was analyzed with a quasi-three-dimensional model by Trescott (1975). The basic equation used in the model is the following partial differential equation:

$$\frac{\partial}{\partial x}\left(K_{xx}b\frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_{yy}b\frac{\partial h}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_{zz}b\frac{\partial h}{\partial z}\right) = S_s\frac{\partial h}{\partial t}\pm W(x,y,z,t)$$

In this equation, K_{xx}, K_{yy}, and K_{zz} are the principal components of hydraulic conductivity; h is the height of ground water above some arbitrary datum; b is the saturated thickness of the aquifer; t is time; S_s is the specific storage of the aquifer; and x, y, and z are Cartesian coordinates. The variable, W, includes well discharge, transient leakage from confining beds, direct recharge for precipitation, and evaporation. The sign depends on whether water is being added to (+) or withdrawn from (-) the system.

The ground-water system was modeled with two layers. The upper layer (layer 2) represents the unconfined aquifer, whereas the lower layer (layer 1) represents the confined aquifer. In areas of no confining bed, the unconfined aquifer was modeled in both layers 1 and 2. The geohydrologic section of the model shown in figure 15 is based on the geologic section shown in figure 4.

A finite-difference grid was constructed (fig. 16) for placing the hydrologic data in a form compatible with the digital model. The grid divides the study area into rectangular-shaped blocks. Model-parameter values, obtained by averaging the values over the entire area of the block, are entered at the center of each block, defined as the node or nodal point. The grid, representing an area of 172 mi^2 (16 mi by 10 3/4 mi), consists of 54 rows and 87 columns. With two layers the number of nodes in the model is 9,396. There are 5,016 node blocks (each 800 ft by 800 ft) in the central part of the modeled area. Block sizes increase toward the model boundaries so that the largest blocks are at the boundaries. Differences in block size are never greater than 1.5 times the size of any adjacent blocks.

In areas where the confining bed is absent, the aquifer was separated into two layers at points determined by projecting the altitude of the top of the confining bed across these areas and interpolating values between the points.

Streams were modeled as internal-leakage boundaries (fig. 16). The model calculates vertical leakage at each stream node by taking the product of the vertical hydraulic conductivity of the streambed, the gradient resulting from the difference between river stage and the water level in the aquifer below, and the ratio between the area of the stream and the area of the finite-difference block in which the stream is located.

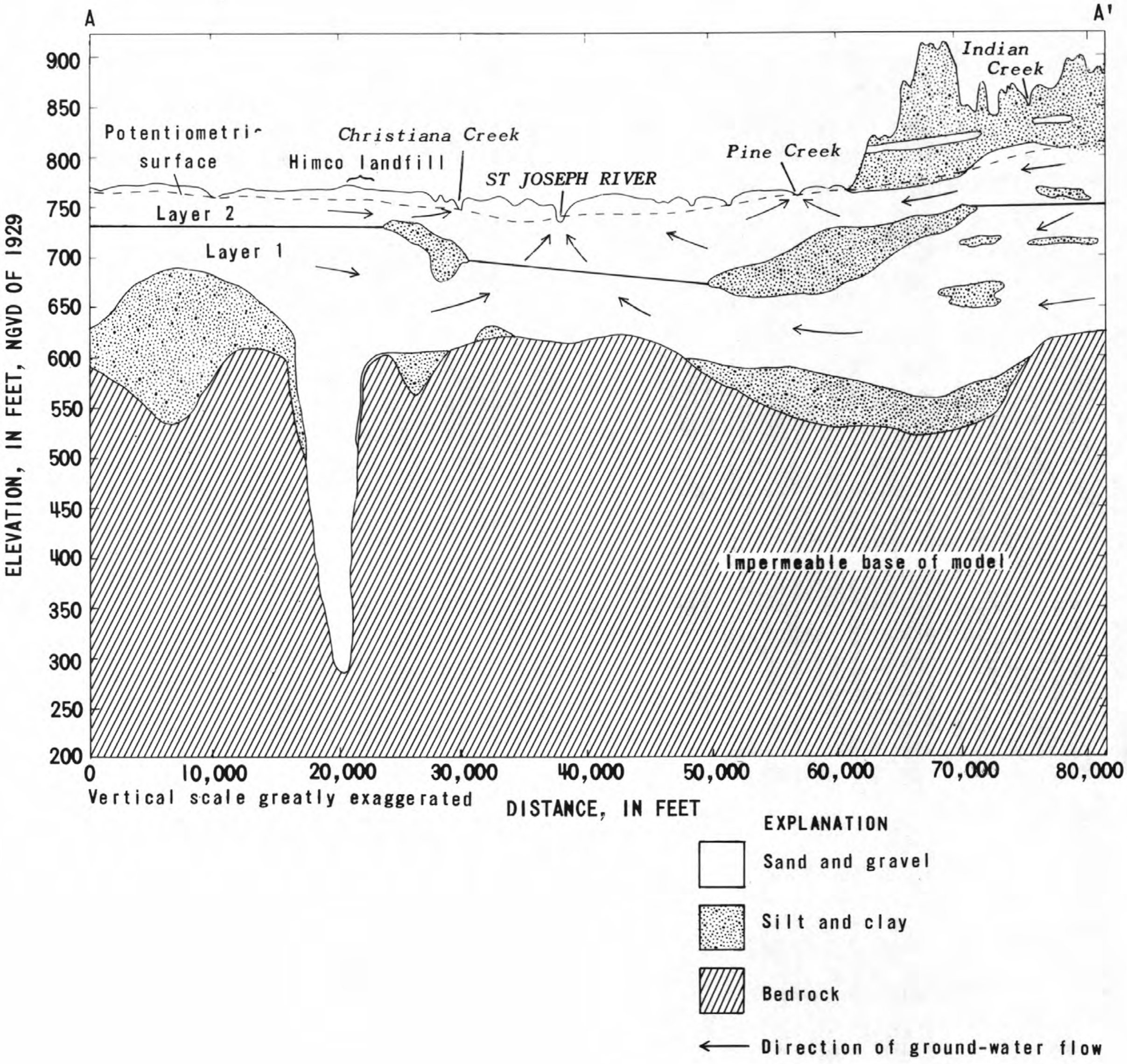


Figure 15.-- Geohydrologic section showing layering scheme for model.

Boundary Conditions

One of the most important aspects of any model is the treatment of the boundaries. Boundaries must be represented as accurately as possible or be far enough away from any simulated stresses so that they will not be significantly affected by those stresses.

Two types of boundary conditions were used in the flow-model analyses, constant head and constant flux. Constant-head boundaries allow no drawdown at the boundaries, even when the system is highly stressed nearby. The model provides all the water necessary to maintain the zero drawdown at the boundary nodes. This causes maximum flow into the system from the boundaries in response to a simulated pumping stress. Constant-flux boundaries do not allow additional flow in or out of model boundaries other than that at steady state. For this reason, constant-flux boundary conditions produce maximum drawdowns.

During the calibration procedure, all model boundaries for the upper and lower layers were treated as constant head. Constant heads are usually used because heads can be measured, whereas fluxes must be calculated. Constant-flux and constant-head boundaries were used in all steady-state and transient pumping simulations. Use of the two boundary conditions provides a range of possible effects.

Model Calibration

Before the model can be used as a tool to predict the effects of any new stress, it must simulate steady-state conditions as accurately as possible with the available data. The calibration procedure is a series of experiments that involves varying the model parameters until they match the measured water levels and river seepages.

One of the least known parameters used in the model was the vertical hydraulic conductivity of the confining bed. This parameter was initially set to 0.007 ft/d. With this value, model-simulated vertical head gradients in the area where the confining bed is present were greater than measured values. Subsequent experiments were made by increasing the vertical hydraulic conductivity of this unit. The value of vertical hydraulic conductivity that produced the best match over the modeled area was 0.07 ft/d. This result lies in an upper range determined in a series of aquifer tests done for tills in Ohio, Illinois, and South Dakota (Norris, 1962).

In areas near the St. Joseph River where the confining bed is absent, measurements of upward head gradients between layers were not reproduced by the model. Model-simulated gradients were approximately 1 ft or less between

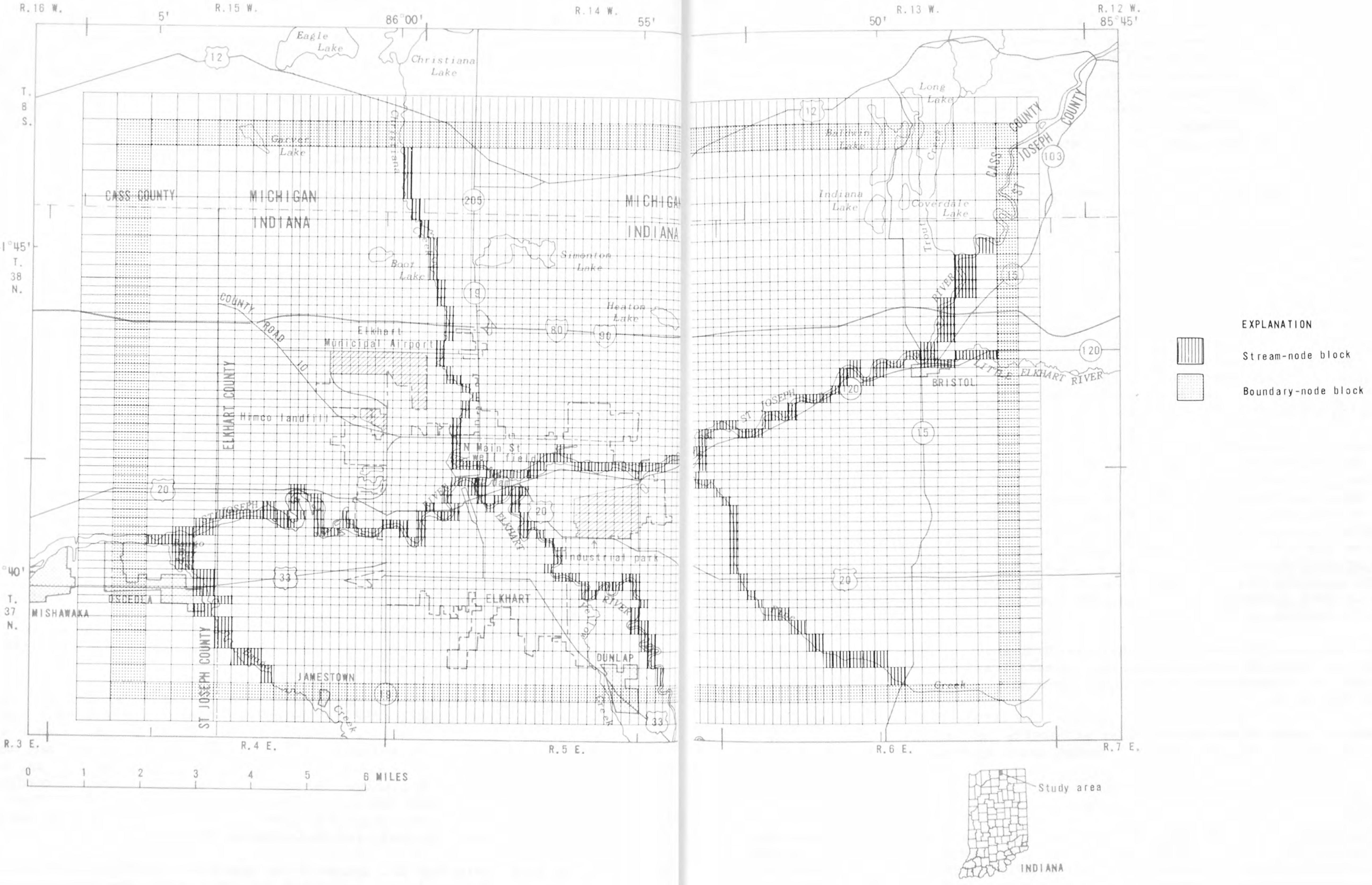


Figure 16.-- Finite-difference grid used in the model.

layers 1 and 2 in this area, whereas measured gradients were approximately 3 ft. The reason for this difference may be that the computed head at the node is averaged over the entire grid block. Another possible explanation is that not enough layers representing the aquifer system were simulated in the model, where the confining bed is absent, for accurately reproducing the measured vertical head gradients. Representing the aquifer system with smaller grid blocks and (or) more layers in the area would produce more accurate vertical head gradients.

Streambed-leakance values were calibrated by matching model-simulated river seepages for each reach to the range of measured seepages and also by matching model-simulated water levels near the streams to field-measured water levels. (See table 1.) Calibrated values for streambed leakance at river nodes ranged from 0.07 to 1.00 ft/d, but most values were near 0.5 ft/d. These values were simulated for a streambed thickness of 1 ft for all rivers included in the model. The value 0.07 ft/d corresponds to a short stretch of the Elkhart River near the mouth, where the river flows over sandy clay, whereas the 1.00 ft/d value corresponds to a short stretch of Christiana Creek north of the Michigan State line, where the streambed seems to be composed of coarse gravel. A high value of 6.7 ft/d was used for three river nodes on Christiana Creek at the North Main Street well field because the recharge pond beds are periodically cleaned to allow maximum recharge to the pumping wells.

Changes in transmissivity were minor during model calibration. Transmissivity was changed by more than 30 percent from the original values in only one area. This area was the bedrock valley, where the model-simulated water levels were lower than measured values by about 6 ft. The hydraulic conductivity of the deposits in the valley was reduced to 40 ft/d. This reduction was applied in only layer 1, as the bottom of layer 2 was modeled to terminate above the rim of the valley. Lowering the transmissivity raised water levels in both layers 2 to 3 ft in the vicinity of the bedrock valley and produced a better match of model-simulated and measured water levels. The change in this parameter did not significantly affect water levels or river leakages elsewhere in the model.

Final model-calibrated transmissivity distributions for the unconfined and confined aquifers are presented in figures 17 and 18, respectively. The transmissivity distribution in the area where the confining bed is absent is shown in figure 19.

Areal recharge values of 12 in./yr within the outwash valley and 4 in./yr in the till plains (fig. 2) were used during model calibration.

Table 1.--Model-simulated and measured seepages

Streams and reaches	Model-simulated seepages Positive value, river reach gaining; negative value, river reach losing (ft^3/s)	Measured seepages Range based on measurement error. Positive value, river reach gaining; negative value, river reach losing (ft^3/s)
Christiana Creek		
Reach 1	12.3	4.4 ------- 18.3
Reach 2	2.6	-3.7 ------- 11.7
Reach 3	-12.4	-18.5 ------- -3.7
Baugo Creek		
Reach 4	4.5	4.4 ------- 6.8
Elkhart River		
Reach 5	13.4	3.6 ------- 50.5
Reach 6	8.9	-34.9 ------- 9.9
Reach 7	11.5	2.5 ------- 45.5
Pine Creek[1]		
Reach 8	2.8	1.8 ------- 3.3
Reach 9	1.5	1.4 ------- 2.3

[1] Seepages for Pine Creek were measured in September 1979. Seepages for all other streams were measured in May-June 1979.

After each calibration run, model-simulated heads and river seepages were compared with measured values. The root mean square error (RMSE) was used to compute a measure of the degree of fit for head values in each calibration run:

$$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{N} (h_i^m - h_i^s)^2}{N}} ,$$

where h_i^m is the measured head, h_i^s is the model-simulated head, and N is the total number of observation points. A ratio (R) was calculated for layers 1 and 2 in each calibration run by the equation:

$$R = \frac{\mathrm{RMSE}}{\Delta h} ,$$

where RMSE is the calculated root mean square error value and h is the difference between the minimum and the maximum values of model-simulated heads in the modeled area. RMSE and R were used to determine which parameter values produced the best match of simulated water levels. Computed R values for the final calibrated model were 0.04 and 0.03 for layers 1 and 2, respectively.

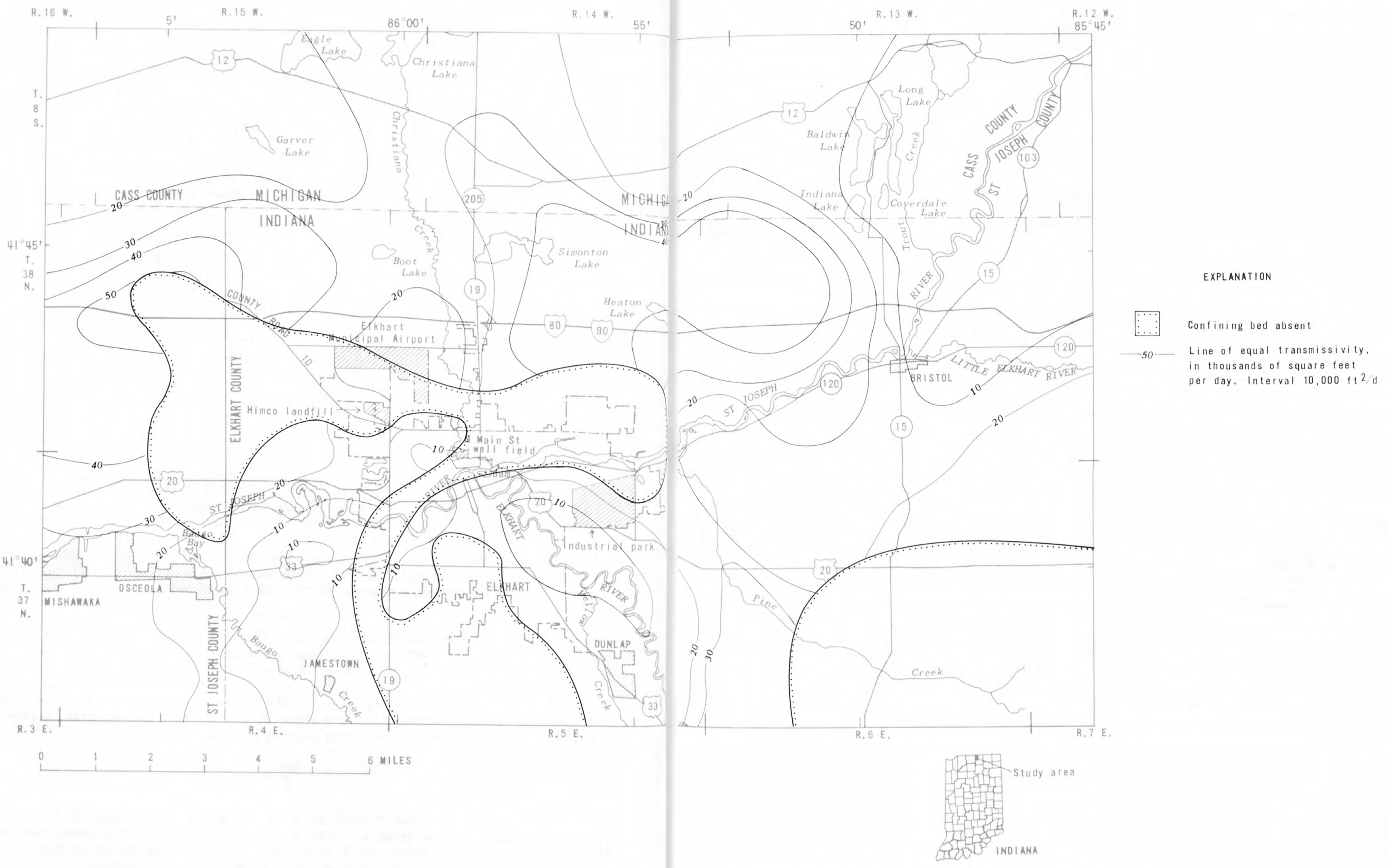


Figure 17.-- Model-calibrated transmissivity of the unconfined aquife where confining bed exists.

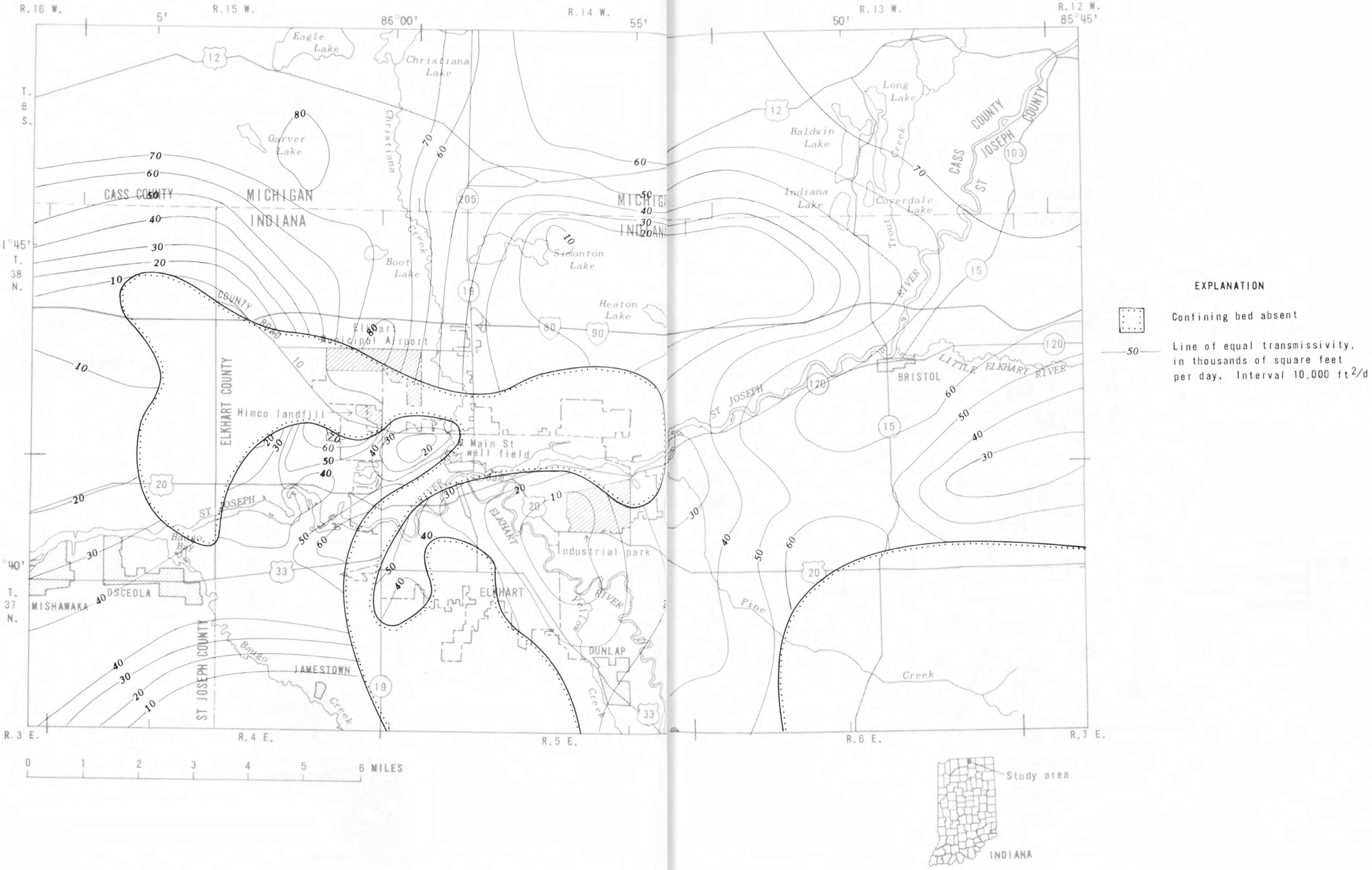


Figure 18.-- Model-calibrated transmissivity of the confined aquifer.

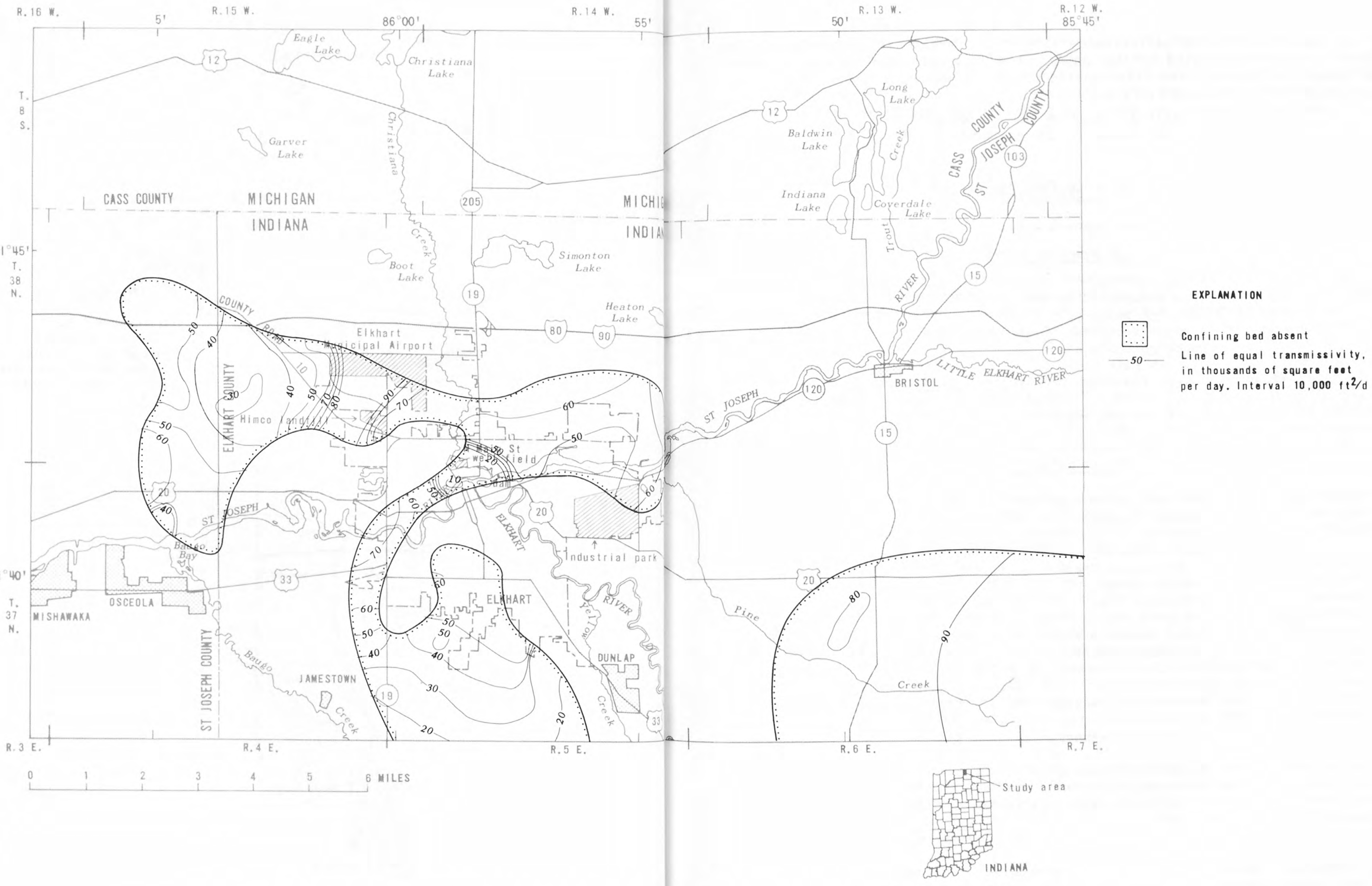


Figure 19.-- Model-calibrated transmissivity of the unconfined aquifer where the confining bed is absent.

A comparison of the calibrated steady-state results for water levels in layers 1 and 2, simulated by the model, with measured water levels is shown in figures 20 and 21. The distribution of sources and discharges in the calibrated model is shown in table 2.

Table 2.--Sources and discharges simulated in the calibrated model

[Data in cubic feet per second]

Sources of water		
Areal recharge		118.1
Flow across boundaries		204.7
Recharge ponds at major industry		2.5
Recharge ponds at North Main Street well field		12.4
Total sources		337.7
Discharges of water		
Flow across boundaries		43.4
Pumping (layers 1 and 2)		17.6
River leakage:		
Christiana Creek	17.3	
Baugo Creek	5.7	
Elkhart River	37.6	
Pine Creek	4.6	
St. Joseph River	209.0	
Little Elkhart River	2.8	
Total river leakage		277.0
Total discharges		338.0
Percent difference between sources and discharges		.09

Comparison of the model-simulated water-level maps with the contour maps prepared from measured water levels (figs. 11 and 12), suggests that the model has reproduced the general flow pattern. The model was not able to reproduce some small flow anomalies, such as the one near the hydroelectric dam, probably because of the large-sized grid blocks used in modeling these areas.

Sensitivity Analyses

After model calibration, sensitivity analyses were done to test the response of the model to changes in transmissivity, streambed leakance, vertical hydraulic conductivity of the confining bed, and areal recharge rates. Areas of the model that are sensitive to certain parameter adjustments were defined. Model-prediction capability could be improved by concentrating data collection in these areas.

The first step in the sensitivity analyses consisted of making a series of runs in which one parameter was reduced by half over the entire model area while all others were held to their calibrated value. After one parameter was reduced, it was returned to its calibration value, and another parameter was similarly reduced. Model runs were done until all parameters were individually reduced.

Root mean square error values were calculated for each sensitivity run. All computed values were less than 2.5 feet. The low values indicate that the parameter adjustments used for the sensitivity analyses were not sufficient to produce large changes in head throughout the modeled area.

The changes in water levels produced by the transmissivity adjustment for layers 1 and 2 are shown in figures 22 and 23, respectively. Decrease in head was greatest near the St. Joseph River west of the hydroelectric dam. Head increased generally north of the St. Joseph River. Flow into rivers and across model boundaries simulated in the calibrated model and the sensitivity analyses are presented in table 3. Decreasing transmissivity throughout the model area decreases ground-water flow between recharge and discharge areas. This tends to reduce heads at and near the streams and to increase them away from the streams. Flow into the rivers and across model boundaries was reduced.

The model was most sensitive to the decrease in transmissivity at the major industrial pumping center. Change in head was greatest in layer 1, representing the confined aquifer, where most of the pumping takes place. This result indicates that the model is most sensitive to the adjustment in transmissitivity at stressed areas.

As shown in table 3, decreasing the vertical hydraulic conductivity of the streambed reduced the amount of ground water discharging into the rivers. Consequently, there was a net increase in flow leaving the system across model boundaries. Reducing the stream-aquifer connection raised water

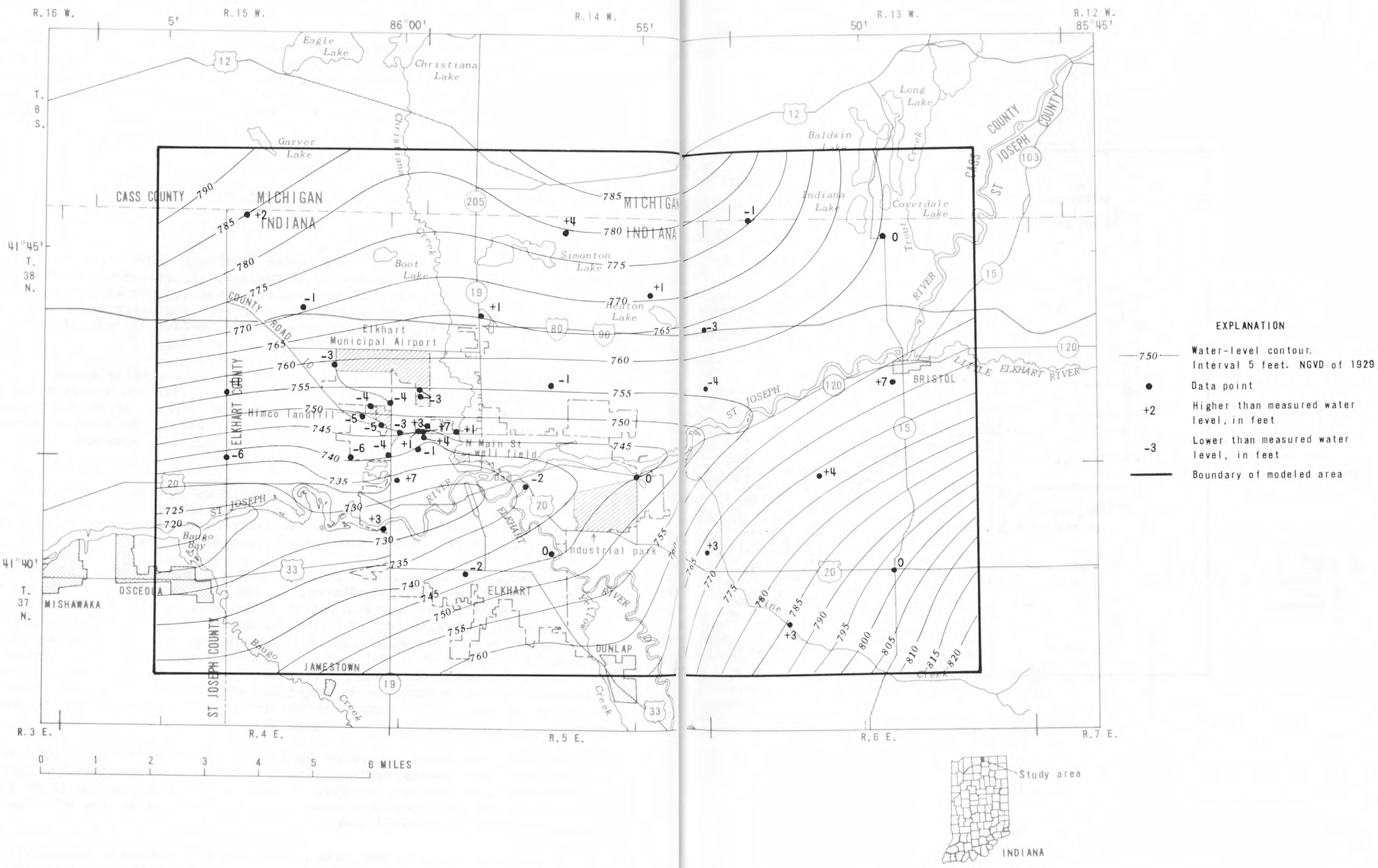


Figure 20.-- Model-simulated steady-state water levels, layer 1.

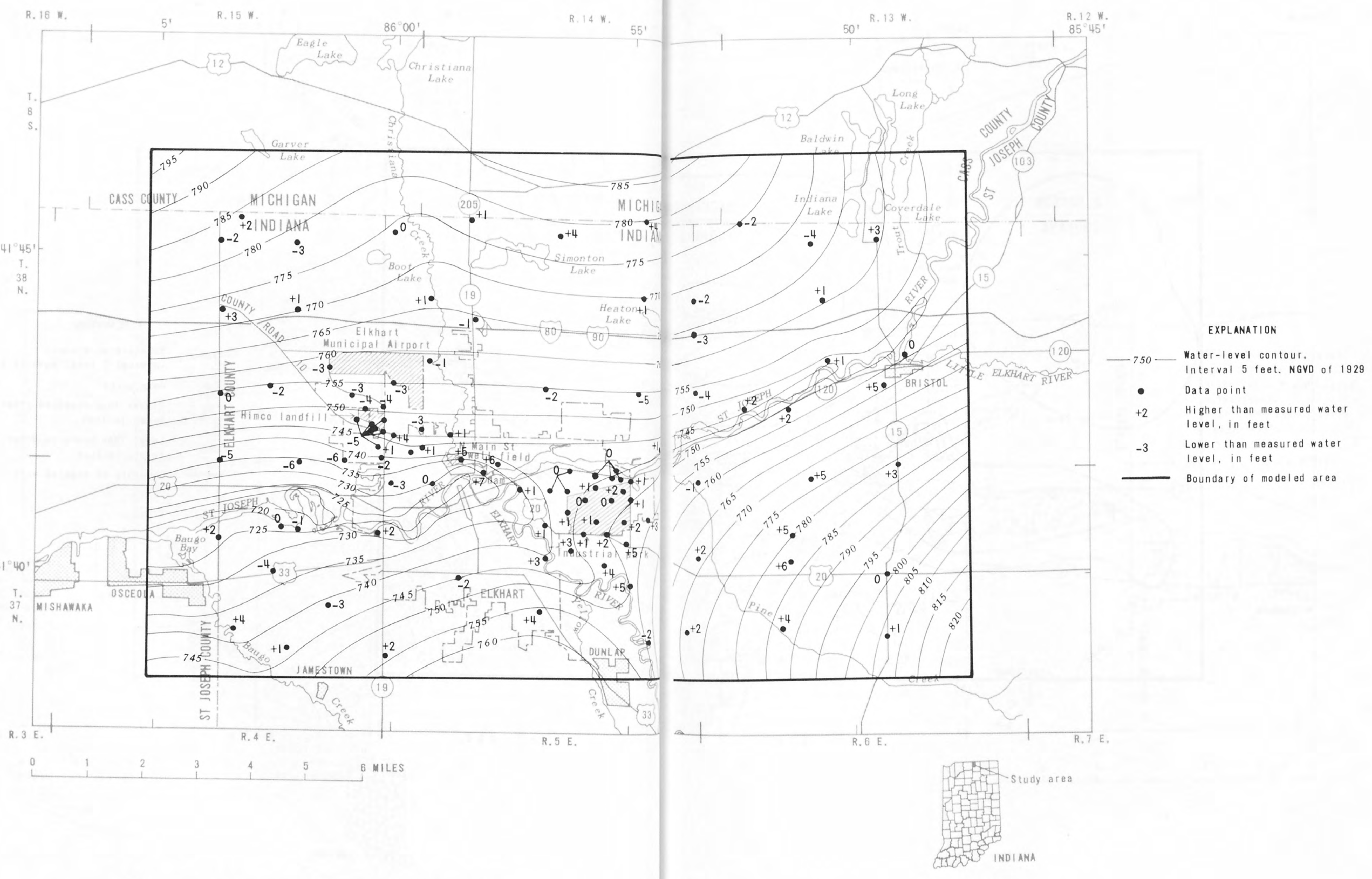


Figure 21.-- Model-simulated steady-state water levels, layer 2.

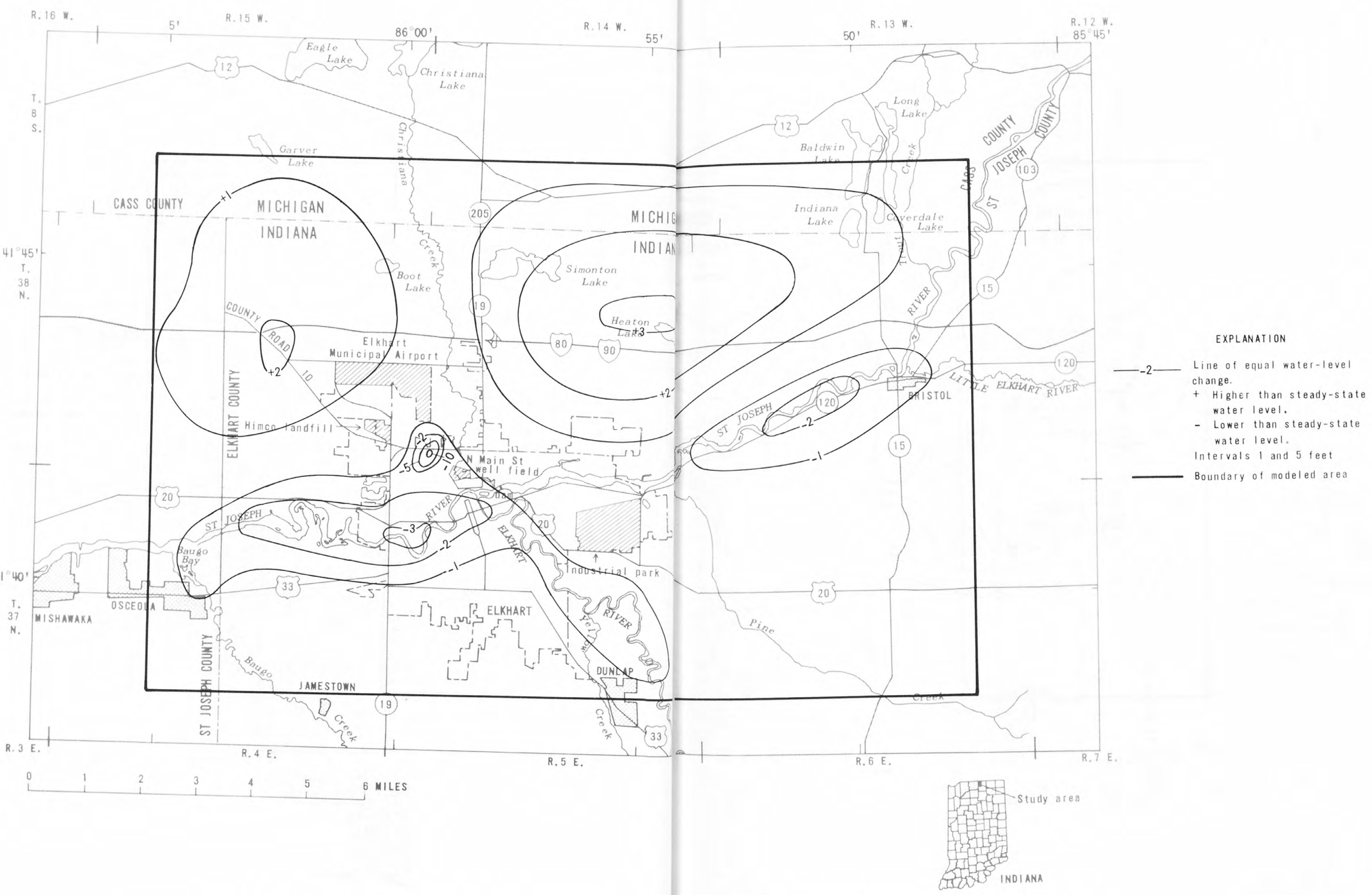


Figure 22.-- Changes in water levels in layer 1 due to a 50-perce decrease in the calibrated transmissivity.

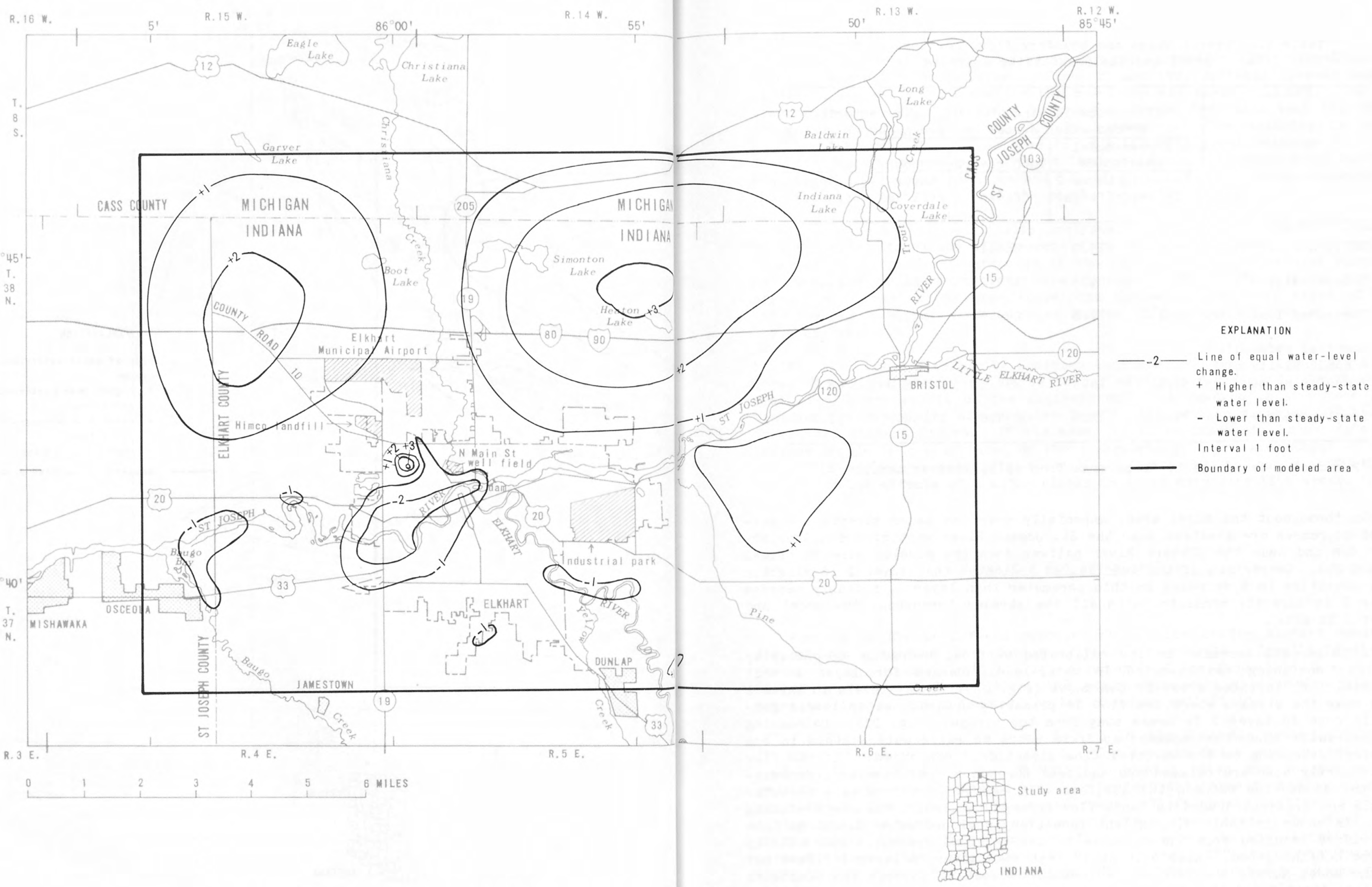


Figure 23.-- Changes in water levels in layer 2 due to a 50-percent decrease in the calibrated transmissivity.

Table 3.--River leakage and boundary flows for the calibrated model and the sensitivity analyses

	Flow leaving ground-water system			Flow entering ground-water system	
	Across model boundaries		Into rivers	Across model boundaries	
	Layer 1 (ft^3/s)	Layer 2 (ft^3/s)	(ft^3/s)	Layer 1 (ft^3/s)	Layer 2 (ft^3/s)
Calibrated model	28.5	15.0	266.1	140.1	64.6
Transmissivity[1]	15.7	9.8	169.2	64.4	28.7
Streambed leakance[1]	31.1	17.0	249.5	133.5	61.9
Vertical hydraulic conductivity of confining bed[1]	32.1	14.4	254.3	133.8	63.8
Areal recharge[1]	24.9	12.5	234.0	154.2	71.7

[1] Parameters decreased by 50 percent from calibrated values.

levels throughout the model area, especially near the major streams. Water-level increases are greatest near the St. Joseph River west of the hydroelectric dam and near the Elkhart River halfway down the modeled stretch (figs. 24 and 25). Comparison of the two figures indicates that layer 2 is slightly more sensitive to a decrease in this parameter than layer 1, probably because layer 2 is directly connected with all the streams throughout the model and layer 1 is not.

A 50-percent decrease in the calibrated vertical hydraulic conductivity of the confining bed resulted in water-level changes for layer 1 that exceeded 1 ft in three areas of the model (fig. 26). Water levels in layer 1 rose near the streams where the flow is primarily upward. Water levels generally rose in layer 2 in areas away from the streams (fig. 27). Decreasing the hydraulic connection between aquifers tends to raise water levels in one aquifer, depending on the vertical flow direction. For example, if the flow is primarily downward between two aquifers through a confining bed, decreasing the connection between the aquifers will result in an increase in water levels and gradient needed to cause flow through the less permeable confining bed. As shown in table 3, a slight reduction in ground-water discharge into the rivers resulted from the decrease in the vertical hydraulic conductivity of the confining bed. Also evident is that some water in layer 1 flowed out of the model across the boundaries instead of leaking through the confining bed into layer 2 and then discharging into the rivers.

Reducing the areal recharge by 50 percent produced slightly greater drawdowns in layer 2 than in layer 1 (figs. 28 and 29), probably because layer 2 is directly areally recharged in the model whereas layer 1 is not. The similarity in water-level declines in the two layers indicates that the layers are hydraulically well connected. An increase in flow entering the model across the boundaries, in response to adjustment in areal recharge, is shown in table 3. The boundary conditions imposed in the calibrated model buffered the effect of reducing recharge and lowering water levels inside the model by increasing the net flow across the constant-head boundaries.

In another set of sensitivity analyses, each of the previously discussed parameters was increased 100 percent from calibrated values. The resulting water-level changes were generally in the same area and of the same magnitude as those where the parameters were decreased 50 percent but of opposite sign. For example, for the "50-percent decrease" analyses, areas of head increase are areas of head decrease in the "100-percent increase" analyses, and conversely.

Results of the sensitivity analyses indicate that only a few areas of the model were significantly sensitive to any of the parameter adjustments. The most significant result of the analyses was that changes in streambed leakance and transmissivity affect water levels near certain stretches of the St. Joseph and Elkhart Rivers. If the model is to be improved, future data collection should be concentrated on these parameters in and near these stretches. This approach especially applies if the model is to be refined for predicting the affects of a major stress in these more sensitive areas.

Simulated Ground-Water Pumping

Pumping at the well field proposed for a site at the Elkhart Municipal Airport was simulated in a series of three experiments. Results of the experiments were used to estimate the drawdown and the reduction in streamflow in nearby streams. In addition, the experiments were used to determine the potential impact of the landfill on the proposed well field.

The Elkhart Water Works plans to use 10 pumping wells spaced 600 to 800 ft apart at the proposed well field. The average daily pumping rates will probably range from 7.5 to 10 Mgal/d. Pumping rates for short-term peak demands will probably reach 20 Mgal/d. Each of the three preceding rates was simulated in the model at nine nodes spaced 800 ft apart. Little or no confining bed is present in a large part of the airport area. The upper layer of the model (layer 2) averages 35 ft in thickness in the vicinity of the airport, whereas the lower layer (layer 1) averages 150 ft. The pumping was simulated in layer 1 to take advantage of the full saturated thickness of the aquifer.

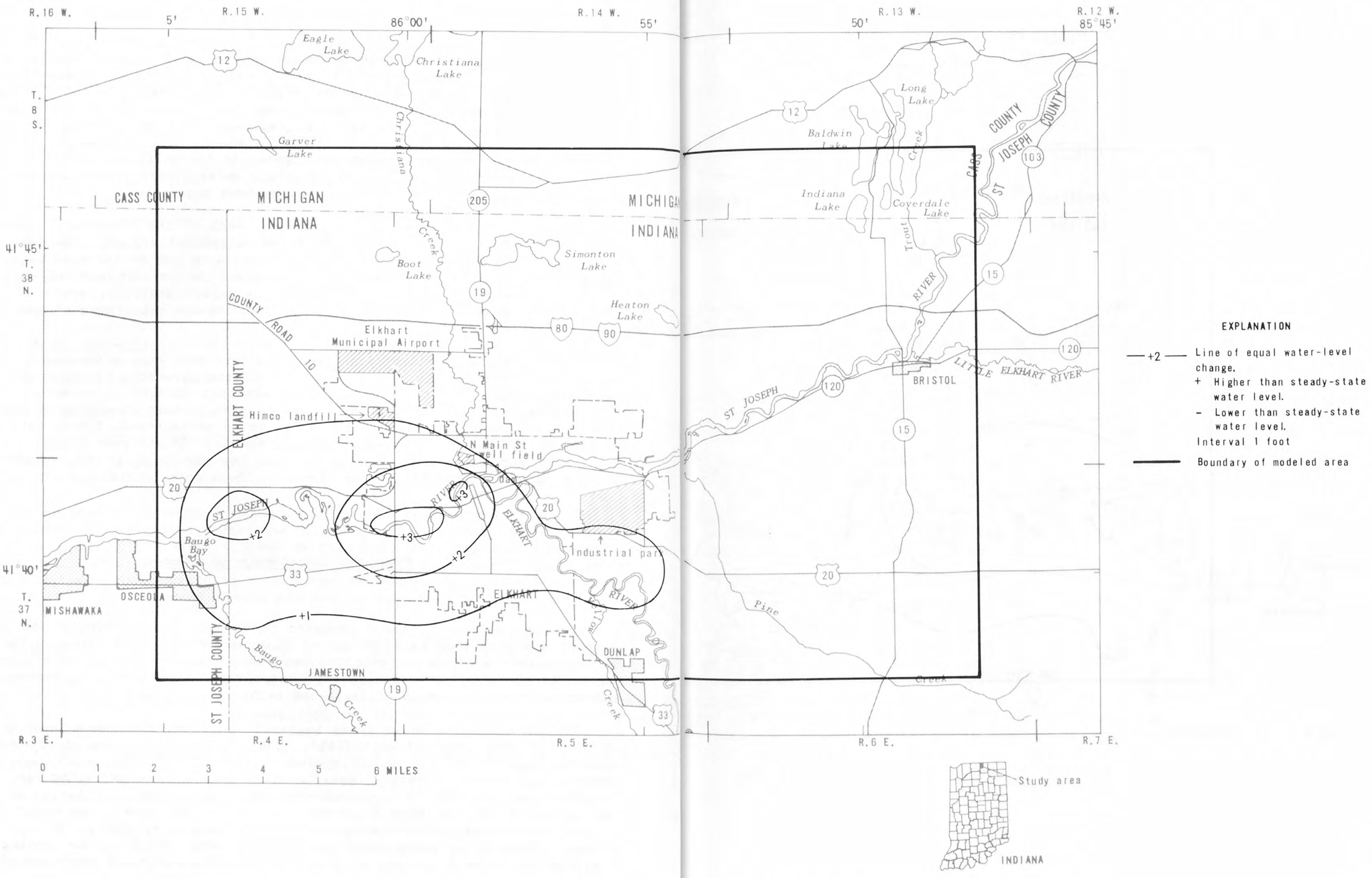


Figure 24.-- Changes in water levels in layer 1 due to a 50-percent decrease in the calibrated streambed leakance.

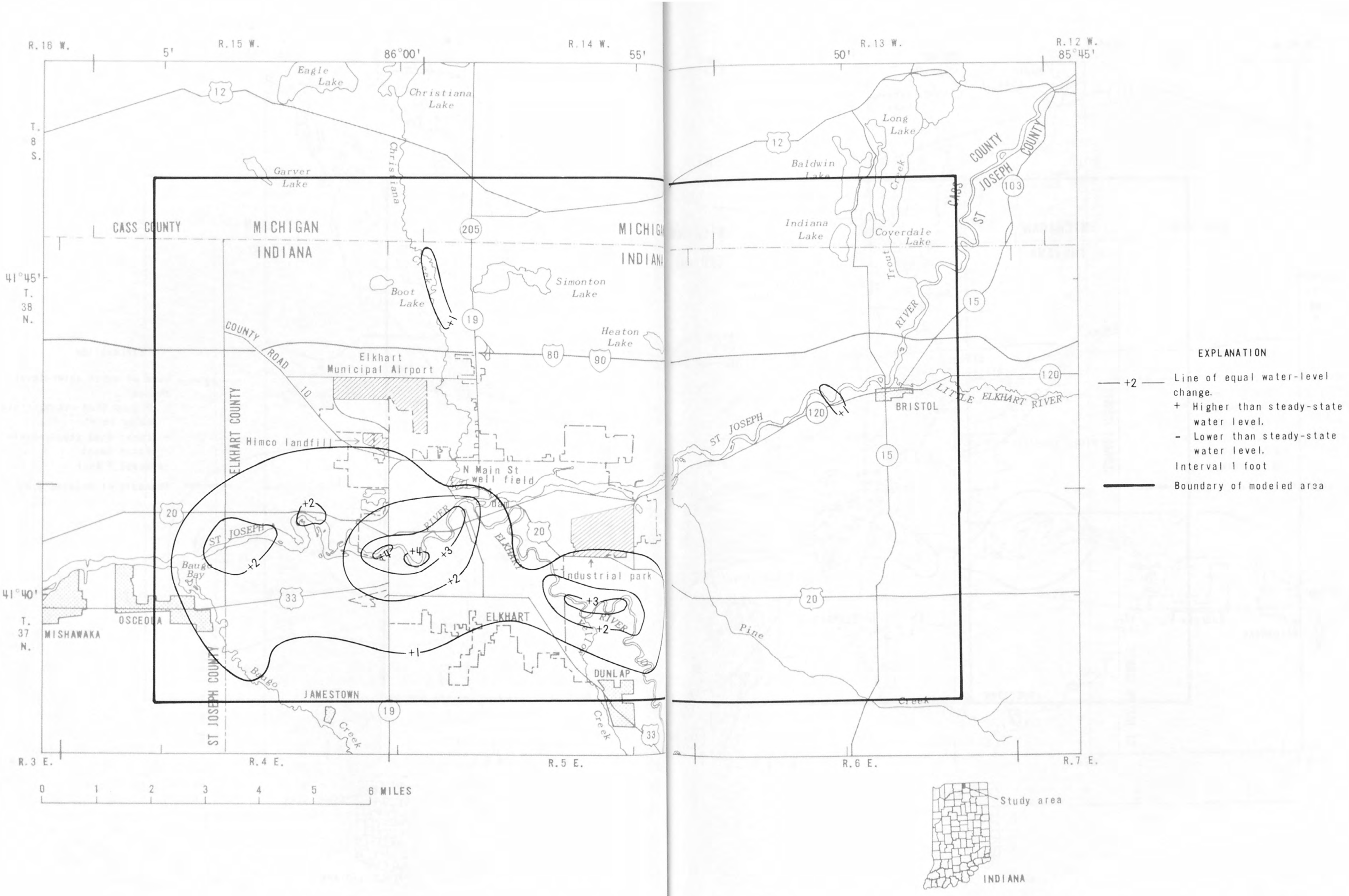


Figure 25.-- Changes in water levels in layer 2 due to a 50-percent decrease in the calibrated streambed leakance.

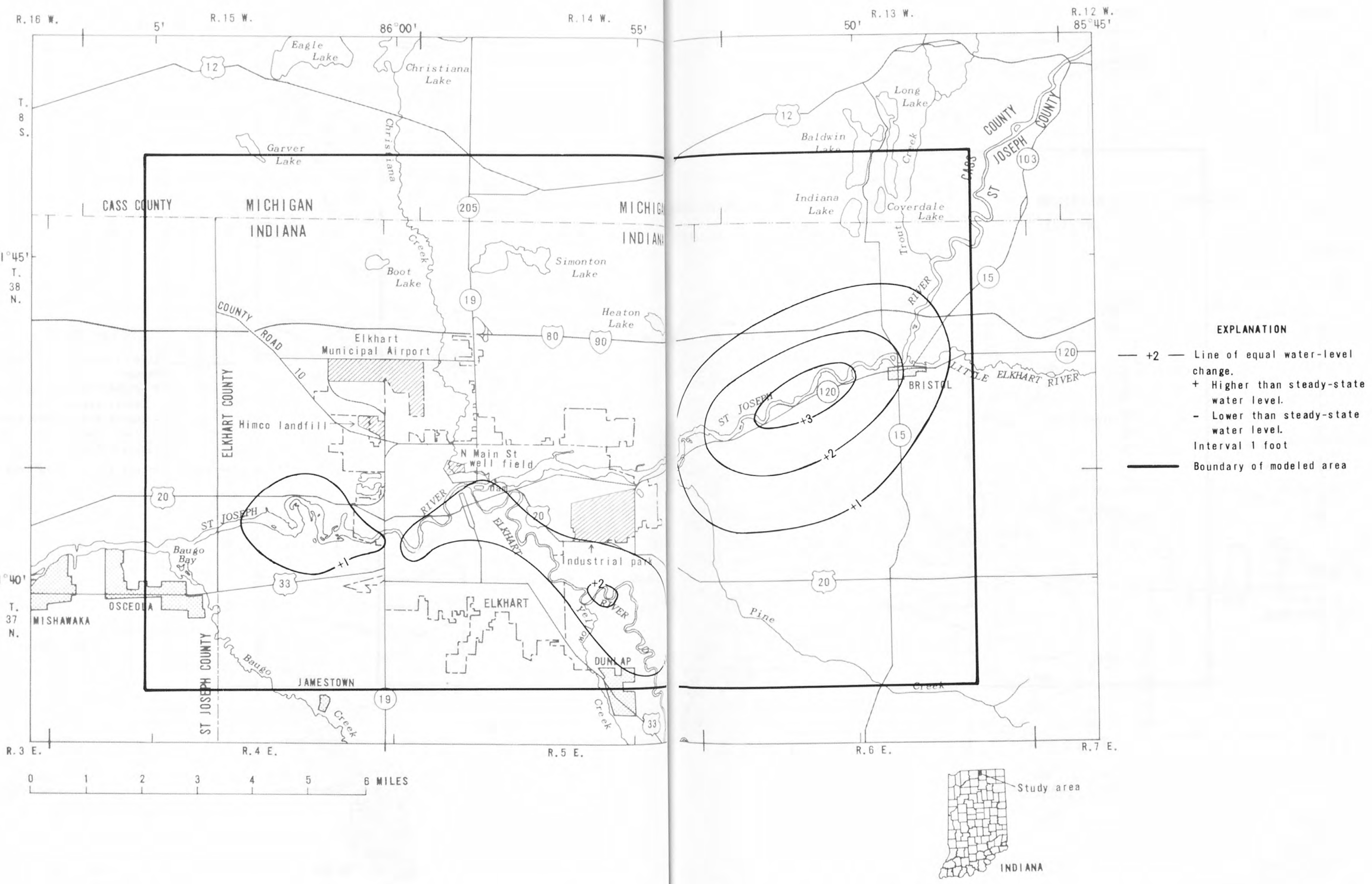


Figure 26.-- Changes in water levels in layer 1 due to a 50-percent decrease in the calibrated vertical hydraulic conductivity of the confining bed.

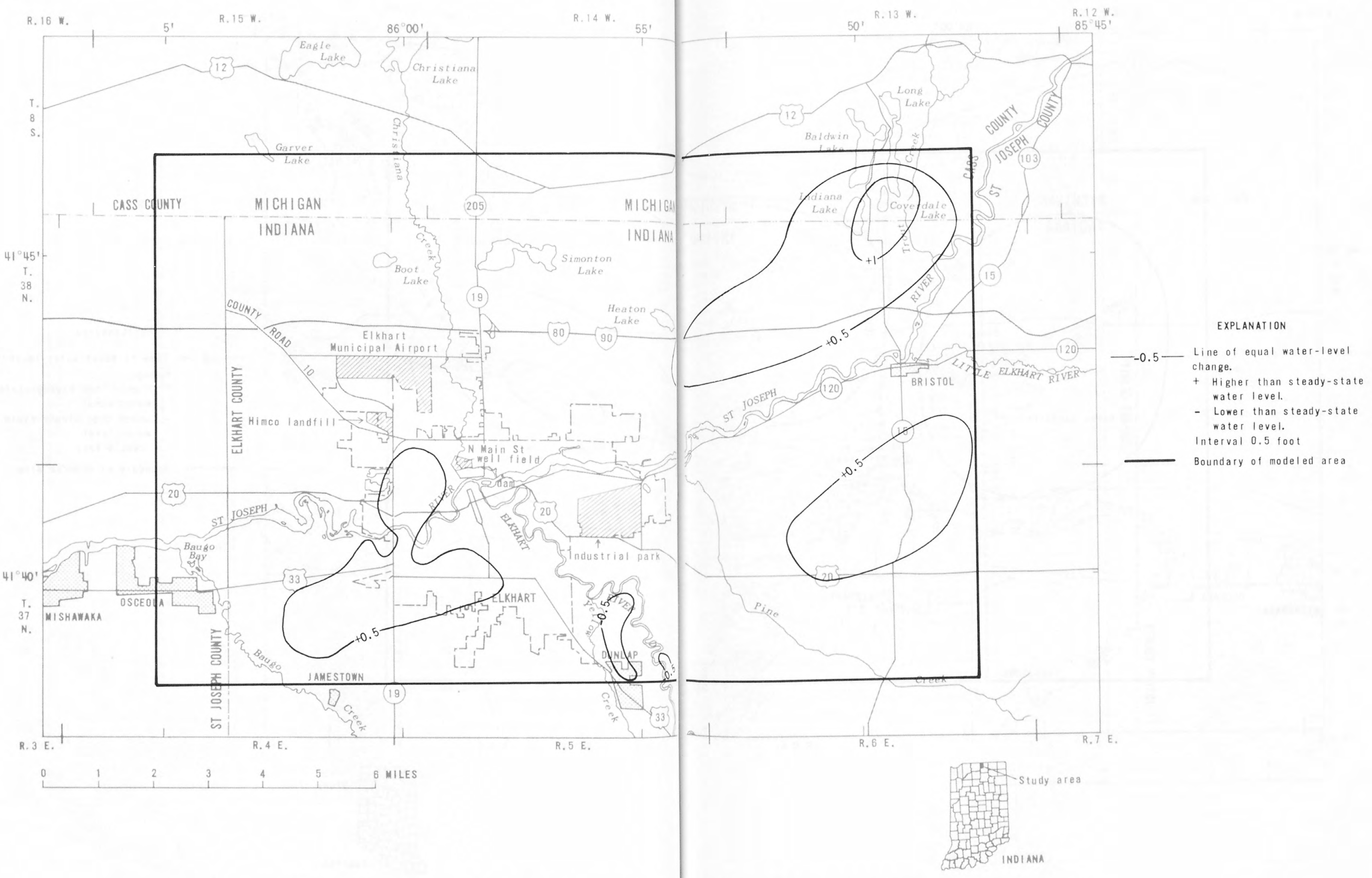


Figure 27.-- Changes in water levels in layer 2 due to a 50-percent decrease in the calibrated vertical hydraulic conductivity of the confining bed.

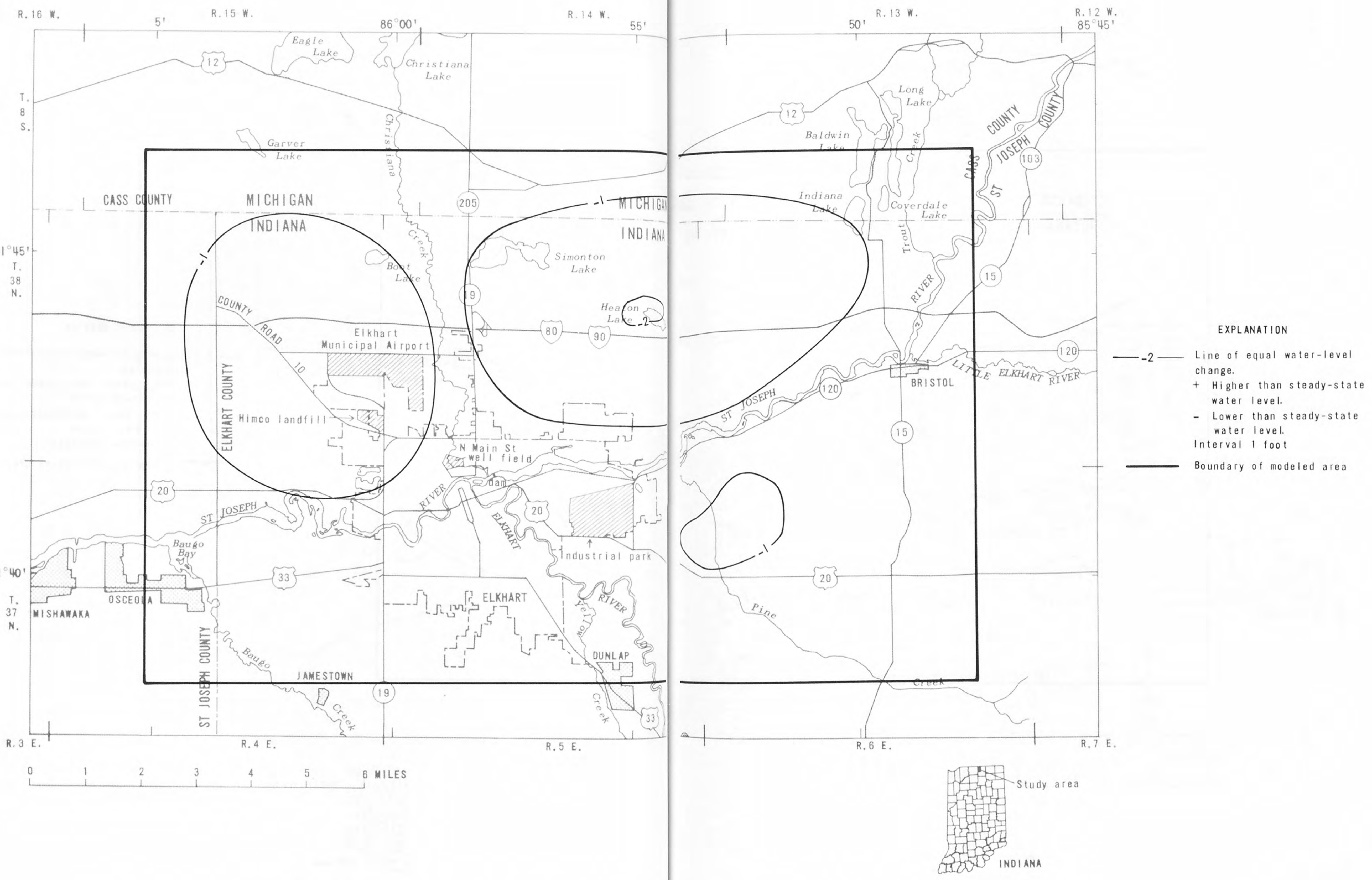


Figure 28.--Changes in water levels in layer 1 due to a 50-percent decrease in the calibrated areal recharge.

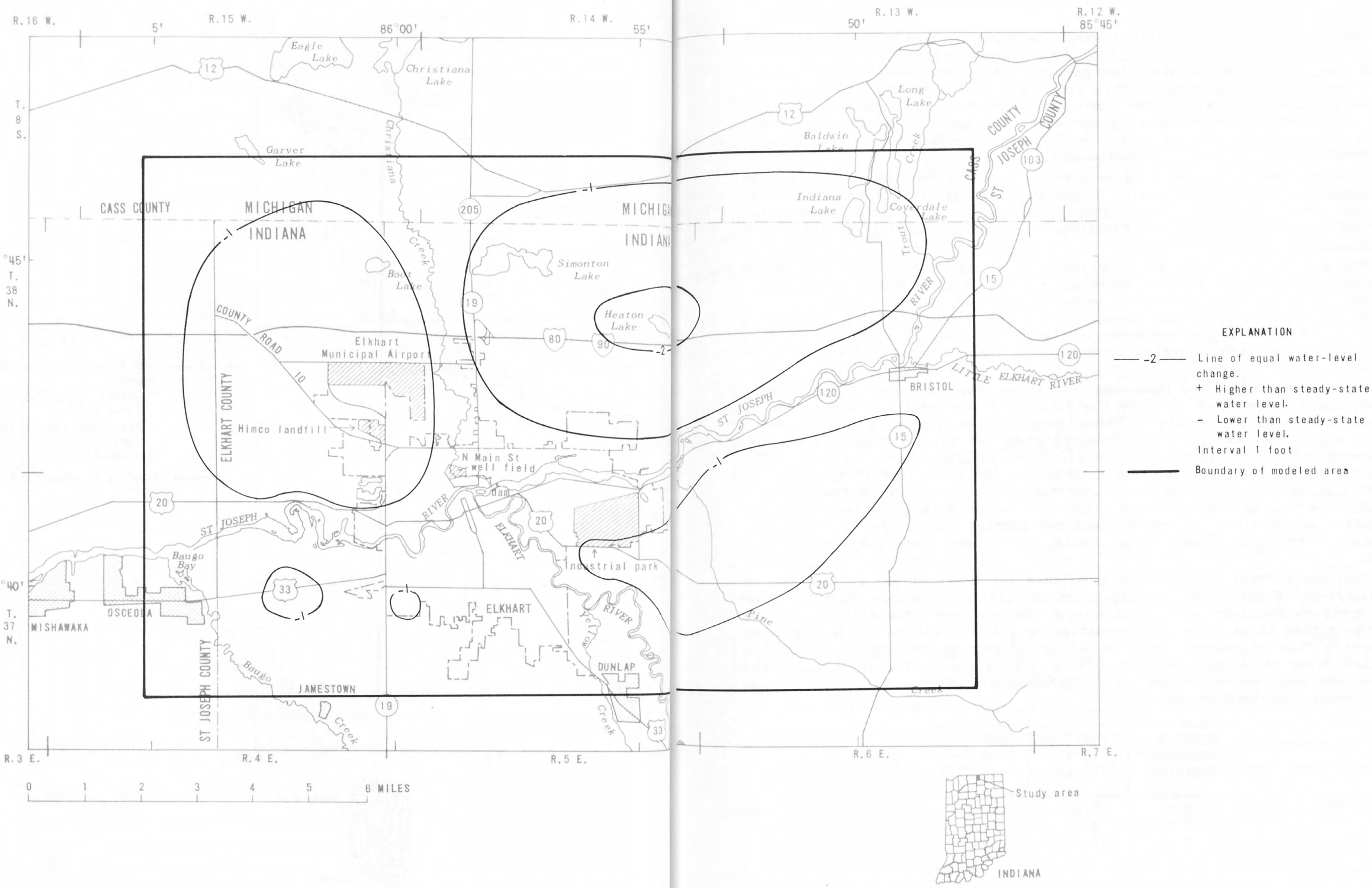


Figure 29.-- Changes in water levels in layer 2 due to a 50-perce decrease in the calibrated areal recharge.

The three pumping experiments were simulated to steady state with both constant-head and constant-flux boundaries. Drawdowns and streamflow reductions resulting from the simulated pumpings were similar for both boundary conditions in each experiment. This similarity indicated that the boundaries did not significantly affect the pumping experiments. All maps showing drawdowns or water levels produced by the model are based on experiments where constant-flux boundaries are used. Use of these boundaries resulted in maximum drawdowns and streamflow reductions.

Drawdown and water-level distributions produced by the model in each experiment were similar in layers 1 and 2. This similarity indicates that the two layers in the area of interest simulated in the model are hydraulically well connected. For simplification, only maps of layer 1 resulting from the pumping experiments are shown.

The steady-state drawdowns produced by the model for the three pumping experiments, contoured directly from the model-generated plots, are shown in figures 30-32. The steady-state water levels for each experiment in the vicinity of the airport are shown in figures 33-35, and the prepumping steady-state water levels are shown in figure 36. Flow lines were drawn perpendicular to the equipotential lines to establish direction of ground-water flow.

The drawdown maps show that simulated water-level declines at the airport range from an average of 5 ft for the 7.5-Mgal/d pumping to an average of 12 ft for the 20-Mgal/d pumping. These pumpings are highly feasible because steady-state drawdowns resulting from the 20-Mgal/d maximum pumping are about 8 percent of the saturated thickness of the aquifer at the airport. The steady-state water-level-distribution maps for the 7.5- and 10-Mgal/d simulated pumpings indicate that no flow should be diverted from the landfill area to the pumping center at the airport. The steady-state water-level-distribution map for the 20-Mgal/d simulated pumping showed that a small percentage of the total flow to the pumping center may come from the landfill area.

Christiana Creek was affected more than any other stream in the pumping simulations. A discharge measurement on Christiana Creek just upstream from the North Main Street well field during the May-June 1979 seepage run was used as a base to estimate the percentage loss of streamflow due to simulated pumpage. This discharge, measured at an 84-percent flow duration on the Elkhart River at Goshen, was 79.4 ft^3/s ± 5 percent. The following table summarizes the loss of streamflow of Christiana Creek for the three pumping experiments described previously.

Pumping experiment (Mgal/d)	Streamflow loss (ft^3/s)	Percentage loss from base flow
7.5	7.5	9
10.0	9.8	12
20.0	18.6	23

The simulated pumping experiments had a negligible effect on the flow of the St. Joseph River.

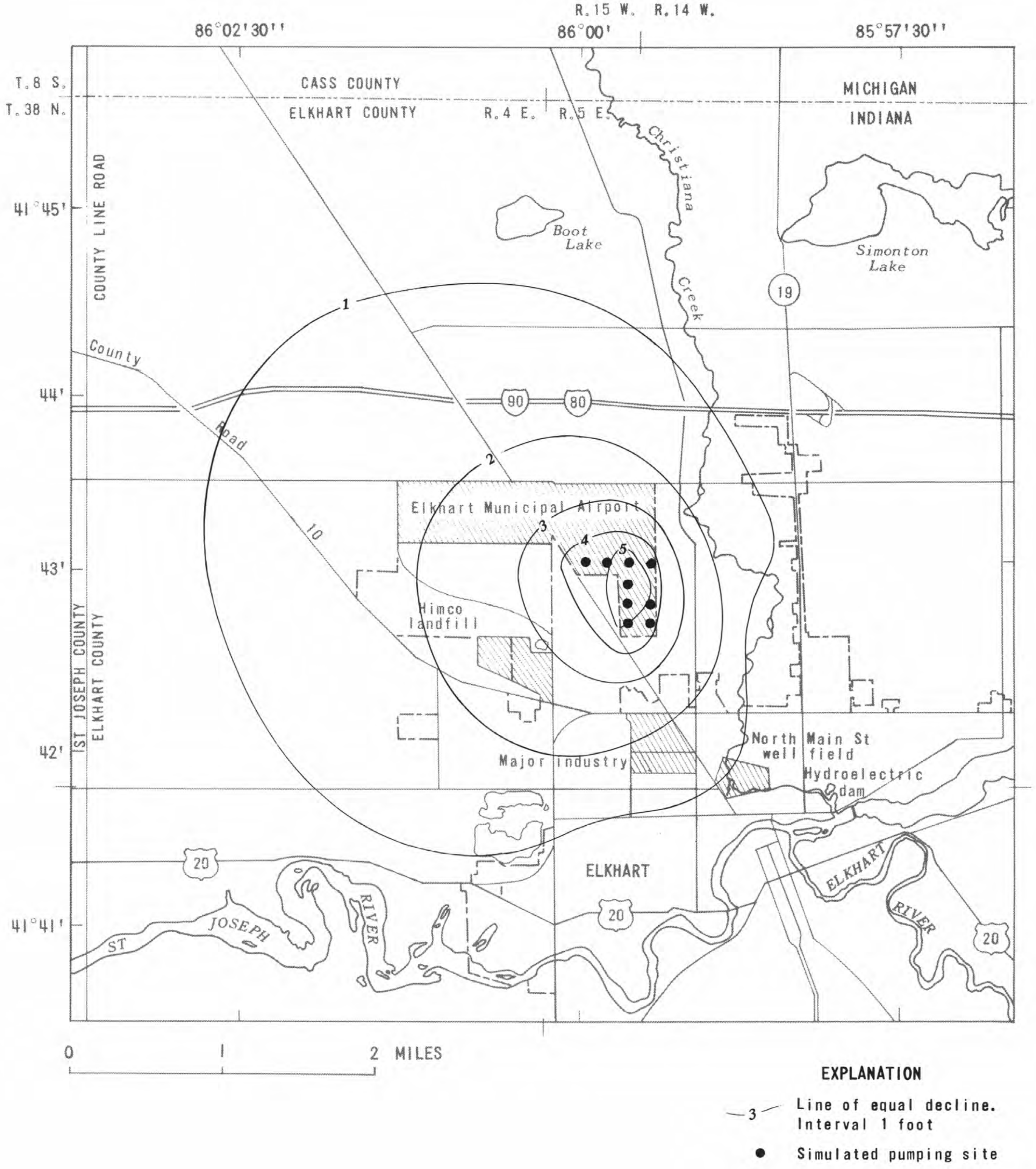


Figure 30.-- Model-simulated drawdowns in layer 1 due to 7.5-Mgal/day pumping at the airport.

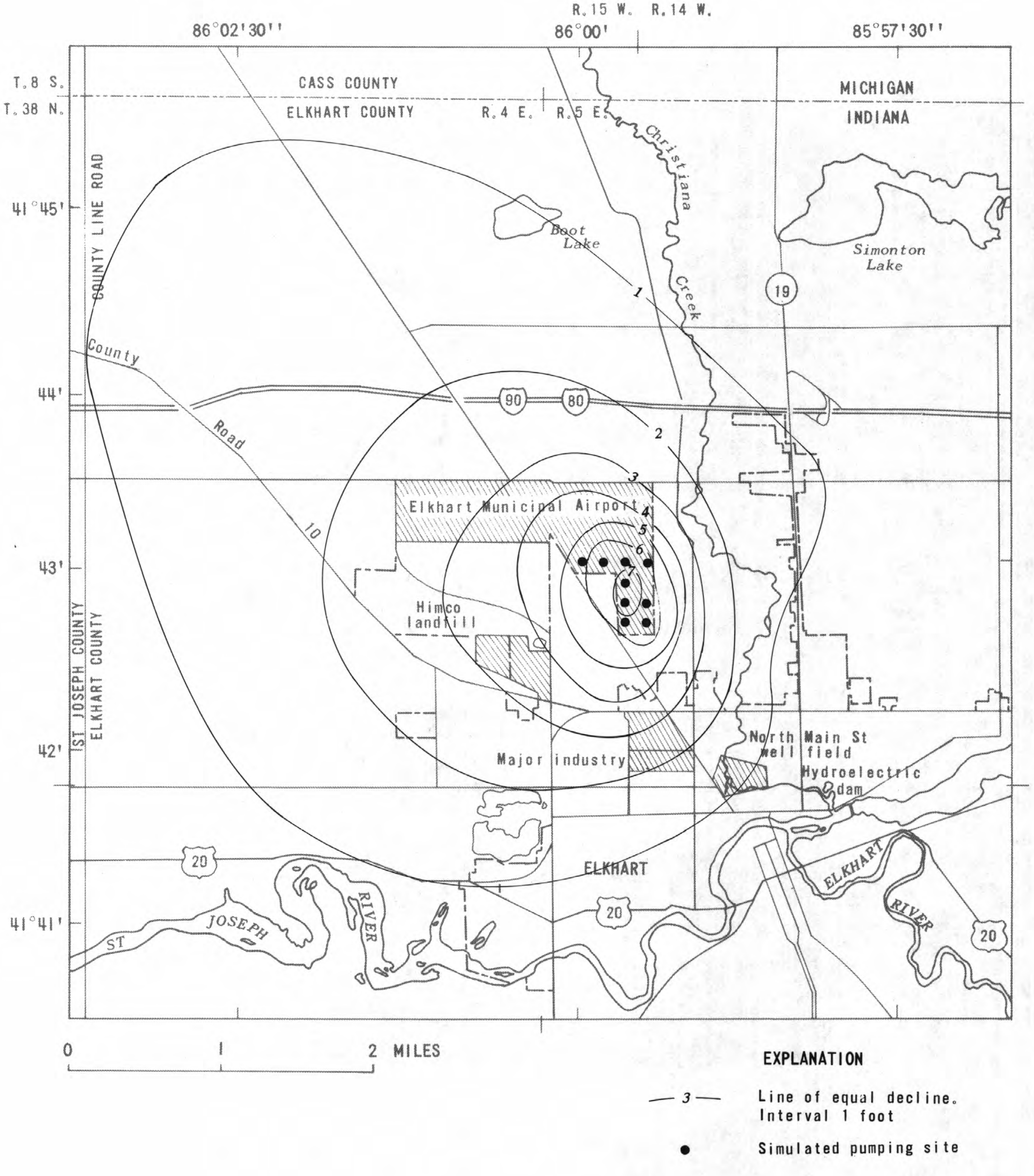


Figure 31.-- Model-simulated drawdowns in layer 1 due to 10-Mgal/day pumping at the airport.

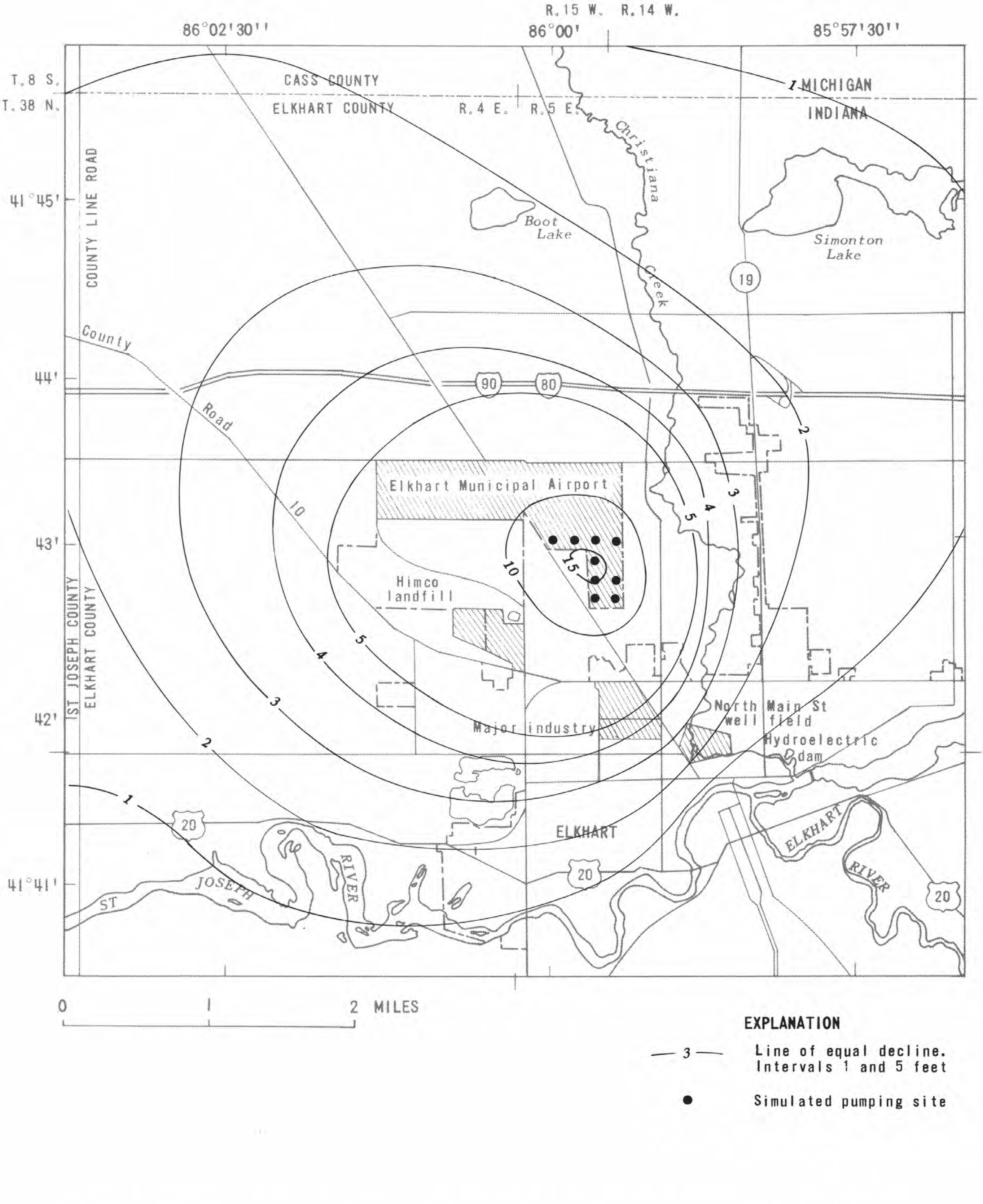


Figure 32.-- Model-simulated drawdowns in layer 1 due to 20-Mgal/day pumping at the airport.

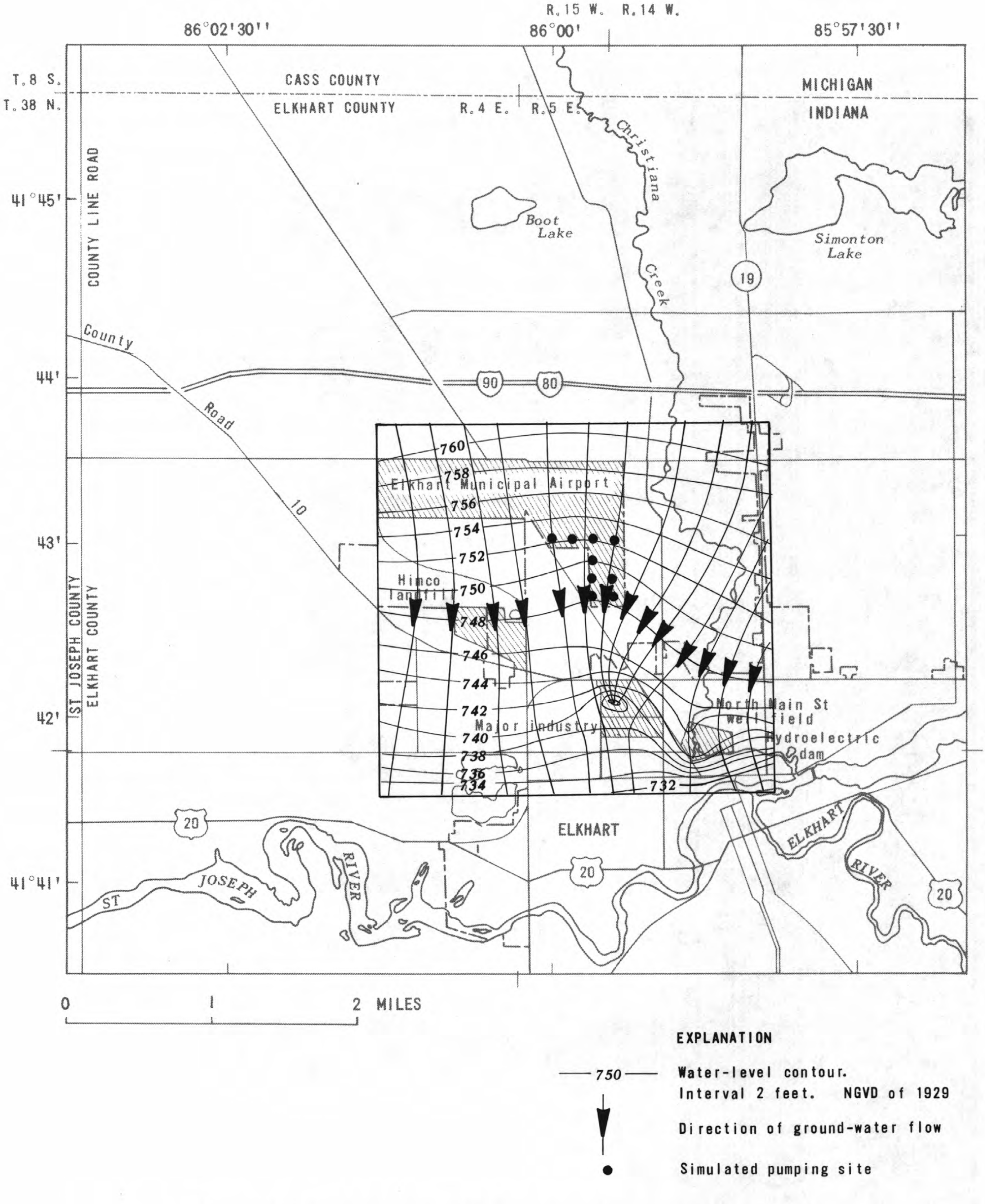


Figure 33.-- Steady-state, model-simulated water-level distribution in layer 1 due to 7.5-Mgal/day pumping at the airport.

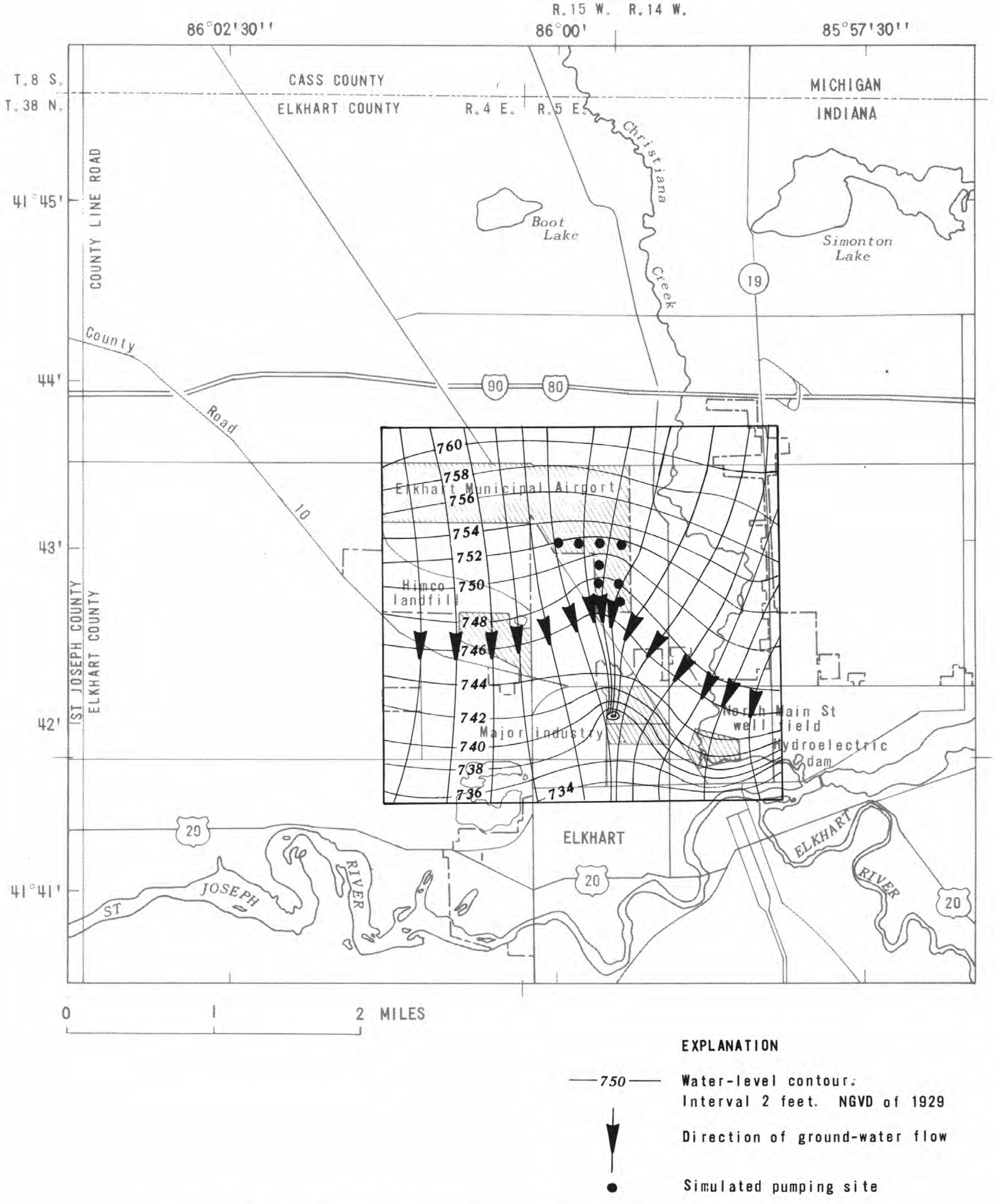


Figure 34.-- Steady-state, model-simulated water-level distribution in layer 1 due to 10-Mgal/day pumping at the airport.

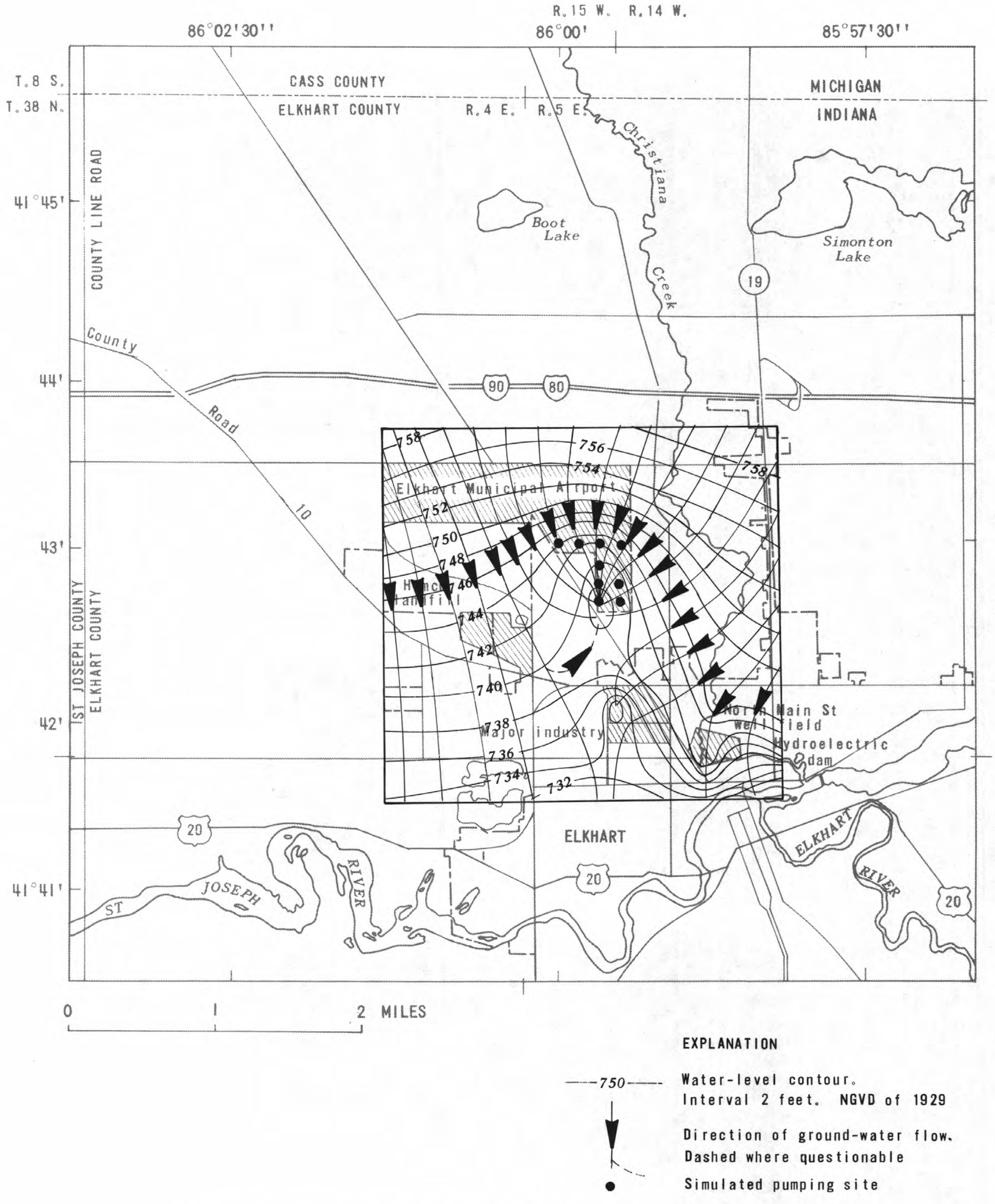


Figure 35.-- Steady-state, model-simulated water-level distribution in layer 1 due to due to 20-Mgal/day pumping at the airport.

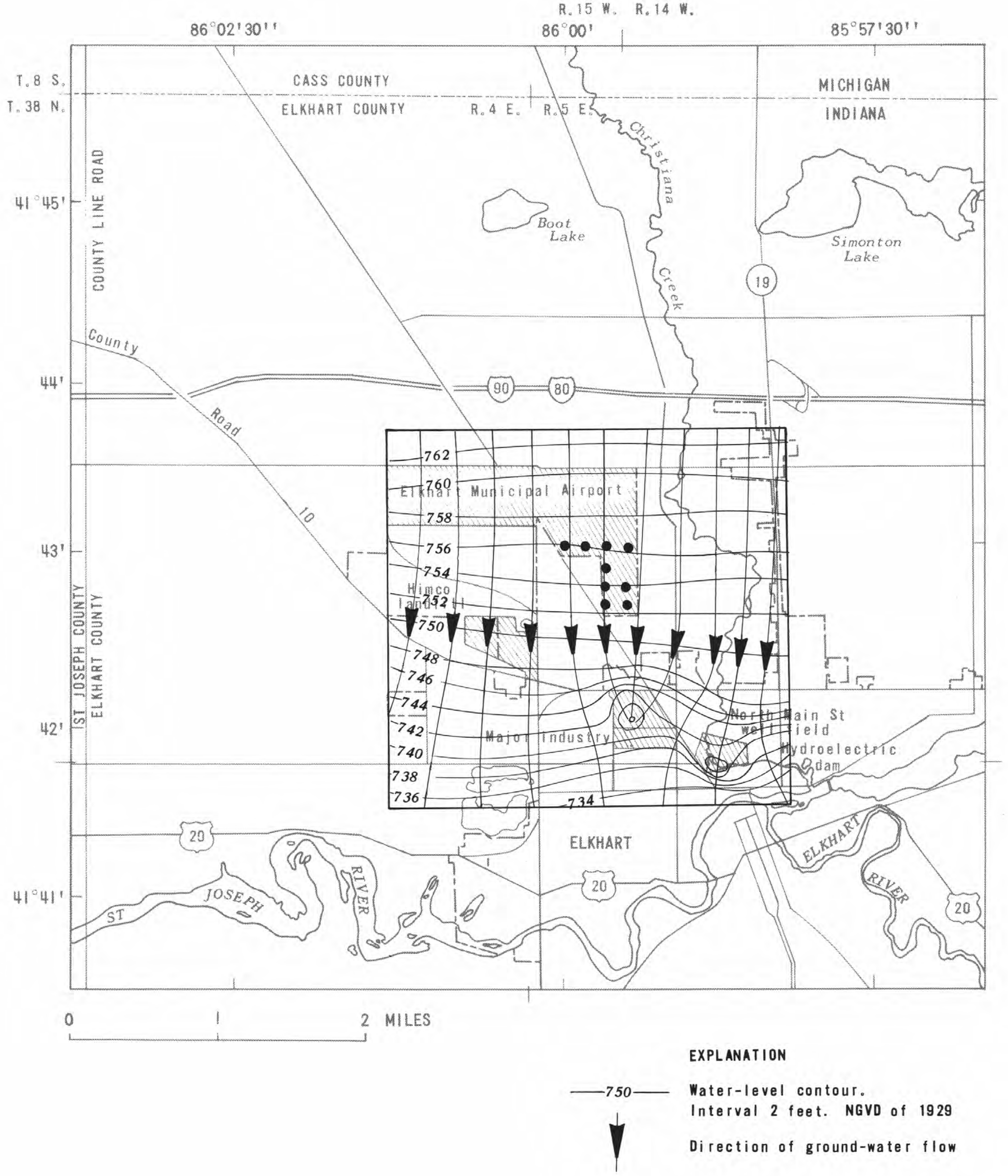


Figure 36.-- Prepumping, steady-state, model-simulated water-level distribution in layer 1.

Additional steady-state pumping experiments were done to determine the effect of the proposed 10-Mgal/d pumping at the airport if lower than calibrated values were used for the streambed leakance of Christiana Creek and the transmissivity of the aquifers. In one experiment, the streambed leakance of Christiana Creek was reduced to one-tenth of the calibrated value. In the second experiment, the transmissivities of both layers were reduced by the same factor. The resultant water-level distributions and associated flow patterns in the vicinity of the airport from the two experiments are shown in figures 37 and 38, respectively. The experiments show that little if any ground water from the landfill area would be drawn into the pumping wells at the airport even if either parameter is significantly lowered.

A transient simulation of 10-Mgal/d pumping at the airport was done to approximate the length of time required for the ground-water system to reach equilibrium. Figure 39 is a graph of drawdown plotted against time for a pumping node near the center of the simulated well field. Steady state was reached in approximately 600 days.

PART 2. WATER-QUALITY INVESTIGATION

In part 2 of the study, the general ground-water quality of northwest Elkhart County was evaluated, the areal extent of the effects of the Himco landfill and the industrial park on the local ground-water systems were determined, and the effect, if any, of the leachate from the landfill on the quality of water at the proposed well-field site was investigated.

APPROACH

Water samples were collected from observation wells throughout the study area for determining general ground-water quality and areal extent of the effects of the landfill and the industrial park on the local ground-water system. Field analyses included determinations of temperature, specific conductance, pH, dissolved oxygen, and redox potential. Laboratory analyses included determination of major constituents (calcium, magnesium, sodium, potassium, chloride, bicarbonate, sulfate, fluoride, silica, and bromide), selected trace elements (iron, manganese, aluminum, mercury, arsenic, selenium, lead, chromium, cadmium, barium, and boron), nutrients (ammonia, nitrate, nitrite, organic nitrogen, and orthophosphate), silica, cyanide, dissolved solids, dissolved organic carbon, and several specific groups of organic compounds (phenols, volatile organics, and base/neutral extractable

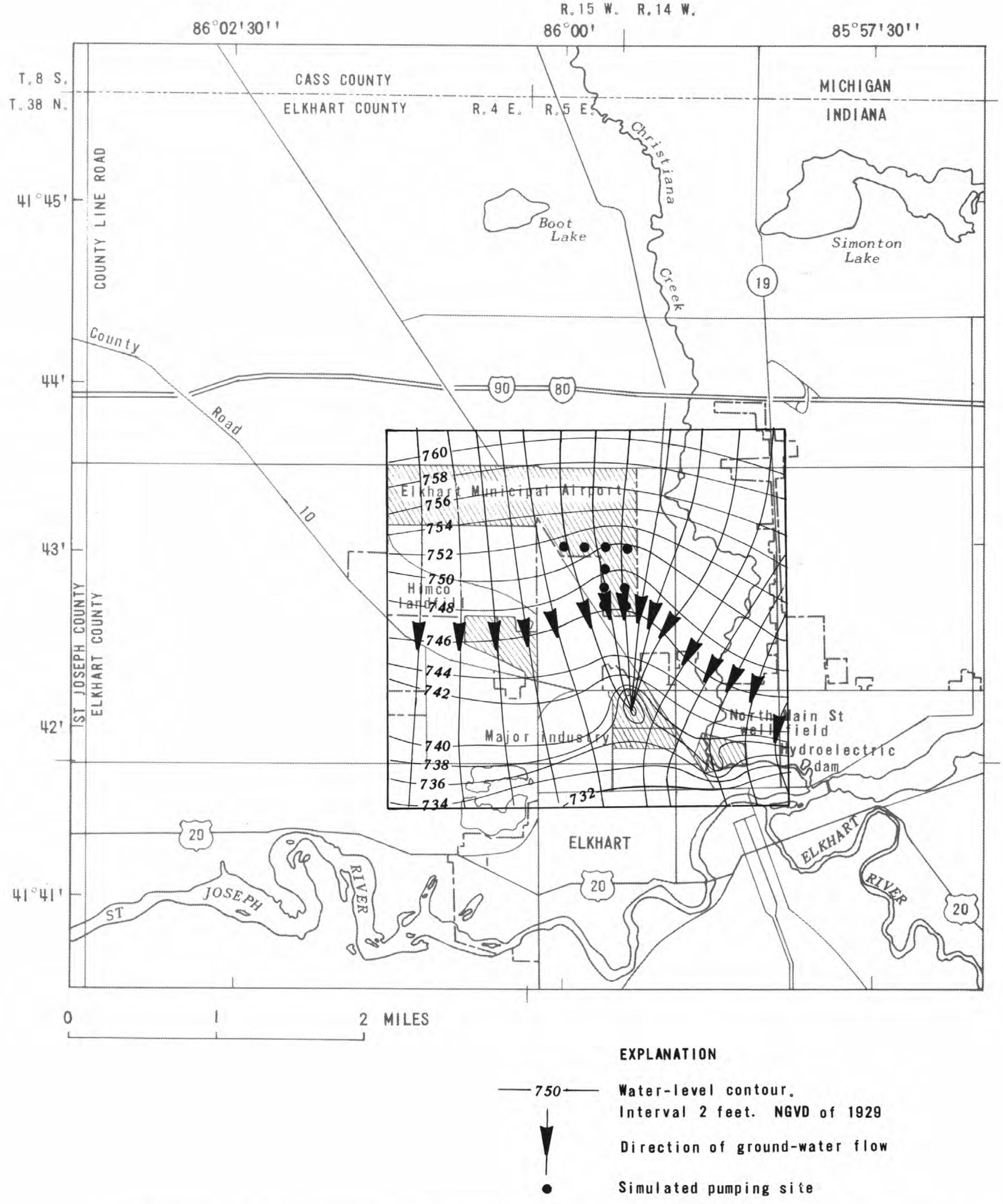
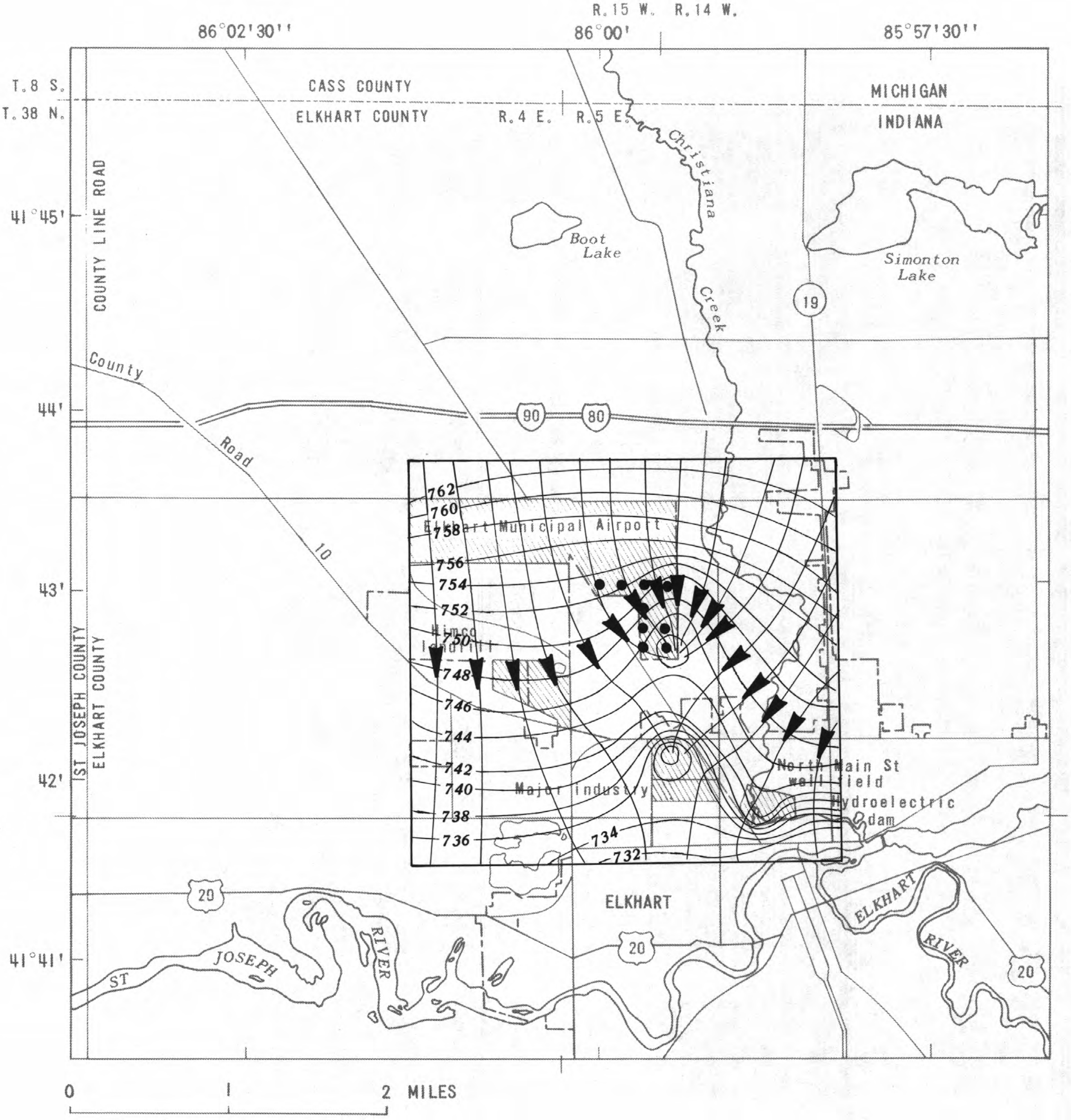


Figure 37.-- Model-simulated water-level distribution in layer 1 due to 10-Mgal/day pumping at the airport and a 90-percent decrease in streambed leakance of Christiana Creek.



EXPLANATION

Water-level contour. Interval 2 feet. NGVD of 1929

Direction of ground-water flow

Simulated pumping site

Figure 38.-- Model-simulated water-level distribution in layer 1 due to 10-Mgal/day pumping at the airport and a 90-percent decrease in transmissivity.

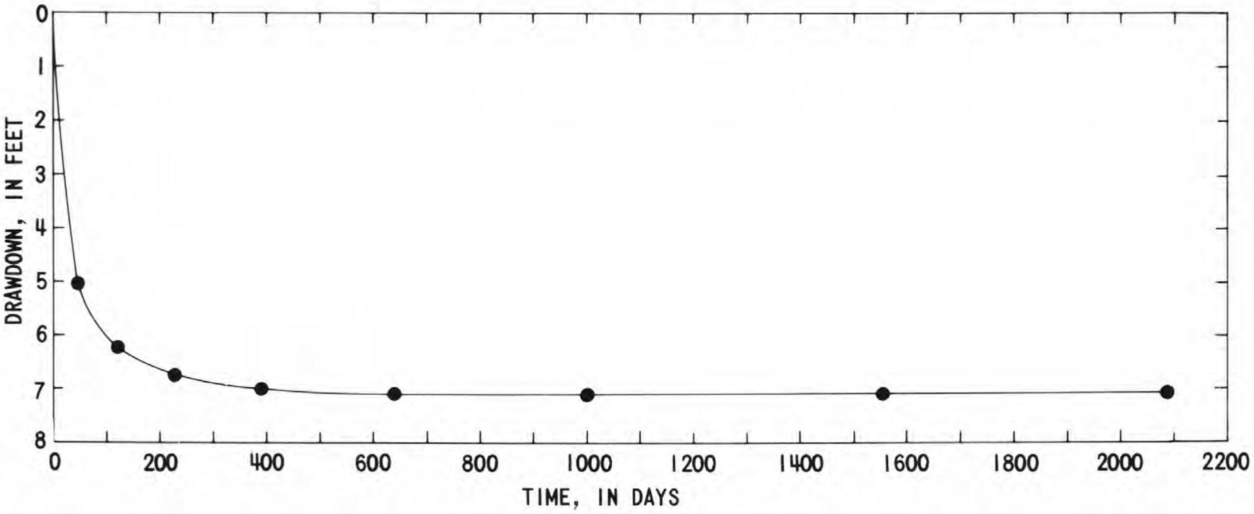


Figure 39.-- Relation of drawdown to time of pumping 10 Mgal/day at the airport.

organics[1]). The results were statistically summarized, and the most important factors influencing the results were determined by an analysis of variance. Constituent concentrations were compared with National Interim Primary and Secondary Drinking Water Regulations (U.S. Environmental Protection Agency, 1975, 1979a). Data were plotted areally and, where applicable, vertically, to explain the ground-water quality in the landfill and the industrial-park areas.

WATER-QUALITY METHODS

Sampling-Well Network

Water samples were collected from 104 of the 168 observation wells installed for the study. The sampled wells were divided into three groups: (1) county wells, (2) landfill wells, and (3) industrial-park wells.

---

[1] Extractable in methylene chloride from a strongly alkaline solution. See Keith and Telliard (1979, p. 416-423).

Wells in the first group, referred to as "county" wells in the remainder of this report, were principally outside the Elkhart city limits in Elkhart County. At many sites two wells were set, a shallow well (≤45 ft deep) in the unconfined aquifer and a deep well (≥110 ft deep) in the confined aquifer. In areas where the confining bed was absent, both shallow and deep wells were screened in the unconfined aquifer. Because most of these wells are in areas away from the influence of either the landfill or the industrial park, 35 county wells were sampled at 24 sites to obtain background concentrations for all water-quality parameters of interest. The site locations of all county wells sampled are shown in figure 40.

Forty-five wells were sampled at 17 sites containing from 1 to 5 wells, either upgradient or downgradient from the landfill (fig. 41), to define the areal extent of any leachate plume. These wells are called "landfill" wells in the remainder of the report. At the sites with more than one well, wells were screened at different depths for determining the vertical extent of the leachate. The screen depths are shown in table 4.

Table 4.--Screen depths of landfill wells sampled

Well designation	Screen depth (ft)	Well designation	Screen depth (ft)	Well designation	Screen depth (ft)
A1	133	E3	174	J1	40
A2	13	F1	32	J2	18
B1	495	F2	153	J3	152
B2	12	F3	15	K1	60
B3	129	F4	24	K2	15
B4	173	F5	195	K3	183
C1	341	G1	50	L1	60
C2	12	G2	16	L2	185
C3	195	G3	169	L4	17
C4	128	H1	18	M1	24
D1	19	H2	39	M2	103
D2	174	H4	105	N1	30
D3	88	I1	172	O1	30
E1	80	I2	15	P1	24
E2	17	I3	35	Q1	24

At the industrial park, 24 wells were sampled at 22 sites in the area south of the St. Joseph River and east of the Elkhart River (fig. 42). These wells are called "industrial-park wells" in the remainder of the report. All wells in this area were screened in the unconfined aquifer and were less than 45 ft deep.

Well-Numbering System

Wells in the three groups are identified in subsequent illustrations and tables as follows: County wells are identified by site numbers followed by "S" or "D" (shallow or deep); landfill wells are identified by site letters A-Q followed by single-digit numbers; industrial-park wells are identified by three-digit site numbers 101 to 122, except 109, which has three wells identified as 109A, 109B, and 109C.

Sampling Frequency

Samples representing the full range of hydrologic conditions were collected in March-May and September-October, periods of seasonally high and low water levels, respectively.

Twenty-eight of the county wells were sampled in June and July of 1978. On the basis of the data from these samples, 10 sites, with both shallow and deep wells, were selected for additional sampling in March-April and September of 1979.

Samples were collected from the landfill wells in April-May and October of 1978 and April-May and September of 1979. Sites A-F and I were sampled all four times. Sites G, H, J, K, and L were sampled only during 1978. Wells M-Q, installed in April 1979 to determine if a leachate plume were present, were sampled only the two times in 1979.

The industrial-park wells, installed in the summer of 1978, were sampled in September 1978 and in April and September 1979.

Field Sampling Procedures

A uniform procedure was used to sample all observation wells. The techniques used were primarily modifications of techniques recommended by Wood (1976). A summary of the field sampling procedure follows:

(1) Water levels were measured in all wells at each sampling site.

(2) The well to be sampled was then flushed by pumping the well with a a centrifugal pump.

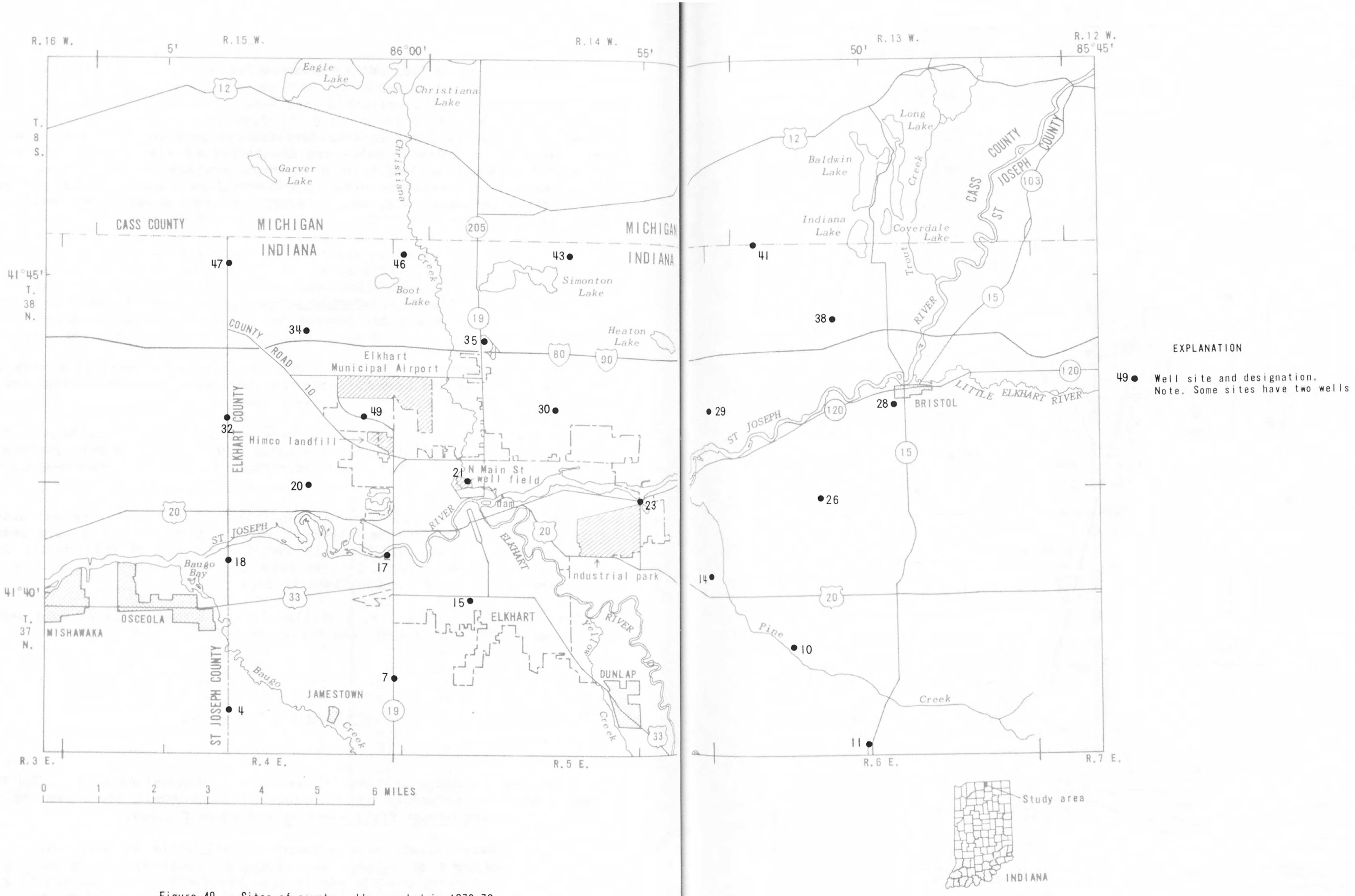


Figure 40.-- Sites of county wells sampled in 1978-79.

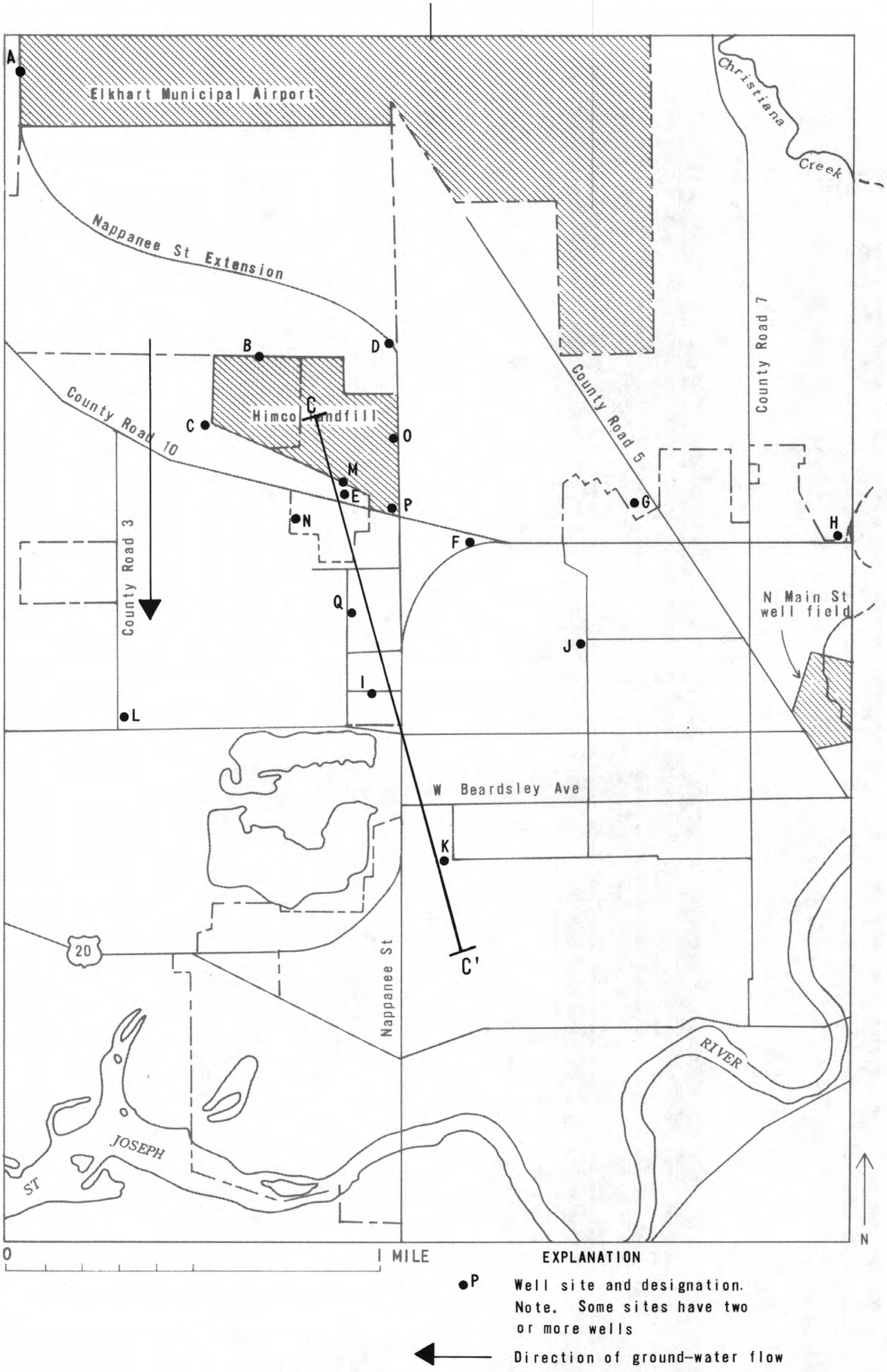


Figure 41.-- Sites of landfill wells sampled in 1978-79.

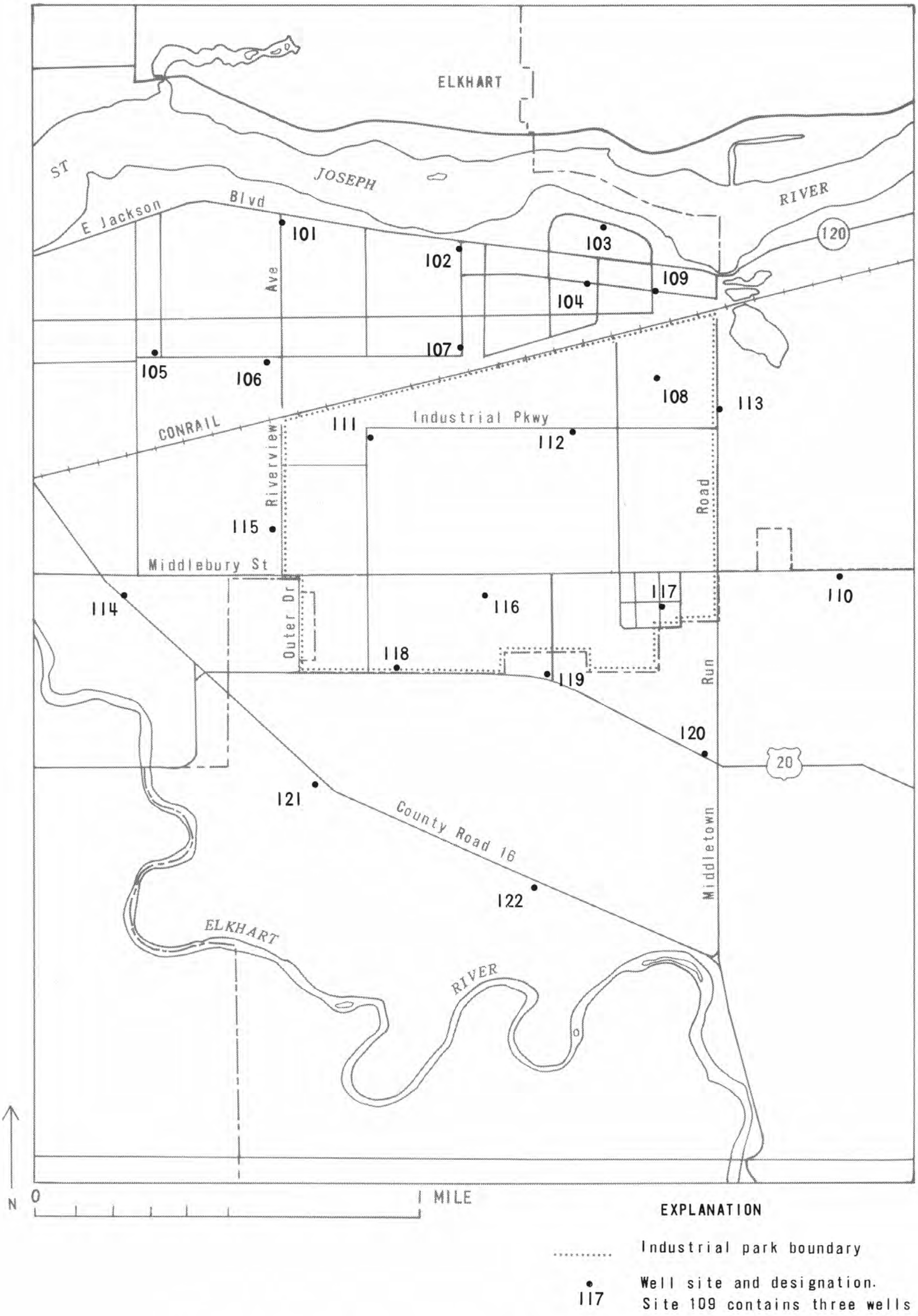


Figure 42.-- Sites of industrial-park wells sampled in 1978-79.

(3) The flushing rate of each well was measured.

(4) Properties and concentrations of constituents dissolved were measured in the flushing water with a Hydrolab 6-D multiprobe instrument. Readings were taken in 5-in. diameter wells by lowering the multiprobe sonde into the casing as close to the screen as possible. For 2-in. diameter wells, the sonde was placed in a flow chamber at the surface. Water was pumped into the inlet valve at the bottom and was allowed to flow up through the chamber, past the probes, and out. Values for field parameters determined by this apparatus agreed closely with those measured directly in the wells.

(5) Two criteria were used to determine the length of time a well would be flushed before sampling: A well was pumped until at least two casing volumes of water had been removed, or a well was pumped until the specific conductance readings measured with the Hydrolab had stabilized. These criteria were used to insure that water representative of conditions in the aquifer was sampled. The time necessary for the first criterion to be met was calculated by dividing an estimate of casing volume by the flushing rate. The second criterion was met by measuring specific conductance every 5 min until three successive readings differed by less than 10 µmho/cm at 25° C. Generally, stabilization of specific conductance required more time than removal of the two casing volumes of water.

(6) After a well was properly flushed, samples were collected with a peristaltic pump. This type of pump was used because it minimized contamination, aeration, and degassing of the sample as no pump parts ever contacted the pumped water and its discharge line was physically convenient for sampling and for connecting to the flow-chamber inlet. The intake for the peristaltic pump was always positioned at a greater depth than the centrifugal flushing line to insure that the water sampled was representative of conditions in the aquifer.

(7) Samples that did not need filtering were obtained directly from the discharge line of the peristaltic pump and were preserved by standard procedures (Skougstad and others, 1979). Samples for determining dissolved constituents were pressure filtered with nitrogen through a 0.45µ membrane, collected in the sample container, and preserved. Dissolved organic carbon samples were obtained by the technique described by Malcolm and Leenheer (1973).

(8) Samples were also collected for the field determination of dissolved oxygen by the azide modification of the Winkler titration and of alkalinity by an electrometric titration (Skougstad and others, 1979).

Laboratory Procedures

All samples were analyzed at the Geological Survey Laboratory in Doraville, Ga., by procedures in Skougstad and others (1979). Dissolved organic carbon was determined by the procedure outlined in Goerlitz and Brown (1972). Volatile and base/neutral extractable organic samples were analyzed by procedures described in U.S. Environmental Protection Agency (1979c).

Statistical and Computer Treatment of Data

Water-quality data were evaluated by a group of statistical computer programs called SAS or Statistical Analysis System (Helwig and Council, 1979). Because one of the basic assumptions of the use of parametric statistics is that the data are normally distributed, SAS was first used to test the normality of the data for each parameter. The data for most parameters were not normally distributed. Therefore, the distributions were made more normal by application of a natural logarithmic transformation of the form $y = \ln (x + 1)$. These transformed data were used in all subsequent calculations and presentations.

SAS was next used to calculate standard measures of central tendency (mean, median, and mode), dispersion (standard deviation, variance, and coefficient of variation), quartiles, and ranges for all parameters in selected groupings of the wells. Differences in water quality between groupings were hypothesized to result from the effects of four main factors: (1) season--autumn or spring; (2) site type--county, landfill, or industrial park; (3) aquifer depth--shallow ($\leq$45 ft) or deep (>45 ft); and site location--102 individual wells. Combinations of two or three of these factors--such as the interaction between site type and aquifer depth (shallow county, deep county, shallow industrial park, shallow landfill, or deep landfill--were also considered to be influencing factors. Factors that significantly affected the results were identified by an analysis of variance (ANOVA). The ANOVA is a statistical procedure that allows computation of the variances of each component factor, assesses the relative importance of these factors, and identifies significant differences between group means at a chosen confidence level (Zar, 1974). Preliminary calculations indicated that all two- and three-factor combinations, except the interaction between site type and aquifer depth, either would not account for a significant part of the total variance or would have been difficult to explain physically. For these reasons, the five factors (season, aquifer depth, site type, site location, and interaction between site type and aquifer depth) and a residual or error term to make up the total variance observed were used in the analysis of variance. An unbalanced nested-factorial design ANOVA was done on 25 of the parameters measured in this study. (See the general linear-models procedure in SAS, Helwig and Council, 1979.)

Though the ANOVA indicated significant differences between well-group means within a certain factor, it did not identify the particular means that differed. Duncan's Multiple Range Comparison Test (Duncan, 1955) was used by SAS to determine the significance of differences between the well-group means.

The geochemistry at several sites near the landfill was evaluated by WATEQF, a chemical equilibrium computer model by Plummer and others (1976). This model uses equilibrium thermodynamic data and dissolved-constituent concentrations to calculate the theoretical saturation conditions for more than 190 minerals. These calculations indicate which minerals are thermodynamically supersaturated or undersaturated and, therefore, are possibly precipitating or dissolving in the sampled ground-water system.

QUALITY-CONTROL RESULTS AND DISCUSSION

Precision and Accuracy

Several quality-control measures were used to test the variability introduced into the water-quality results by sampling, shipment, and analysis.

Triplicate sampling of a landfill well demonstrated that the results of nearly all parameters were reproduced within 10 percent. DOC (dissolved organic carbon), with a coefficient of variation of approximately 100 percent, was the principal exception. This variation may have been introduced by use of vinyl tubing as a sampling line or contamination of the laboratory equipment used to obtain the sample.

Differences between field and laboratory measurements of pH, specific conductance, and alkalinity were generally less than 5 percent. These differences indicate that chemical changes between the time of collection and analysis were minimal.

Analyses of a three-way split of one sample showed analytical variabilities of less than 10 percent for each parameter determined.

For most samples, the cation-anion balance agreed within ± 5 percent.

These quality-control findings indicate that the sampling and the analytical techniques used were effective for obtaining precise and accurate results for all parameters except dissolved organic carbon. Because the analytical variation of DOC was minimal, some type of sampling error was apparently responsible for the observed variations of this parameter.

Comparison of Well-Casing Materials

PVC (polyvinyl chloride) casing has been cited as the preferred casing material in most studies of ground-water quality, owing to its chemical inertness (U.S. Environmental Protection Agency, 1977, p. 146; and Hughes and others, 1974). PVC is resistant to chemical degradation or dissolution by all common chemicals except ketones, esters, and aromatics. The principal limitation to its use is its lack of physical strength and susceptibility to damage.

Metallic casing materials may be subject to chemical degradation or dissolution under reducing or acidic conditions. Dissolution could introduce iron, manganese, and other trace metals into the well water. However, metallic casings are still desirable because of their ease of installation, physical strength, and resistance to damage.

Several types of well casing were used in the study: PVC, low-carbon steel, galvanized steel, and stainless steel. An assumption made at the beginning of the study was that, if a well casing was properly flushed before sampling, the casing material would have very little affect on the quality of water in the wells. Additional samples were collected at landfill site F and at industrial-park site 109 to test this assumption.

Chemical analyses of water from three wells at site F (wells F1, F3, and F4), screened in the same shallow aquifer, are presented in table 5. Although the pH was not acidic, the ground-water environment was a reducing one (absence of dissolved oxygen and redox potentials <200 mv). Iron and manganese concentrations of water from the low-carbon steel well were in between those of water in the two PVC wells. These results are probably due to the increase in reducing conditions with depth, indicated by decreasing redox potentials, rather than a result of casing dissolution. This conclusion is supported by the virtually equal dissolved-solids concentrations of water from the three wells. Furthermore, there was practically no difference in the concentrations of specific trace elements in water from the three wells.

At site 109, three wells were installed within 30 ft of each other. The wells were screened at the same depth, but each was constructed of different materials. Well 109A was constructed with galvanized-steel casing and screen, 109B with low-carbon steel casing and screen, and 109C with stainless-steel casing and screen. All wells were pumped identically and were sampled three times during the study. A simple analysis of variance comparing the chemical analyses of water from these three wells was done on all

Table 5.--Chemical analyses of samples collected in September 1979 from three landfill wells constructed of different casing materials

Parameter	Units	Wells at site F and casing materials F3 PVC	 F4 low-carbon steel	 F1 PVC
Well depth	ft	15	24	32
pH	--	7.4	7.7	7.7
Redox potential	mv	+212	+109	+41
Oxygen, diss.	mg/L	<.05	<.05	<.05
Solids, diss.	mg/L	422	421	419
Iron, diss.	µg/L	20	90	630
Manganese, diss.	µg/L	20	60	70
Cadmium, diss.	µg/L	<.5	<.5	<.5
Lead, diss.	µg/L	<.5	1	2
Arsenic, diss.	µg/L	<.5	<.5	3
Chromium, diss.	µg/L	<5	<5	<5
Barium, diss.	µg/L	200	<50	200

parameters. The ANOVA demonstrated that, for the three wells, measurements of iron, manganese, pH, redox potential, and dissolved-solids values were not significantly different at the 95-percent confidence level. The only parameter whose measurements differed significantly was specific conductance, which was highest in the low carbon steel well. However, the explanation for this high value is not increases in iron, manganese, or other dissolved trace metals but, rather, increases in chloride and magnesium. These findings at the site-109 wells support the assumption that the well-casing material has little affect on water-quality results when the well has been properly flushed before sampling.

WATER-QUALITY RESULTS AND DISCUSSION

Analysis of Variance Results

The analysis of variance described in the section "Statistical and Computer Treatment of Data" was used to test the significance of differences between means of certain groups of water-quality data. This statistical procedure was used on 25 parameters determined in the study. The results of the analysis of variance are given in table 6. The values reported in the table are probabilities that differences in the values of a parameter caused by a particular factor are not due to chance alone. For example, the differences in water-temperature data due to sampling season were statistically significant at the 99-percent level; that is, there is a 99-percent probability that unequal mean temperatures result from different sampling seasons and not because of chance alone. Differences significant at less than 80-percent level were not reported.

Well-site location was the factor having a significant effect on the largest number of water-quality parameters (20). This result was no surprise because sampling wells were placed upgradient and downgradient from the landfill specifically for observing differences in water quality. The most important differences noted among the 102 wells evaluated are discussed in the sections "County Wells," "Industrial-Park Wells," and "Landfill Wells" that immediately follow this section.

Though the factors site type and aquifer depth affected only a small number of parameters separately, the interaction of these two factors significantly affected the second largest number of parameters. Differences in water-quality results among the five groups of wells making up this factor (shallow county, deep county, shallow landfill, deep landfill, and shallow industrial park) were significant at the 80-percent or greater level for 14 of the 25 parameters. The results for these five groups are summarized in the sections mentioned in the preceding paragraph.

The Duncan's multiple range test, used to determine significant differences between the group means, pointed out an important general finding in the results of the interaction between site type and aquifer depth. Concentrations of several parameters in samples from the shallow landfill wells were significantly higher than concentrations of those parameters in samples from the other well groups. This pattern for dissolved solids is illustrated in figure 43. Similar patterns were also observed for potassium, sodium, calcium, specific conductance, hardness, ammonia, chloride, sulfate, and bromide. The leachate being produced by the landfill is probably the reason that the concentrations of these parameters were higher in the shallow aquifer near the landfill than elsewhere. The differences between water quality in the shallow landfill wells and the other four groups of wells are discussed in succeeding sections.

Table 6.--Probabilities for five factors affecting water-quality parameters of county, landfill, and industrial-park wells, calculated by analyses of variance

[All values are percentages; --, significant at <80-percent level]

Parameters	Factors				
	Site type	Aquifer depth	Interaction of site type with aquifer depth	Well-site location	Season
Water temperature	99	99	--	--	99
pH	--	--	--	99	99
Alkalinity	--	--	95	99	--
Hardness	--	--	90	99	--
Specific conductance	--	80	95	99	--
Dissolved solids	--	80	95	99	--
Dissolved oxygen	95	--	--	99	--
Redox potential	98	95	80	--	--
Silica	--	90	--	99	90
Chloride	--	80	--	99	--
Sulfate	98	--	--	99	--
Bromide	90	--	--	99	--
Calcium	--	--	80	99	--
Magnesium	--	--	80	99	--
Sodium	--	--	99	99	--
Potassium	--	95	98	99	--
Iron	80	--	95	98	90
Manganese	--	--	98	99	--
Aluminum	--	--	--	--	99
Arsenic	--	80	--	99	98
Boron	95	--	95	--	--
Lead	--	--	--	--	90
Dissolved organic carbon	--	--	--	99	--
Ammonia	--	80	80	99	--
Nitrate	80	--	80	95	--

Sampling season significantly affected the fewest parameters (7 of 25) of any factor considered in the analysis of variance. In addition, the significant differences detected for 6 of the 7 constituents affected were within the sampling or analytical error of these parameters. These types of errors were sources of variation not separated out in the ANOVA. Thus, temperature

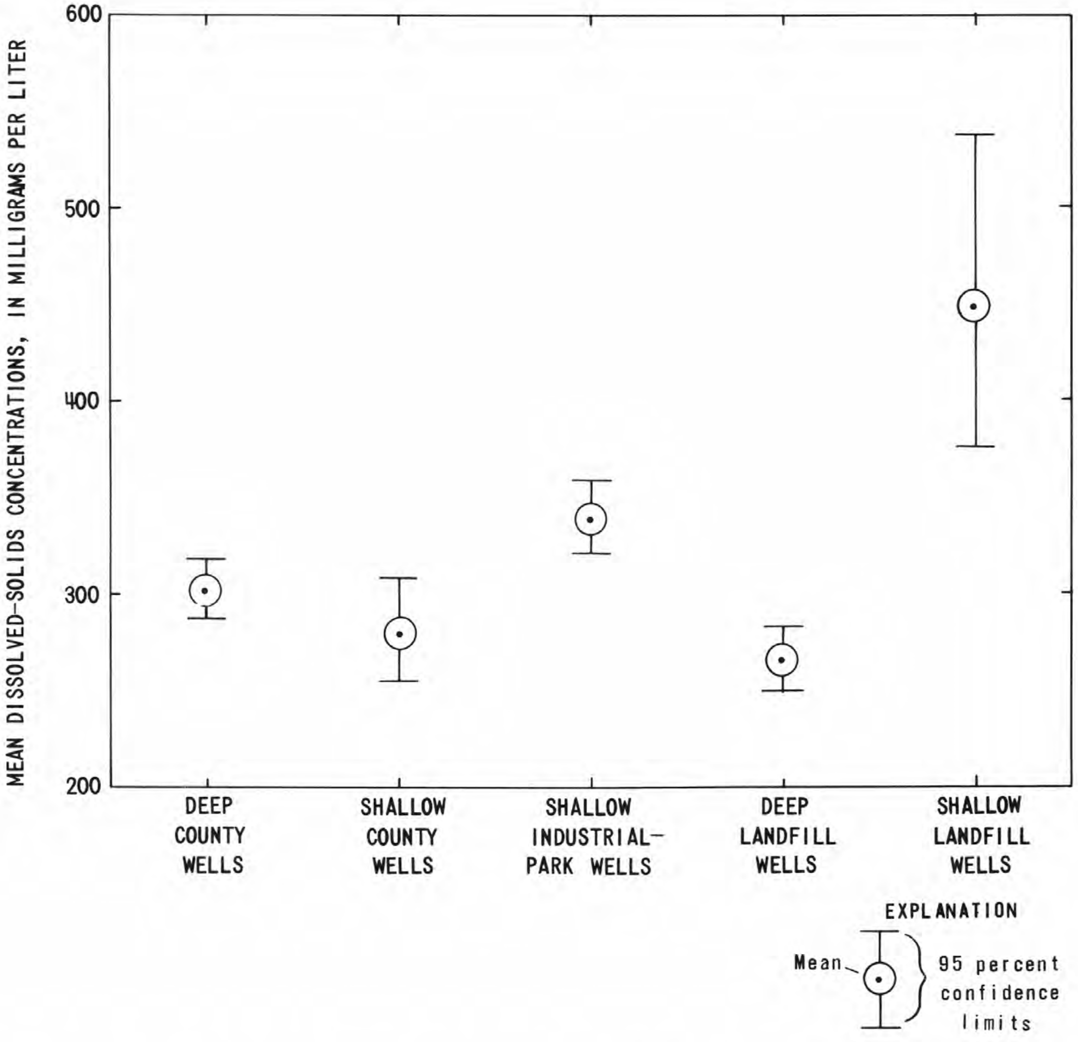


Figure 43.-- Variation of mean dissolved-solids concentrations for five groups of wells.

was probably the only parameter truly affected by sampling season. The temperature of the water in the shallow aquifer paralleled the ambient air temperature. The temperature of samples collected in autumn (September and October) was consistently higher than the temperature of samples collected in the spring (March and April). Possible explanations why other water-quality parameters were not affected by sampling season are (1) seasonal differences in concentration of chemical species are small because there is little hydrologic variation seasonally in the volume of the ground-water reservoir, and (2) the differences in temperature (<2° C) are not great enough to alter chemical equilibria in the ground-water system.

County Wells

Most county wells are in areas away from the probable influence of the landfill and the industrial park. For this reason, the water-quality results obtained from these wells were assumed to represent the background concentrations for the chemical constituents of ground water in northwest Elkhart County.

A statistical summary of the water-quality analyses for the shallow (≤ 45 ft) county wells is presented in table 7, and a similar summary for the deep county wells is presented in table 8. The means presented for all parameters except redox potential, are the antilogs of the natural-log transformed means, which approximate the geometric means. The coefficients of variation presented differ widely from parameter to parameter. Most of the high coefficients of variation can be attributed to parameter values being at or below the detection limit of the analytical procedure used.

Results for the two sets of county wells include several general similarities. The two sets have slightly basic pH (medians of 7.7) and very high hardness (averages of 210 and 240 mg/L as $CaCO_3$). High alkalinities (averages of 150 and 250 mg/L as $CaCO_3$) indicated that both waters were well buffered by carbonate equilibria. Dissolved-solids concentrations for the two groups averaged 280 and 302 mg/L. The water type of more than 90 percent of the county well samples analyzed was calcium bicarbonate. These basic water-quality characteristics are typical of ground water in glacial deposits in the midwestern United States (Freeze and Cherry, 1979, p. 284).

The principal difference between the shallow and the deep county wells was in their redox environments. The water in the deep county wells had significantly lower average dissolved-oxygen concentrations (0.4 mg/L) and redox potentials (+62 mv) than that in the shallow county wells (2.4 mg/L and +246 mv), respectively. Reducing conditions were present in most deep wells, whereas oxidizing conditions were present in most shallow wells. Reducing conditions may be caused by microbiological processes and chemical reactions that deplete the oxygen in ground water where organic material or other oxygen-consuming substances are present. If the aquifer is in contact with the atmosphere, as a shallow unconfined aquifer is, the dissolved oxygen can be replenished by gaseous exchange and diffusion. However, deeper in the unconfined aquifer or in a confined aquifer, dissolved oxygen that is consumed is not readily replaced. Furthermore, the capacity of the reducing system may be greater than the rate of reaeration by diffusion. For several water-quality parameters, the variation in concentration between the deep and the shallow county wells was probably caused by the dissimilarity in the redox potentials.

Mean nitrate and sulfate concentrations were significantly lower in the deep county wells than in the shallow--0.17 and 4 mg/L in the deep wells and 2.3 and 27 mg/L in the shallow wells, respectively. The lower concentrations in the deep wells may have been due to nitrate and sulfate acting as oxidizing agents after the dissolved oxygen was depleted. Also, the shallow aquifer may have received nitrate-enriched water by infiltration from fertilized fields and residential lawns.

Dissolved-solids concentrations and specific-conductance values were higher in the deep wells than in the shallow, owing to higher concentrations of iron, manganese, bicarbonate, silica, magnesium, and sodium in the deep wells. Iron and manganese concentrations were significantly higher in the deep wells (0.71 and 0.10 mg/L) than in the shallow (0.03 and 0.01 mg/L), respectively, probably because iron and manganese compounds are more soluble under reducing conditions. Bicarbonate, the principal contributor to the alkalinity, was also probably solubilized, owing to reducing conditions, as it can be influenced by biologically mediated redox reactions. Concentrations of silica, magnesium, and sodium were also significantly higher in water from the deep wells, presumably because of longer contact time of water with the aquifer minerals.

In aqueous solution, trace elements should behave similarly to iron and manganese for two reasons: (1) Several trace elements are redox-affected parameters, as are iron and manganese, and (2) their concentrations have been shown to correlate closely with concentrations of these two elements in most systems, owing to adsorption and coprecipitation with iron and manganese hydrous oxides (Jenne, 1968). However, concentrations of the trace elements determined in this study (aluminum, cadmium, chromium, mercury, lead, arsenic, boron, selenium and barium) were uniformly low (at or near detection limits for several) in both shallow and deep county wells.

Mean DOC concentrations were higher in the shallow wells (4.4 mg/L) than in the deep (3.2 mg/L). The significance of this difference is questionable because of the large variability in DOC concentrations in replicate samples discussed in the section "Quality-Control Results and Discussion." Leenheer and others (1974) analyzed 100 samples from wells and springs in 27 States to obtain background concentrations of DOC. The median concentration of samples from sand-and-gravel aquifers was 0.7 mg/L. The range of concentration was from 0.1 to 15 mg/L, and 85 percent of the concentrations were less than 2 mg/L. Leenheer and others (1974) recommended 5 mg/L as a DOC threshold concentration, which, if exceeded, would indicate organic ground-water contamination. Although dissolved-organic carbon concentrations for the county wells were higher than the background median concentration for sand-and-gravel aquifers, they were less than 5 mg/L.

The National Interim Primary Drinking Water Regulations (NIPDWR) of the U.S. Environmental Protection Agency (1975) established the maximum concentrations for certain inorganic chemicals in community-treated water systems. These concentrations (table 9) are estimates of the concentrations of toxic constituents that might adversely affect public heatlh.

Concentrations of 9 of the 10 inorganic constituents listed in table 9 were determined in the study. Silver was not determined because it is rarely detected, except in areas of silver mineralization or areas receiving photographic wastes. Concentrations of silver determined by the Indiana State Board of Health on untreated water from the three Elkhart city wells were

Table 7.--Statistical summary of water-quality analyses for shallow county wells, 1978-79

[All units of measure in milligrams per liter except pH or as indicated; ND, not determined]

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Temperature (°C)	44	13.0	5	19.5	9.5
pH	44	[1]7.7	2	8.5	7.1
Specific conductance (μmho/cm at 25° C)	44	484	5	1,170	289
Oxygen, diss.	44	2.4	62	9.5	.1
Redox potential (mv)	44	+246	55	+480	+37
Hardness, tot. (as $CaCO_3$)	44	210	5	370	120
Dissolved solids[2]	44	280	6	626	168
Alkalinity, tot. (as $CaCO_3$)	44	150	8	340	69
Chloride, diss.	44	17	32	210	4
Sulfate, diss.	44	27	15	250	11
Fluoride, diss.	24	.1	92	.1	<.05
Bromide, diss.	10	.1	53	.1	<.05
Silica, diss.	44	11	7	15	7
Calcium, diss.	44	57	7	100	32
Magnesium, diss.	44	16	9	30	10
Sodium, diss.	44	8	42	170	2
Potassium, diss.	44	1.1	37	4.7	.3
Cyanide, tot.	10	<.005	ND	<.005	<.005
Iron, diss.	44	.03	75	.03	<.005

Table 7.--Statistical summary of water-quality analyses for shallow county wells, 1978-79--Continued

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Manganese, diss.	44	0.01	97	0.19	<0.005
Aluminum, diss.	44	<.005	79	.04	<.005
Arsenic, diss.	20	.002	58	.007	<.0005
Boron, diss.	20	.01	65	.28	<.005
Barium, diss.	20	<.05	205	.1	<.05
Cadmium, diss.	20	<.0005	244	.001	<.0005
Chromium, diss.	20	<.005	91	.02	<.005
Mercury, diss.	10	<.0005	ND	<.0005	<.0005
Lead, diss.	20	.001	102	.006	<.0005
Selenium, diss.	10	<.0005	300	.001	<.0005
Phenols, tot.	10	<.0005	ND	<.0005	<.0005
Organic carbon, diss.	44	4.4	39	24	.3
Ammonia as N, diss.	44	.02	251	.32	<.005
Nitrate, as N, diss.	44	2.3	76	15	<.005
Nitrite as N, diss.	44	.01	395	.42	<.005
Organic nitrogen, as N, diss.	44	.08	132	.78	<.005
Orthophosphate as P, diss.	44	.01	148	.05	<.005

---

[1] Median value.

[2] Residue on evaporation at 180° C.

Table 8.--Statistical summary of water-quality analyses for deep county wells, 1978-79

[All units of measure in milligram per liter except pH or as indicated; ND, not determined]

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Temperature(°C)	24	13.0	5	19.0	10.5
pH	24	[1]7.7	1	8.2	7.3
Specific conductance (µmho/cm at 25° C)	24	548	2	718	448
Dissolved oxygen	24	.4	40	.7	.1
Redox potential (mv)	24	+62	59	+125	+8
Hardness, tot. (as $CaCO_3$)	23	240	2	290	180
Dissolved solids[2]	24	302	2	403	238
Alkalinity, tot. (as $CaCO_3$)	24	250	1	280	210
Chloride, diss.	24	15	34	60	2
Sulfate, diss.	24	4	64	40	<.05
Fluoride, diss.	4	.2	38	.2	.1
Bromide, diss.	10	.1	50	.2	<.05
Silica, diss.	24	15	3	17	12
Calcium, diss.	23	60	2	68	48
Magnesium. diss.	23	21	7	29	12
Sodium, diss.	23	18	28	70	6
Potassium, diss.	23	1.3	28	3.3	.7
Cyanide, tot.	9	<.005	ND	<.005	<.005
Iron, diss.	23	.71	.17	2.10	.01

Table 8.--Statistical summary of water-quality analyses for deep county wells, 1978-79--Continued

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Manganese, diss.	23	0.10	16	0.28	0.01
Aluminum, diss.	23	<.005	138	.20	<.005
Arsenic, diss.	19	.003	53	.014	<.0005
Boron, diss.	20	.02	42	.12	<.005
Barium, diss.	19	.1	36	.2	<.05
Cadmium, diss.	19	<.0005	308	.002	<.0005
Chromium, diss.	19	<.005	88	.01	<.005
Mercury, diss.	10	<.0005	ND	<.0005	<.0005
Lead, diss.	19	.001	87	.007	<.0005
Selenium, diss.	10	<.0005	ND	<.0005	<.0005
Phenols, tot.	10	<.0005	ND	<.0005	<.0005
Organic carbon, diss.	24	3.2	33	11	.6
Ammonia as N, diss.	24	.12	70	.31	<.005
Nitrate as N, diss.	24	.17	231	2.2	<.005
Nitrite as N, diss.	24	.01	228	.01	<.005
Organic nitrogen, as N, diss.	24	.09	117	.46	<.005
Orthophosphate as P, diss.	24	.02	201	.17	<.005

---

[1] Median value.

[2] Residue on evaporation at 180° C.

Table 9.--Drinking-water standards for selected inorganic chemicals

[Source, U.S. Environmental Protection Agency (1975 and 1979a), primary and secondary regulations, respectively]

National Interim Primary Drinking Water Regulations		National Secondary Drinking Water Regulations	
Parameter	Concentration (mg/L)	Parameter	Concentration (mg/L)
Arsenic	0.05	Chloride	250
Barium	1.0	Copper	1
Cadmium	.010	Iron	.3
Chromium	.05	Manganese	.05
Fluoride	[1]1.4-2.4	pH	6.5-8.5
Lead	.05	Sulfate	250
Mercury	.002	Dissolved solids	500
Nitrate	10	Zinc	5
Selenium	.01		
Silver	.05		

[1] Concentration varies with temperature.

less than 0.010 mg/L (Sam Lyle, Elkhart City Water Works, written commun., 1980). Of the remaining nine constituents, only nitrate exceeded its NIPDWR limit. Nitrate concentration (as N) exceeded the 10-mg/L limit in wells 38S and 46S, 15 and 11 mg/L, respectively, the only time each was sampled. The probable cause for both of these high nitrate values was the infiltration of water into the shallow unconfined aquifer from fertilized farm fields in the immediate vicinities of these wells.

The NSDWR, or National Secondary Drinking Water Regulations (U.S. Environmental Protection Agency, 1979a), recommends maximum limits of inorganic chemicals and water properties that might affect the esthetic quality, taste, or odor of drinking water. These limits are also given in table 9. Copper and zinc were not determined because, in preliminary analyses, concentrations of the two metals were less than or equal to 0.010 mg/L. Concentrations of all other parameters included in the NSDWR were determined.

Dissolved-solids concentration exceeded the 500-mg/L limit in water from wells 35S, 47S, and 49S the only time that they were sampled. In wells 35S and 47S, sodium and chloride concentrations were responsible for the high dissolved-solids concentration, whereas, in water from well 49S, calcium and sulfate were responsible for the high concentration. The high sodium and chloride concentrations may have been caused by infiltration of roadside runoff containing deicing salts.

Iron concentrations exceeded the 0.3-mg/L NSDWR limit in 23 percent of the shallow county well analyses and in 78 percent of the deep county well analyses. Manganese concentrations exceeded the 0.05-mg/L limit in 32 percent and 87 percent of the shallow county and deep county well analyses, respectively. Two possible explanations for the frequency of these high values are: (1) The area has a natural abundance of the two elements in the aquifer materials, and (2) the reducing conditions at depth in the aquifers cause dissolution of minerals and coatings containing iron and manganese or permit these elements to remain in solution.

Industrial-Park Wells

Because all complaints about ground-water quality in the industrial-park area involved shallow wells in the adjacent residential area, all water-quality sampling was done in observation wells screened at depths of less than 45 ft. This depth criterion allowed direct comparison of the results for these wells with those for the shallow county wells. The differences between results for these two groups of wells indicate the effect of the industrial park on ground-water quality. The statistical summary of the industrial park water-quality results is shown in table 10.

Inorganic Water Quality

Several general similarities between the inorganic water quality of the shallow industrial-park ground water (table 10) and the shallow county ground water (table 7), respectively, were observed--slightly basic pH (medians, 7.6 and 7.8), high hardness (averages, 260 and 210 mg/L as $CaCO_3$), high alkalinity (averages, 210 and 150 mg/L as $CaCO_3$), and oxidizing redox potential (averages, +261 and +246 mv). Calcium bicarbonate water type was predominant in both groups of wells. Furthermore, owing to the similarity in oxidizing conditions, concentrations of redox-affected parameters (iron, manganese, sulfate, and trace metals) were comparable in water from both groups of wells.

Table 10.--Statistical summary of water-quality analyses for shallow industrial-park wells, 1978-79

[All units of measure in milligram per liter except pH or as indicated; ND, not determined

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Temperature (°C)	66	13.5	7	19.0	7.5
pH	66	[1]7.6	1	8.3	7.2
Specific conductance (µmho/cm at 25° C)	66	585	4	998	396
Dissolved oxygen	66	1.6	64	7.7	.1
Redox potential (mv)	66	+261	43	+418	+15
Hardness, tot. (as $CaCO_3$)	66	260	3	380	190
Dissolved solids[2]	66	339	4	572	223
Alkalinity, tot. (as $CaCO_3$)	66	210	3	290	150
Chloride, diss.	66	29	32	110	6
Sulfate, diss.	66	28	20	57	<.5
Fluoride, diss.	22	.1	ND	.1	.1
Bromide, diss.	22	.1	33	.2	.1
Silica, diss.	66	11	5	14	9
Calcium, diss.	66	74	4	110	53
Magnesium, diss.	66	19	6	27	14
Sodium, diss.	66	15	24	55	3
Potassium, diss.	66	1.4	31	4.2	.7
Cyanide, tot.	22	<.005	ND	<.005	<.005
Iron, diss.	66	.01	97	4.10	<.005

Table 10.--Statistical summary of water-quality analyses for shallow industrial-park wells, 1978-79--Continued

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Manganese, diss.	66	0.01	62	0.70	<0.005
Aluminum, diss.	66	<.005	105	.20	<.005
Arsenic, diss.	22	.001	45	.011	.001
Boron, diss.	22	.04	23	.23	<.005
Barium, diss.	22	<.05	124	.4	<.05
Cadmium, diss.	22	<.0005	160	.003	<.0005
Chromium, diss.	22	<.005	212	.52	<.005
Mercury, diss.	--	ND	ND	ND	ND
Lead, diss.	22	.001	119	.003	<.0005
Selenium, diss.	--	ND	ND	ND	ND
Phenols, tot.	22	<.0005	274	.037	<.0005
Organic carbon, diss.	66	3.1	48	16	.2
Ammonia as N, diss.	66	.04	332	1.0	<.005
Nitrate as N, diss.	66	2.1	65	9.9	<.005
Nitrite as N, diss.	66	.02	262	.33	<.005
Organic nitrogen as N, diss.	66	.10	125	.78	<.005
Orthophosphate as P, diss.	66	.01	195	.10	<.005

---

[1] Median value.

[2] Residue on evaporation at 180° C.

Chromium concentration of water in well 116 (0.52 mg/L) exceeded the NIPDWR limit of 0.05 mg/L (table 9) during the September 1979 sampling. The reason for this single high concentration is not known, although chromium is often associated with metallic plating wastes. Confirmation of this result is needed before any further conclusions can be made.

Dissolved-solids concentration and specific conductance were both generally higher in the industrial-park wells than in the county wells (fig. 43). The 500-mg/L NSDWR limit for dissolved solids (table 9) was exceeded in water from two wells, 101 and 103, during both autumn samplings. These high dissolved-solids concentrations were primarily due to increases in chloride, sodium, calcium, and bicarbonate concentrations. These constituents were probably introduced into the shallow ground-water system by infiltration of surface runoff containing road salt, lawn fertilizers, septic-tank effluent, and other chemicals commonly associated with urbanized areas.

Concentrations of iron and manganese exceeded their secondary drinking-water limits in water from only 9 and 18 percent of all shallow industrial-park wells, respectively. Where present, these high concentrations were attributed to the natural abundance of iron and manganese in the area and not to any man-induced input.

Concentrations of all other inorganic parameters were less than their respective drinking-water limits.

Organic Water Quality

The organic water quality of the industrial-park area ground water was of particular interest because eight volatile organic compounds had been detected by the U.S. Environmental Protection Agency in four nearby residential wells in 1976 (Billy Fairless, U.S. Environmental Protection Agency, written commun., 1976). Water samples were initially collected for analysis to define the distribution of dissolved organic carbon. Subsequently, samples were analyzed for three specific groups of organic compounds: acid-extractable (phenolic) organic compounds, base/neutral extractable organic compounds, and volatile organic compounds. Acid-extractable and base/neutral extractable organic compounds (together, the nonvolatile fraction) were determined in addition to the volatile fraction to learn if any other types of organic compounds were a problem. Concentrations of a fourth group, pesticides and herbicides, were not determined because previous analyses by the USEPA had indicated that these compounds were not in the ground water (Ralph C. Pickard, Indiana State Board of Health, written commun., 1977).

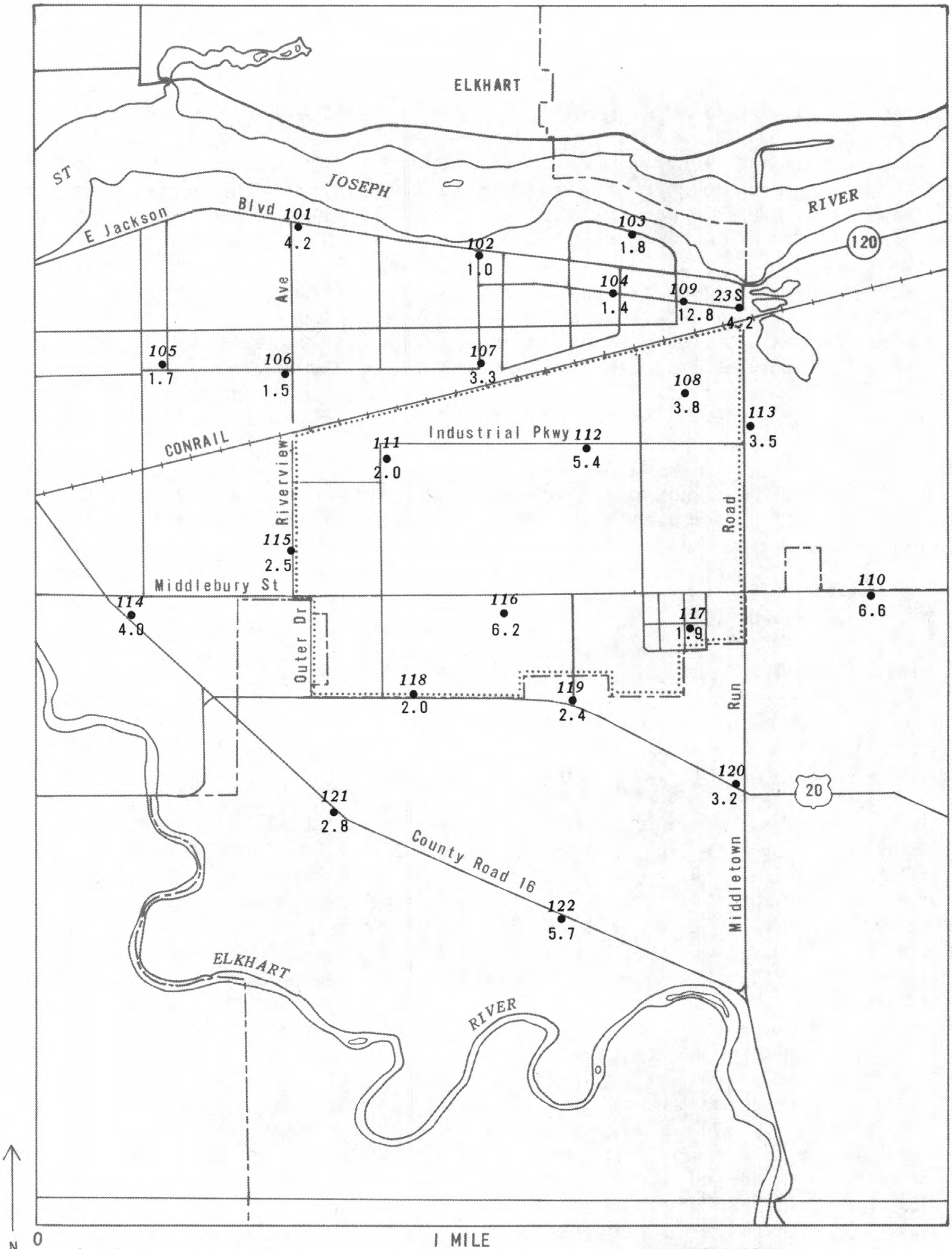
Hughes and others (1974) reported that dissolved organic carbon was a useful and definitive indicator in outlining areas of degraded water in an alluvial aquifer in Barstow, Calif. However, samples collected for determining this constituent were not useful in determining the distribution of organic compounds in the current study. The mean concentrations of the constituent for the industrial-park wells and the shallow county wells were not significantly different. The mean DOC concentrations of water from the wells in the industrial-park network are shown in figure 44. (Site 23S is included because of its proximity to the industrial park.) The figure indicates that only water from wells 109, 110, 112, 116, and 122 had concentrations greater than the 5-mg/L dissolved-organic carbon concentration suggested as a threshold concentration for organic contamination of ground water by Leenheer and others (1974). No areal pattern of DOC concentrations could be defined from the results. Either the variation in concentration due to sampling and analytical errors masked the areas of high concentrations of organic compounds or the concentrations were too low to be measured by the analytical procedure. Subsequent sampling for specific types of organic compounds was done to cover as much as possible of the area where organic compounds seemed to be a problem in the ground water.

Concentrations of phenols, the principal compounds constituting the acid-extractable organic compounds, were determined for samples from all industrial-park wells in the autumn of 1979. Concentrations of phenols in water from wells 108 and 115, 6 and 37 µg/L, respectively, exceeded the suggested 1 µg/L standard (U.S. Environmental Protection Agency, 1976). The reason for the high concentrations is not known.

Samples containing base/neutral extractable organic compounds were collected and analyzed in the spring of 1979. Results indicated trace amounts ($\leq$1 µg/L) of only one compound (butyl phthalate) in this class. Because no significant concentrations of this type of organic compound were found, no additional samples were collected.

The third group of organic compounds investigated was the volatile organic compounds. Samples for determining concentrations of these compounds were collected in April and September 1979. For both samplings, a complete gas chromatography-mass spectrometry scan was initially run on a sample from the stainless-steel well 109C. This well was chosen because it was close to the wells where the USEPA had detected the volatile organic compounds in 1976 and its stainless-steel construction minimized the possibility of contamination of the organic samples. Concentrations of compounds identified in the sample from well 109C were then determined in subsequent samples. Concentrations of the voltile organic compounds determined are given in table 11.

The combined volatile-organic-compound concentration made up only a small fraction of the dissolved organic carbon in almost every sample. This fact explains why DOC was not an effective indicator for organic degradation of ground water in the industrial park. Approximately one-fourth of the samples collected from the same wells in both April and September 1979 had concentrations of volatile organic compounds that differed by an order of one magnitude or more. Some compounds detected in one set of samples were not detected in the next. Differences were also noted between the results from the three wells at site 109, which were screened at the same depth and were within 30 ft of each other. Analytical recoveries determined by "spiking" samples with known concentrations of four of the volatile organic compounds,



EXPLANATION

.... Industrial park boundary

122 ● 5.7 Well site and designation

Mean dissolved-organic carbon concentration, in milligrams per liter. Site 109 contains three wells

Figure 44.-- Mean dissolved-organic carbon concentrations for the industrial-park wells, 1978-79.

Table 11.--Volatile-organic-compound determinations for the industrial-park wells, 1978-79

[All values are microgram per liter: top line is April 1979 sampling result: bottom line is September 1979 sampling result; ND, not determined; --, below detection limit]

Sites	Methylene chloride	Trichlorofluoromethane	Chloroform	1,1-Dichloroethane	1,1,1-Trichloroethane	Trichloroethylene	Toluene	1,2-Dichloropropane
101	--	--	--	--	--	--	--	ND
	ND	ND	ND	ND	ND	ND	ND	ND
102	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
103	--	--	--	--	87	--	--	ND
	--	--	--	--	52	--	--	--
104	51	--	--	37	1,160	358	--	ND
	--	--	--	--	606	459	--	--
107	--	--	--	--	--	82	--	ND
	--	--	--	--	--	364	--	--
108	31	--	--	--	31	17	--	ND
	--	--	--	--	18	29	--	--
109A	1,819	126	49	513	220	--	--	ND
	76	--	--	221	193	15	--	--
109B	4,400	113	--	850	156	--	--	ND
	794	--	--	--	81	--	--	424
109C	430	77	4	120	176	--	11	ND
	181	15	--	--	214	22	724	855
110	--	--	--	--	--	--	--	ND
	--	--	--	--	--	--	--	--
111	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
112	--	--	--	--	--	40	--	ND
	--	--	--	--	--	531	--	--
113	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	315	--	--
115	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
116	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
117	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
118	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
119	ND	ND	ND	ND	ND	ND	ND	ND
	--	--	--	--	--	--	--	--
23S	--	--	--	--	--	--	--	ND
	--	--	--	--	--	--	7	--
Percent analytical recoveries	71	70	ND	ND	87	100	ND	ND

shown at the bottom of table 11, demonstrated that analytical variation could introduce as much as a 30-percent error in some results. Losses of the volatile compounds during sampling and analysis, as well as the overall low concentrations (a few micrograms per liter) of these compounds were the factors responsible for the variability in concentrations.

In spite of the limitations of the data, several important observations and conclusions can be drawn. Of the eight volatile organic compounds detected (table 11), four (methylene chloride, toluene, 1,1,1-trichloroethane, and 1,1-dichloroethane) were also detected in the analyses by the USEPA of samples from residential wells in 1976 (Billy Fairless, U.S. Environmental Protection Agency, written commun., 1976). These results verified the presence of certain volatile organic compounds in the ground water of the industrial-park area and indicated that several of these compounds have been there at least 3 yr.

Volatile organic compounds were detected in water from only eight wells: 103, 104, 107, 108, 109, 112, 113, and 23S. As shown in figure 45, all these sites are in the northeast part of the industrial park and the adjacent residential area.

Concentrations of the volatile organic compounds were not necessarily higher in water from wells within the industrial park than in water from wells outside the park. The highest average concentrations for all organic compounds were at sites 104 and 109 in the adjacent residential area. Furthermore, the concentrations of trichloroethylene and 1,1,1-trichloroethane, the two most widely distributed compounds, were dissimilar in areal distribution. Several explanations for these findings are possible: (1) The source of the organic compounds is not within the industrial park; (2) the source is within the industrial park but is north of the observation wells sampled; or (3) there is more than one source for the different organic compounds. Additional data would be needed to determine the source of these compounds. Because the organic compounds were localized over a small area, the industrial-park observation-well network was probably not sufficiently detailed for accurately locating a source or delineating a plume of the detected organic compounds.

None of the National Interim Primary Drinking Water Regulations were exceeded in the analyses for organic compounds because the only organic compounds having limits are total trihalomethanes (U.S. Environmental Protection Agency, 1979b, p. 68625) and several pesticides and herbicides (U.S. Environmental Protection Agency, 1975). Only one trihalomethane (chloroform) was identified in the study, and its concentration did not exceed the 100-μg/L limit.

Only one volatile organic compound (trichloroethylene) exceeded USEPA Health Effects Advisory Limits (National Research Council, 1980). The average concentration of this compound in water from industrial-park wells 104, 107, 112, and 113 exceeded the 200-μg/L short-term maximum limit (Joseph Prince, U.S. Environmental Protection Agency, written commun., September 25, 1979). The 20-μg/L long-term limit was exceeded in at least one sample from six of the seven wells (104, 107, 108, 109C, 112, and 113), where trichloroethylene had been detected.

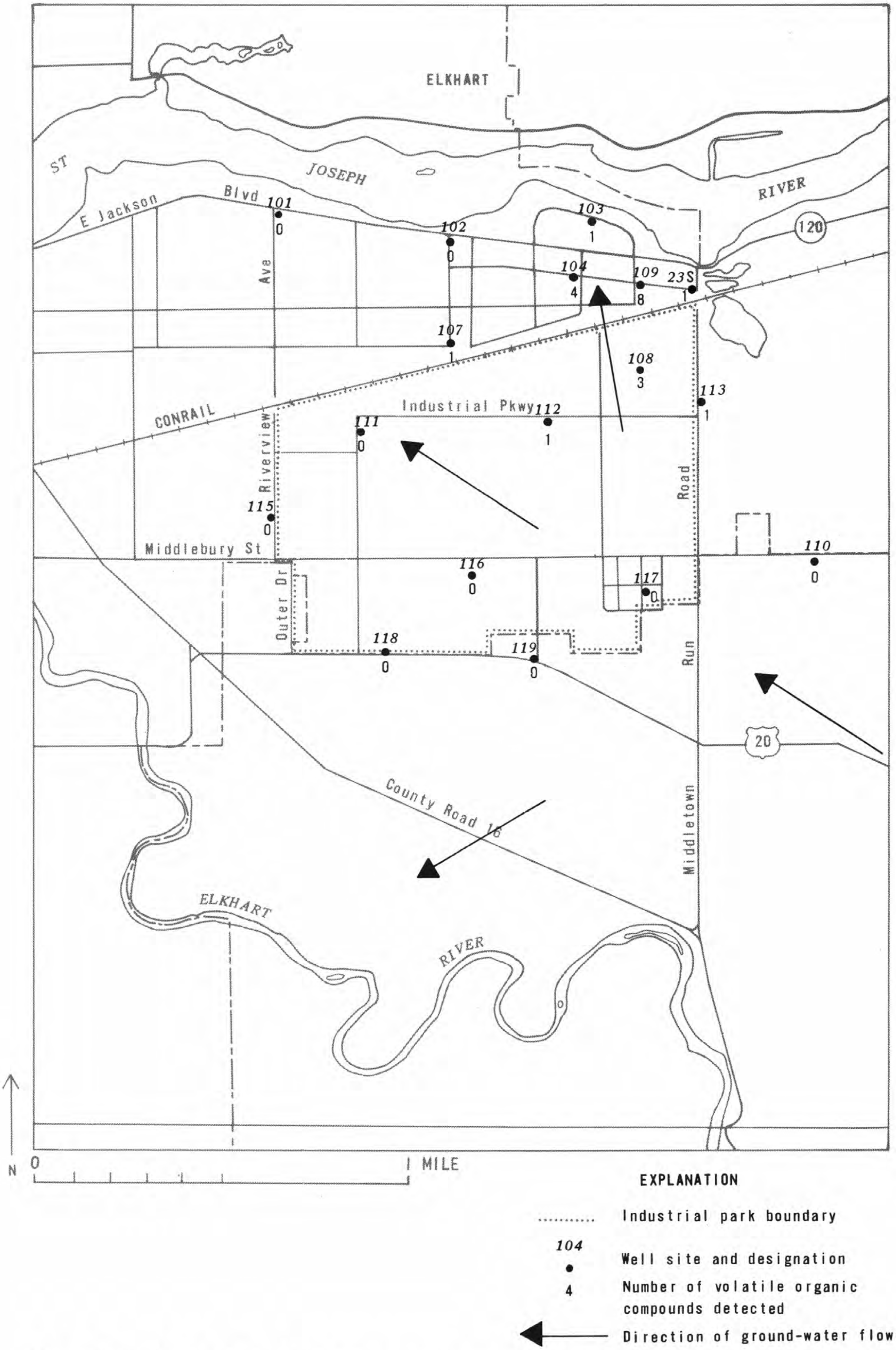


Figure 45.-- Areal extent of volatile organic compounds detected in industrial-park wells, 1979.

Though the methylene chloride concentrations detected did not exceed the short-term (7-day) advisory limit of 5 mg/L (National Research Council, 1980, p. 128), concentrations in water from wells 109A and 109B were 1.8 and 4.4 mg/L, respectively. Whether these concentrations exceed any long-term limit for this compound is not known because data for calculating such a standard are insufficient.

Landfill Wells

Shallow and deep landfill wells were separated into groups by depth (≤ 45 ft and >45 ft) to allow simplified analysis of water-quality data and comparison of these data with shallow and deep county-well data. This separation by depth resulted in groups of wells that correspond fairly closely to the unconfined and the confined aquifers at locations where a confining silt and clay layer was present. The statistical summaries of the water-quality data for these shallow and deep landfill wells are given in tables 12 and 13, respectively.

There are several general similarities in water quality between the shallow and the deep landfill wells--slightly basic pH (medians, 7.5 and 7.6), high alkalinity (averages, 210 and 210 mg/L as $CaCO_3$), and high hardness (averages, 280 and 230 mg/L as $CaCO_3$), respectively. The water type of more than 80 percent of the samples collected from the landfill wells was calcium bicarbonate.

Comparison of the shallow and the deep landfill-well data showed that partially oxidizing conditions (average redox potential, +161mv; average dissolved-oxygen concentration, 0.9 mg/L) were present in the shallow landfill well water, whereas reducing conditions (average redox potential, +26 mv; average dissolved-oxygen concentration, <0.05 mg/L) were present in the deep landfill well water. Concentrations of iron and silica in water from the deep wells were significantly higher than the concentrations of these constituents in water from the shallow wells, probably because of the reducing conditions and longer contact times between the water and the aquifer materials in the deep wells. Dissolved-solids, chloride, sulfate, calcium, sodium, and potassium concentrations and specific-conductance values were higher in the shallow aquifer than in the deep aquifer. The differences are probably due to the presence of landfill leachate and its effects on water quality in the shallow wells.

Comparison of the water-quality data for deep county wells with those for deep landfill wells (tables 8 and 13) shows few differences. The data indicate that reducing conditions and concentrations of most major cations and anions, trace elements, nutrients, and dissolved organic carbon in water from the two groups of wells are similar.

The National Interim Primary Drinking Water Regulations were not exceeded in any of the samples from the deep landfill wells. However, the National Secondary Water Regulations (table 9) for dissolved solids, iron, and manganese were exceeded. Dissolved-solids concentrations of water from wells M2 and E1 (fig. 41), which are downgradient from the landfill, exceeded 500 mg/L. Seventy-eight and 88 percent of the iron and manganese concentrations, respectively, of all the deep landfill-well samples exceeded the National Secondary Drinking Water Regulations for these two metals. This similarity of water quality in the deep landfill wells and the deep county wells indicates that the ground water underlying the Elkhart area is naturally high in iron and manganese.

The 5-mg/L limit suggested by Leenheer and others (1974) as a threshold concentration for organic degradation was exceeded in all four samples collected from well E1. This high dissolved-organic carbon concentration was probably caused by leachate from the landfill.

Comparison of the data for shallow county and shallow landfill wells (tables 7 and 12) shows several significant differences in water quality. Samples from the shallow landfill wells generally had little dissolved oxygen (0.9 mg/L) and only partially oxidizing conditions (+161 mv). Mean dissolved-solids concentrations and specific-conductance values of water from shallow landfill wells were higher than those of water from the shallow county wells. Concentrations of chloride, sulfate, calcium, sodium, potassium, and iron in water from the shallow landfill wells were also higher. Leaching of waste materials from the landfill into the shallow aquifer was probably the cause of the differences.

Nitrate concentrations of water from the shallow landfill wells were lower than those concentrations of water from the shallow county wells. This may have been caused by denitrification or nitrate behaving as an oxidizing agent in the landfill well water having a low dissolved-oxygen concentration. Ammonia concentrations being higher in the shallow landfill well water than in the shallow county well water supports the idea of nitrate behaving as an oxidizer.

Nitrate concentration exceeded the 10-mg/L NIPDWR limit (table 9) in three samples, one from well E2 and two from well A2 (fig. 41). The high nitrate in water from well E2, directly downgradient from the landfill, is probably caused by the leachate. The high nitrate in water from well A2, upgradient from the landfill but in the median of a road, may have been caused by runoff from adjacent farm fields.

In April 1979, the chromium concentration of water from well M1, in the landfill, exceeded the 0.05-mg/L NIPDWR limit, but, in September 1979, the concentration fell below the limit.

During the study, the NSDWR limit for dissolved solids (table 9) was exceeded in water from wells B2, M1, E2, P1, N1, Q1, and F3. All the high dissolved-solids concentrations except that in well F3 water were probably due to the effect of the landfill leachate. Well F3 is near a main highway, and the high dissolved-solids concentration of its water was probably due to a high chloride concentration contributed by a highway deicing chemical.

Table 12.--Statistical summary of water-quality analyses for shallow landfill wells, 1978-79

[All units of measure in milligram per liter except pH or as indicated; ND, not determined]

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Temperature (°C)	62	12.5	8	20.0	7.5
pH	62	[1]7.5	2	8.7	6.1
Specific conductance (μmho/cm at 25° C)	62	752	10	9,220	316
Dissolved oxygen	62	.9	125	10.0	<.05
Redox potential (mv)	60	+161	91	+490	-172
Hardness, tot. (as $CaCO_3$)	62	280	10	3,400	81
Dissolved solids[2]	62	449	11	8,040	181
Alkalinity, tot. (as $CaCO_3$)	62	210	16	3,100	39
Chloride, diss.	62	32	24	180	6
Sulfate, diss.	62	58	25	1,300	5
Fluoride, diss.	32	<.05	115	.1	<.05
Bromide, diss.	29	.8	105	7.1	<.05
Silica, diss.	62	10	17	56	1
Calcium, diss.	62	76	14	1,200	24
Magnesium, diss.	62	21	18	93	5
Sodium, diss.	62	29	35	750	6
Potassium, diss.	62	3.1	64	53	1.0
Cyanide, tot.	30	<.005	305	.03	<.005
Iron, diss.	62	.15	45	14	<.005

Table 12.--Statistical summary of water-quality analyses for shallow landfill wells, 1978-79--Continued

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Manganese, diss.	62	0.04	54	4.40	<0.005
Aluminum, diss.	62	.01	72	.38	<.005
Arsenic, diss.	46	.001	93	.029	<.0005
Boron, diss.	45	.05	34	1.90	<.005
Barium, diss.	45	<.05	114	.3	<.05
Cadmium, diss.	46	<.0005	110	.002	<.0005
Chromium, diss.	46	<.005	124	.07	<.005
Mercury, diss.	31	<.0005	ND	<.0005	<.0005
Lead, diss.	46	.001	90	.010	<.0005
Selenium, diss.	31	<.0005	387	.001	<.0005
Phenols, diss.	29	.001	400	.001	<.0005
Organic carbon, diss.	62	6.1	66	3,280	.6
Ammonia as N, diss.	62	.88	237	450	<.005
Nitrate as N, diss.	62	.93	124	16	<.005
Nitrite as N, diss.	62	.02	213	.19	<.005
Organic nitrogen as N, diss.	62	.52	579	750	<.005
Orthophosphate as P, diss.	62	.02	368	.60	<.005

---

[1] Median value.

[2] Residue on evaporation 180° C.

Table 13.--Statistical summary of water-quality analyses for deep landfill wells, 1978-79

[All units of measure in milligram per liter except pH or as indicated; ND, not detected]

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Temperature (°C)	74	11.5	2	15.5	10.5
pH	74	[1]7.6	1	8.5	7.3
Specific conductance (µmho/cm at 25° C)	74	484	4	1,050	356
Dissolved oxygen	74	<.05	ND	.1	<.05
Redox potential (mv)	69	+26	164	+118	-179
Hardness, tot. (as $CaCO_3$)	74	230	3	390	180
Dissolved solids[2]	74	266	5	624	137
Alkalinity, tot. (as $CaCO_3$)	74	210	4	390	130
Chloride, diss.	74	10	38	66	1
Sulfate, diss.	74	13	48	210	<.05
Fluoride, diss.	44	.1	43	.3	<.05
Bromide, diss.	30	.4	148	3.1	<.05
Silica, diss.	74	14	5	20	9
Calcium, diss.	74	57	4	100	44
Magnesium, diss.	74	20	5	33	15
Sodium, diss.	74	10	39	100	3
Potassium, diss.	74	1.1	39	6.8	.5
Cyanide, tot.	29	<.005	ND	<.005	<.005
Iron, diss.	74	.48	.11	1.40	.08

Table 13.-- Statistical summary of water-quality analyses for deep landfill wells, 1978-79--Continued

Parameter	Number of analyses	Mean	Coefficient of variation (percent)	Range	
				Maximum	Minimum
Manganese, diss.	74	0.07	20	0.25	<0.005
Aluminum, diss.	74	<.005	85	.03	<.005
Arsenic, diss.	52	.004	53	.030	<.0005
Boron, diss.	52	.02	44	.25	<.005
Barium, diss.	52	<.05	98	.2	<.05
Cadmium, diss.	52	<.0005	128	.002	<.0005
Chromium, diss.	52	<.005	139	.02	<.005
Mercury, diss.	37	<.0005	ND	<.0005	<.0005
Lead, diss.	52	.001	96	.013	<.0005
Selenium, diss.	37	<.0005	ND	<.0005	<.0005
Phenols, tot.	30	<.0005	267	.002	<.0005
Organic carbon, diss.	74	3.6	40	33	.4
Ammonia as N, diss.	74	.15	89	1.3	<.005
Nitrate as N, diss.	74	.01	131	.07	<.005
Nitrite as N, diss.	74	<.005	604	.01	<.005
Organic nitrogen as N, diss.	74	.13	172	3.4	<.005
Orthophosphate as P, diss.	74	<.005	269	.09	<.005

---

[1] Median value.

[2] Residue on evaporation at 180° C.

Forty-eight and 61 percent of the iron and manganese concentrations, respectively, of the shallow landfill-well samples, exceeded the NSDWR limits for these two metals. These percentages were approximately twice those for the shallow county-well samples, which indicate that the landfill was a source of these metals and that the redox environment it created was favorable for them to remain in solution.

Sulfate concentrations of water from wells B2, M1, and E2 exceeded the 250-mg/L NSDWR limit for this constituent. These high concentrations were apparently due to the effect of the landfill.

Dissolved-organic carbon concentrations exceeded the 5-mg/L threshold concentration suggested by Leenheer and others (1974) in at least 75 percent of the samples collected from shallow landfill wells M1, E2, N1, P1, Q1, I3, F3, and A2. All wells except F3 and A2 were at sites directly downgradient from the landfill.

Two volatile organic compounds, trichloroethylene (55 μg/L) and toluene (23 μg/L), were identified in a sample from well M1 in the landfill. However, in water from well E2, 300 ft downgradient from M1, concentrations of these compounds were less than 2 μg/L. These results indicate that the compounds either did not stay in solution or were diluted very rapidly in the ground water to less than detection limits.

Effect of the Landfill on Ground-Water Quality

In the preceding section, the landfill was cited as the probable source of the high concentrations of many dissolved constituents. The high concentrations resulted from the formation of leachate within the landfill and movement of the leachate downgradient from the site. To understand the effect of the landfill on ground-water quality, one must understand the origin of leachate and some of its common chemical characteristics.

Precipitation on a landfill may leave the area as surface runoff, return to the atmosphere by evapotranspiration, or infiltrate the landfill. If it infiltrates, the water percolates through the solid waste and may dissolve chemical species. The percolating water contains dissolved oxygen that can be used by microbiological populations active in the landfill to oxidize organic matter. Because of the large quantity of organic material in most landfills, the supply of dissolved oxygen is usually not sufficient to satisfy the demand. Thus, anaerobic conditions are created in the water while it is still in the landfill. Under these reducing conditions, solvent action results in a solution highly enriched in many dissolved chemical constituents and generally referred to as leachate. Leachate moves from the landfill into the ground-water system and then moves in response to ground-water and density gradients. It may eventually form a plume of water containing high concentrations of dissolved constituents.

The general chemical characteristics of landfill leachate are a low dissolved-oxygen concentration, a low redox potential, and a high dissolved-solids concentration. The specific constituents of the dissolved solids are different for each landfill because of the unique combination of solid wastes buried at each site (Boggess, 1975; Zenone and others, 1975; and Kimmel and Braids, 1980).

In the present study, concentrations of chloride, sulfate, calcium, sodium, potassium, and iron in water from the shallow landfill wells were significantly higher than concentrations of those constituents in water from the shallow county wells. Concentrations of bicarbonate, magnesium, bromide, manganese, dissolved organic carbon, ammonia, and organic nitrogen in water from shallow wells in the vicinity of the landfill were also higher.

The chemistry of the leachate can be explained more clearly if the data for the set of shallow landfill wells along the line C-C' (fig. 41) are examined. Included in this group are wells M1, E2, Q1, I3, and K1. Chemical analyses of samples collected from the first four wells in September 1979 and from well K1 in October 1978 are given in table 14. Mean values for shallow county-well analyses are included for background concentration reference.

The water-quality results obtained from sampling well M1 in the landfill were assumed to represent the concentrations of all parameters in the undiluted leachate. In water from this well, there was no detectable dissolved oxygen, a very low redox potential, and a dissolved-solids concentration more than 25 times that of the mean background concentration of the shallow county well water. All other concentrations for M1, shown in table 14, except nitrate, were at least five times the concentrations of dissolved constituents in the shallow county well water. Even for these high concentrations, the water type was still calcium bicarbonate.

In the absence of dissolved oxygen, bacterial populations in the landfill use nitrate, sulfate, and less-reduced organic compounds as oxidizing agents. Dissolved-nitrate concentrations of water from well M1 were low, but ammonia concentrations were very high, which indicates that most of the nitrate had already been oxidized. Sulfate concentration was high, and a strong hydrogen sulfide odor was noted. This suggests that sulfate was being reduced by bacteria.

The high concentration of manganese in water from well M1 (4.4 mg/L) can be attributed to the reducing conditions under the landfill. Relative to manganese, the concentration of iron was low. Ferrous iron (Fe^{+2}) is known to precipitate with hydrogen sulfide under the pH and redox conditions of the ground water beneath the landfill (Stumm and Morgan, 1970, p. 529). Thus, much of the iron was probably present as an insoluble sulfide.

High concentrations of other constituents were probably due to wastes buried in the landfill. These constituents were mobilized by exchange and dissolution reactions taking place during the formation of leachate.

Table 14.--Chemical analyses of water samples from selected shallow landfill wells, September 1979

[All units of measure in milligram per liter except as indicated]

Parameters	Shallow county wells[1]	Shallow landfill wells				
		M1	E2	Q1	I3	K1[2]
Oxygen, diss.	2.4	<0.05	<0.05	<0.05	<0.05	<0.05
Redox potential (mv)	+246	-133	+130	+51	+21	+22
Dissolved solids[3]	280	7,830	1,640	448	358	222
Specific conductance (µmho/cm at 25° C)	484	9,220	2,840	793	638	356
Alkalinity as $CaCO_3$	150	2,380	1,100	380	300	150
Chloride, diss.	17	98	98	23	17	8
Sulfate, diss.	27	620	190	8	5	36
Bromide, diss.	.1	7.1	5.9	2.3	1.6	.2
Sodium, diss.	8	570	400	54	20	5
Potassium, diss.	1.1	51.0	53.0	6.5	1.3	1.0
Magnesium, diss.	16	90	75	24	26	15
Calcium, diss.	57	1,200	64	77	82	53
Iron, diss.	.03	.64	.93	1.60	1.20	.38
Manganese, diss.	.01	4.40	.24	.10	.10	.14
Ammonia as N, diss.	.02	450	20	4.1	.34	.08
Nitrate as N, diss.	2.3	.09	3.8	.01	.76	<.005
Organic carbon, diss.	4.4	3,280	44	11	6.6	9.1

[1] Shallow county well results are mean values from table 7, which represent background concentrations.

[2] K1 was sampled for bromide only in September 1979. Other data for K1 are from October 1978 sampling.

[3] Residue on evaporation at 180° C.

At well E2, 300 ft downgradient from well M1, several significant changes in the leachate chemistry were observed. Although no dissolved oxygen was detected in the water, an increase in the redox potential indicated that the system had become less reducing. The dissolved-solids concentration decreased to less than a third of the concentration of water from well M1.

The water type changed from calcium bicarbonate to sodium bicarbonate, principally because of a significant decrease in dissolved calcium. Two types of reactions may have been responsible for the observed change in water type. The water may have been softened, immediately downgradient from the landfill, by exchange of dissolved calcium for potassium or sodium on the aquifer materials. The potassium concentration increased slightly in the 300 ft between wells M1 and E2, whereas sodium concentration decreased less than would have been possible by simple dilution or dispersion. Another explanation for the decrease in calcium concentration is that some calcium minerals may have precipitated between wells M1 and E2. WATEQF, a chemical-equilibrium computer program, was run with the data from both M1 and E2. The results showed that solubility products for calcite, aragonite, dolomite, and several aluminosilicates that contain calcium were exceeded in both sets of data. Thus, some calcium minerals may have precipitated under the conditions in the system at site E. A combination of ion exchange and precipitation was probably responsible for the decrease in dissolved calcium downgradient from well M1.

Ammonia concentration of water from well E2 was low in comparison with that of water from well M1. However, nitrate concentration of water from well E2 was high in comparison with that of water from well M1. These findings indicated that nitrate-reducing bacteria were probably the predominant microbiological influence affecting the water quality at well E2. Furthermore, little or no hydrogen sulfide odor was detected in water from well E2. Thus, very few if any sulfate-reducing bacteria were active.

Wells Q1, I3, and K1 are progressively downgradient from site M. Data in table 14 show that concentrations of most parameters gradually decreased to near-background concentrations at well K1. The low dissolved-oxygen and nitrate concentrations and the hydrogen sulfide odor noted in these wells indicated that sulfate-reducing bacteria were active in the shallow unconfined aquifer downgradient from the landfill.

Areal and Vertical Extents of the Landfill Leachate

The areal and the vertical extents of landfill-leachate plumes have classically been determined by observation of concentrations of certain indicator parameters. Clark and Piskin (1977) listed five important characteristics for an effective indicator: (1) Availability. The indicator must be available in significant concentrations in the waste material deposited in the landfill. (2) Mobility. The indicator must be readily solubilized from the solid waste and must move downgradient with the leachate. (3) Persistence. The indicator must be conservative; that is, it must not participate in ion exchange, adsorption, or precipitation reactions or microbiological interactions as it moves downgradient. (4) Contrast. The indicator must be present in significantly higher concentrations in the plume than in background water. (5) Analyticity. The indicator must be readily measurable by a standard method.

Bromide was the best indicator of landfill leachate because it satisfied all the necessary criteria without any qualifications. It had been shown to be a conservative, mobile, and measurable parameter by Jester and Uhler (1974, p. 1) and Jester and others (1977). Further, bromide was available in significant concentrations in the undiluted leachate, 50 to 70 times the background concentrations. Bromide concentration was at background everywhere in the study area except in and downgradient from the landfill.

Dissolved-solids concentration was unsuitable as an areal leachate indicator because it was affected by the infiltration of road salt to the shallow aquifer near streets in the landfill area. However, it was of some use as a vertical-plume indicator because there was sufficient contrast between background and plume water at depth. All other constituents considered as possible indicators were eliminated, primarily because they were not conservative or were derived from sources other than the landfill.

The bromide concentrations obtained from the shallow landfill wells in September 1979 were plotted areally (fig. 46) to indicate the location of the leachate plume. The lines of equal concentration are an estimate of the extent of the affected ground waters. The directional trend of the plume was generally south from the landfill toward the St. Joseph River. This trend coincides with the regional ground-water flow.

The bromide concentrations of water from shallow wells that are lateral and upgradient from the landfill (A2, C2, B2, D1, and O1) were not significantly higher than background concentrations of water from county wells, which indicates that the landfill was the source of the bromide downgradient from it. The east boundary of the plume lies between site F, where bromide was equal to background, and site P, where bromide concentration exceeded background. The west boundary of the plume was less definable because of a lack of data. However, all complaints concerning the quality of well water, received by the ISBH, came from residences directly south of the landfill boundary. These observations indicate that the leachate plume had not spread much laterally. The location of the south edge of the leachate front in September 1979 was between sites I and K, 3,100 to 5,600 ft downgradient from the landfill. Further definition of the front could not be made because of the wide well spacing and the lack of sampling wells in some parts of the area south of the landfill. The dashed line representing the 1-mg/L bromide concentration contour in figure 46 is an estimate of the extent of the plume.

The distance that leachate traveled downgradient from the landfill was calculated by multiplying estimates of ground-water velocity in the shallow aquifer and the length of time that the leachate had been produced. An average value for ground-water velocity of 3 ft/d was estimated for the vicinity

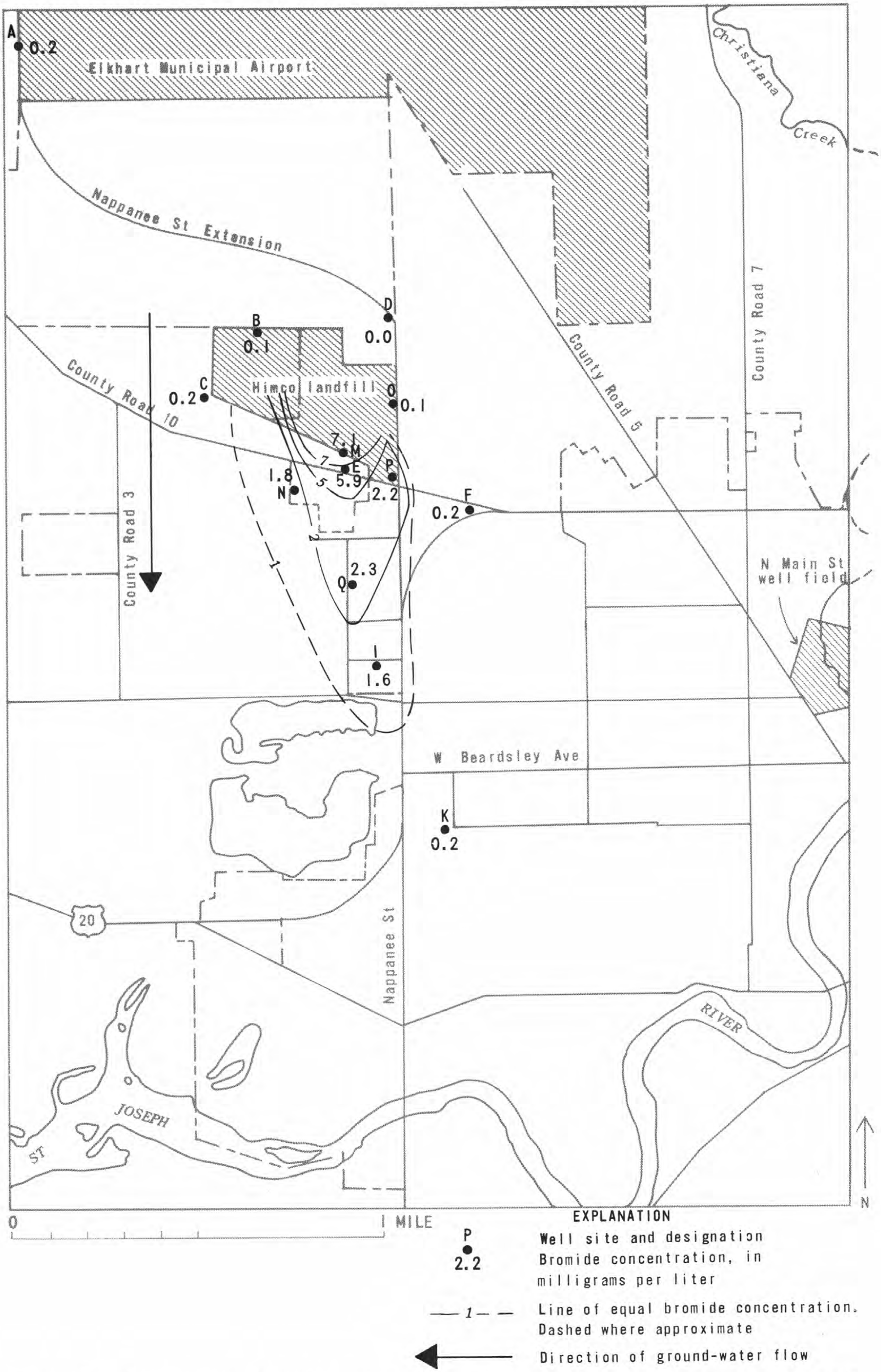


Figure 46.-- Areal extent of bromide concentrations for shallow landfill wells, September 1979.

of the landfill. The length of time that the leachate had been produced was not known, but it was estimated to be 5 yr on the basis of the date (1974) when the residential well complaints were received by the ISBH. The length of the plume was calculated to be approximately 5,500 ft. This length falls within the 3,100 to 5,600 ft distance indicated by the bromide data and demonstrates that the use of the bromide ion to estimate the extent of the leachate plume is reasonable.

The vertical extent of the leachate plume was determined by plotting the bromide data obtained in September 1979 for the sites on geologic section C-C' (fig. 47). Two important observations can be made from this data presentation:

(1) At site I, the low bromide concentrations of water in the 15-ft well and the high bromide concentrations of water in the 35-ft well indicated that the plume was sinking as it moved downgradient. This sinking trend was probably caused by the prevailing ground-water-flow direction and recharge distribution in this location. Difference in density between the leachate and the contiguous ground water was another possible explanation (U.S. Environmental Protection Agency, 1977).

(2) High bromide concentrations of water from the deep wells in and immediately downgradient from the landfill (M2, E1, and E3) indicated that the leachate had penetrated the aquifer to at least the depths of these wells. The high bromide concentration of water from the deep well at site I indicated that the leachate front was between sites E and I in the deep aquifer. The lack of other deep sampling wells in this area prevented a more accurate definition of the front's position.

There are various explanations of why the leachate penetrated the aquifer to the depths of the deep wells at sites M and E. For example, although no downward vertical gradients were detected from the available water-level data, downward flow could still have taken place during the history of the landfill. Also, with the absence of any significant clay or silt layer in the immediate vicinity of the landfill, the leachate may have simply dispersed and diffused down into the wells during the time that the landfill has been used, possibly being driven by density gradients or recharge patterns. Actually, both explanations probably apply.

The dissolved-solids concentrations of water at sites M, E, Q, and I in the autumn of 1979 and at site K in the autumn of 1978 were plotted in geologic section C-C' in figure 48. The vertical extent of the dissolved-solids data was nearly identical with that of the bromide data, where leachate was detected in the deep wells at sites M and E but not at site I. The contrast between the dissolved-solids concentrations of water from the shallow wells at sites I and K and in the plume was not as distinct as in other landfill wells, probably owing to infiltration of road salt.

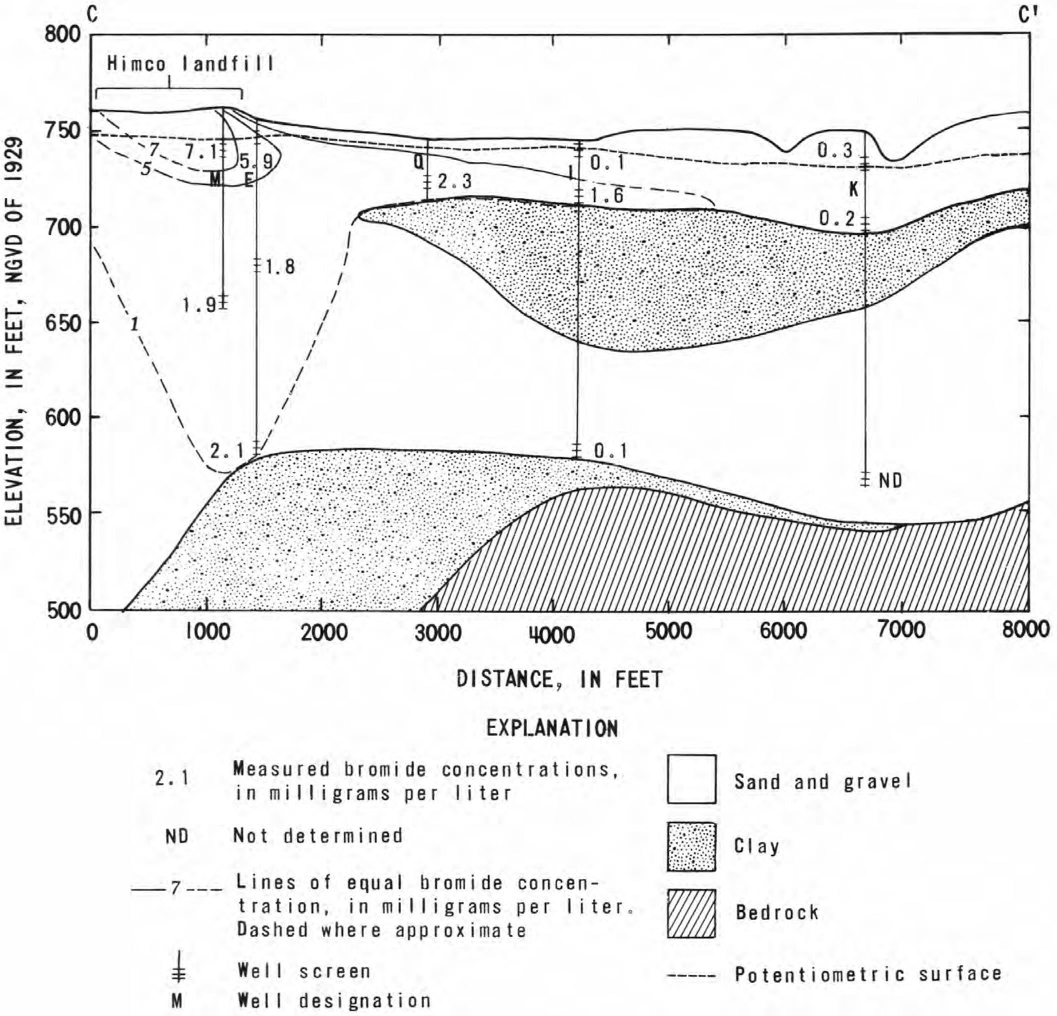


Figure 47.-- Geologic section showing vertical extent of bromide concentrations for landfill wells, 1979.

Attenuation of Leachate

The leachate from a landfill can be attenuated by physical mixing (dispersion, advection, and diffusion) or by chemical reactions and microbiological interactions that involve removal of dissolved chemical constituents from solution or changes in the dissolved chemical species. Attenuation of the leachate is demonstrated in figure 49, where enrichment ratio (sample concentration/background concentration) is plotted against distance downgradient from site M. This ratio decreases as the leachate is attenuated. Bromide, a conservative constituent, is an excellent example of a parameter that undergoes attenuation by physical mixing processes. The enrichment ratio of bromide (71) at well M1 gradually decreased to unity by the time that the leachate reached well K2 (5,600 ft downgradient from site M) because only the physical processes of dispersion, advection, and diffusion were involved.

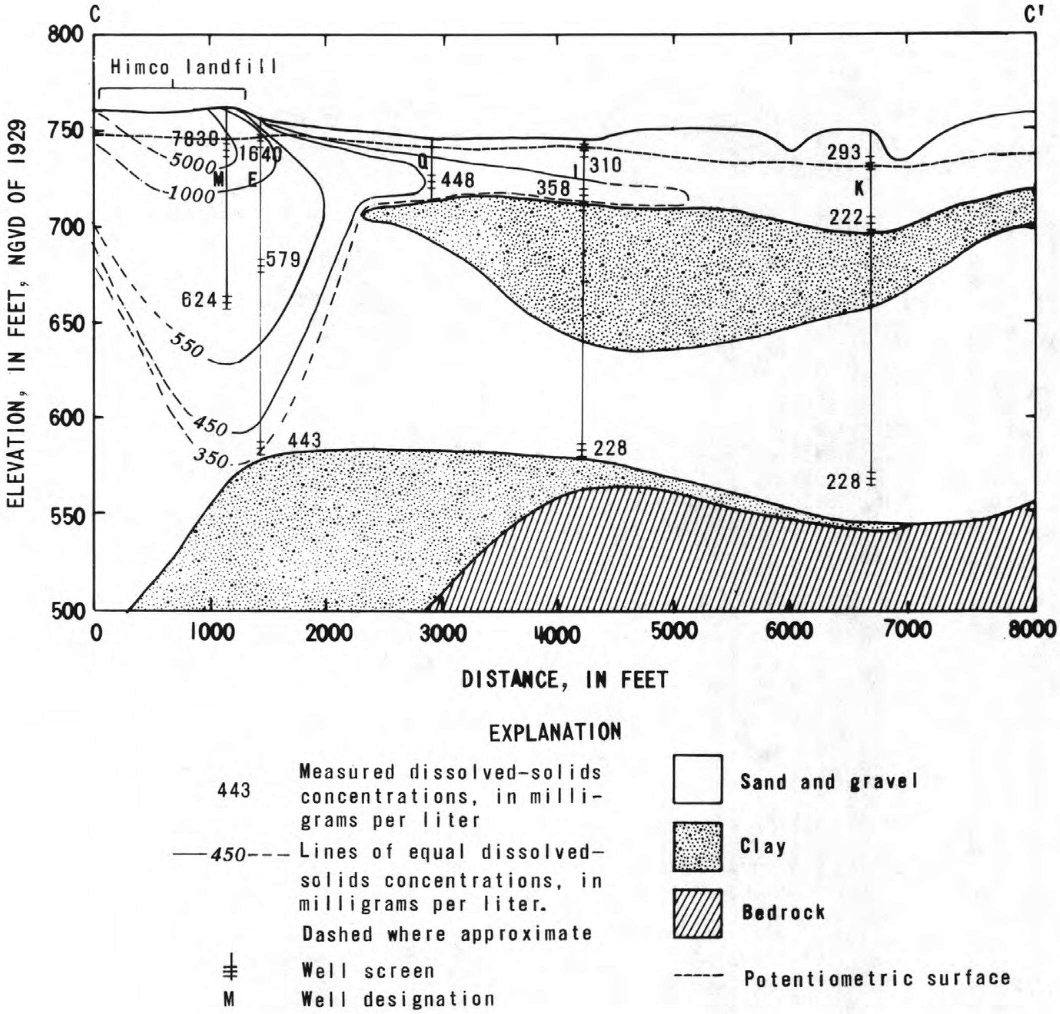


Figure 48.-- Geologic section showing vertical extent of dissolved-solids concentrations for landfill wells, 1978-79.

The landfill contains large quantities of two nonconservative parameters, calcium and sulfate. Calcium is involved in exchange and precipitation reactions, whereas sulfate is involved in microbiologically mediated reactions. The enrichment ratios for these two parameters were also plotted in figure 49. The calcium-enrichment ratio dropped to near unity within 300 ft downgradient from the landfill, whereas the sulfate enrichment ratio decreased to unity within 1,800 ft downgradient at site Q. The concentrations of calcium and sulfate decreased more rapidly than that of bromide because physical, chemical, and microbiological factors were affecting them. These factors restricted the distribution of calcium and sulfate to a localized area around the landfill. Bromide indicates the maximum extent of the leachate plume.

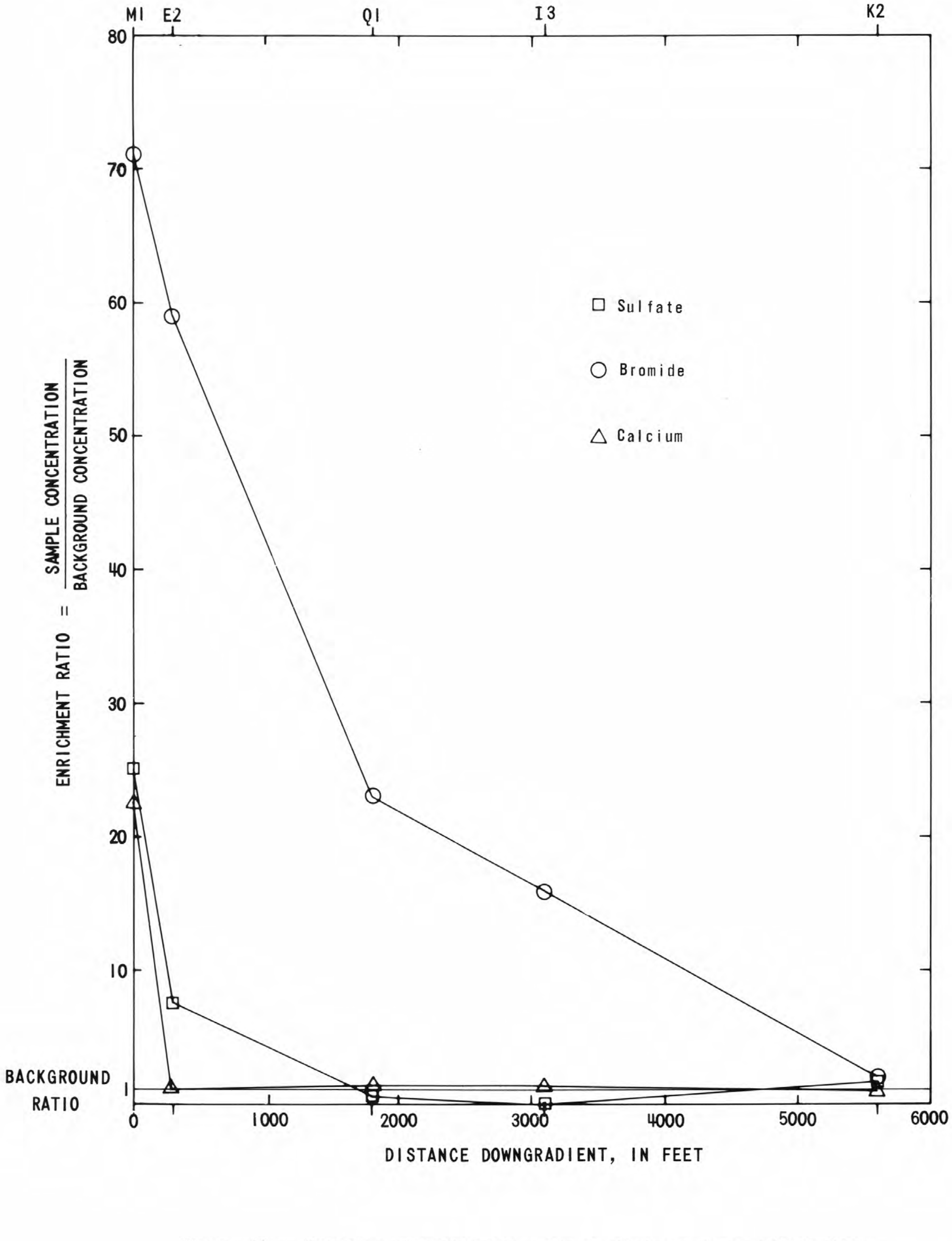


Figure 49.-- Relation of enrichment ratio to distance downgradient from site M for selected landfill wells.

Effects of Simulated Pumping on the Leachate Plume

The simulations in Part 1 indicated that the 20-Mgal/d pumping was the only pumping that could cause even a small percentage of water to flow from the landfill area to the well field proposed for the site at the airport. Even if significant quantities of leachate could be drawn into the area, figure 49 indicates that the leachate would be attenuated to background concentrations by the time it traveled the distance between the landfill and the airport. This distance is nearly the same as that between sites M and K. The conclusion is based on the assumption that the ground water being drawn toward the proposed well field would move with a velocity similar to that of the local ground water. Also supporting the conclusion was the absence of high concentrations of toxic chemicals except within about 300 ft downgradient from the landfill.

Because the 7.5- and 10-Mgal/d pumpings simulated at the airport did not significantly affect the ground-water flow system in the landfill area, analysis of the flow direction indicated that these pumpings should not affect the current position of the leachate plume. The 20-Mgal/d simulated pumping slightly affected the flow system near the landfill. Therefore, analysis of the flow direction indicated that ground water in some new areas may be affected by leachate if that pumping rate is used.

SUMMARY AND CONCLUSIONS

The aquifer system in northwest Elkhart County consists of an extensive unconsolidated outwash deposit averaging 175 ft in thickness. This deposit is mainly composed of coarse sand and gravel. A layer of silt and clay averaging 20 ft in thickness is interbedded in the sand and gravel. The silt and clay bed, where present, confines the underlying aquifer. Where the bed is absent, as in the vicinity of the Himco landfill and the Elkhart Municipal Airport, the entire aquifer is unconfined. Average hydraulic conductivities of the sand, and the sand and gravel, are 80 and 400 ft/d, respectively. The outwash system extends beneath a till plain in the southeast part of the study area. Both unconfined and confined aquifers underlie till.

Flow in the aquifers is generally toward the St. Joseph River. Vertical head gradients are minimal, except in areas near the major streams, where upward gradients exist.

A quasi-three-dimensional ground-water flow model was constructed to simulate flow in the Elkhart area. The model was calibrated to the best match of measured water levels and seepage data. Sensitivity analyses were done to determine which areas of the model were sensitive to parameter adjustments. The model was most sensitive to the vertical hydraulic conductivity of the streambed for certain stretches of the St. Joseph and Elkhart Rivers and the transmissivities of the unconfined and the confined aquifers in areas near these rivers.

Pumpings of 7.5, 10, and 20 Mgal/d were simulated for a well field proposed for a site at the airport. Resulting drawdowns in the vicinity of the pumping center in all simulations were a small fraction of the total saturated thickness of the aquifer. Model-simulated streamflow reduction in Christiana Creek for these pumpings were 9, 12, and 23 percent of base flow, respectively. The model also showed that the 7.5- and 10-Mgal/d pumpings simulated would not alter the flow pattern enough to cause ground-water flow from the landfill to the vicinity of the pumping center. Results from the 20-Mgal/d simulation indicated that a small part of the flow to the pumping center at the airport may come from the landfill area.

Two additional 10-Mgal/d simulations of pumping at the airport were done. In one, the transmissivity of the aquifers was reduced to one-tenth the calibrated value, and, in the other, the streambed leakance of Christiana Creek was reduced by the same factor. Results from these two simulations showed that the flow pattern would still not be altered enough to cause ground-water flow from the landfill area to the airport. A transient simulation showed that equilibrium conditions would be reached in approximately 600 days of 10-Mgal/d pumping.

An analysis of variance demonstrated that site location was the factor that significantly affected the results of the greatest number of water-quality parameters. Following in order from most influential to least were: the interaction of site type and aquifer depth, aquifer depth, site type, and sampling season.

The general quality of ground water included slightly basic pH (7-8), an average hardness >200 mg/L as $CaCO_3$, an average alkalinity >150 mg/L as $CaCO_3$/L, and, for more than 90 percent of all analyses, a calcium bicarbonate water type. Water in shallow wells generally differed from water in deep wells in that the redox potential of the water in shallow wells represented oxidizing conditions, whereas, in deep wells the redox potential of the water represented mostly reducing conditions. National Interim Primary Drinking Water Regulations were exceeded for only one inorganic parameter, nitrate. The National Secondary Drinking Water Regulations limits for dissolved solids, iron, and manganese were surpassed in several county wells.

The industrial-park well water had the same general inorganic quality characteristics as the county well water. The investigation of the organic water quality of these areas showed that base/neutral extractable organic concentrations were no greater than trace amounts and that acid-extractable (phenolic) organic concentrations were greater than limits suggested by USEPA in water from only two wells. Eight volatile organic compounds were detected: trichloroethylene, trichlorofluoromethane, methylene chloride, toluene, 1,1,1-trichloroethane, 1,1-dichloroethane, chloroform, and 1,2-dichloropropane. Volatile organic compounds were detected in water from only eight wells localized in the northeast part of the industrial-park and the adjacent residential area. Health-effects advisory limits recommended by USEPA for

7-day and chronic exposure to trichloroethylene were exceeded in water from four and six of these eight wells, respectively. These results did not define a plume or a source of these compounds.

The deep landfill well water had the same basic quality as the deep county well water, but large differences were noted between the chemical quality of the shallow landfill and shallow county well water, owing to the leachate in the shallow landfill well water. The NIPDWR for nitrate and chromium were exceeded in water from three and one of the 45 wells sampled, respectively. NIPDWR limits for concentrations of dissolved solids, iron, and manganese were surpassed in water from several of the landfill wells.

The water chemistry of the landfill leachate was characterized by no detectable dissolved-oxygen concentrations, highly reducing conditions, and dissolved-solids concentrations 25 times the background concentration. Concentrations of bicarbonate, bromide, chloride, sulfate, ammonia, calcium, magnesium, manganese, potassium, sodium, and dissolved organic carbon were at least 5 times the mean concentrations of these constituents in county well water. Concentrations of most constituents decreased with distance downgradient from the landfill and reached near-background values at a site 5,600 ft downgradient from it.

Bromide and dissolved-solids concentrations were useful indicators of the landfill leachate in the study. Areal plots of bromide concentrations of the shallow landfill well water showed that the landfill was the source of the leachate, the plume had not spread out much farther laterally than the original width of the landfill, and the leachate was present between 3,100 ft and 5,600 ft downgradient from the landfill. The length of the plume was estimated to be 5,500 ft. Bromide and dissolved-solids concentrations plotted in cross section showed that the leachate plume was sinking in the shallow aquifer downgradient from the landfill and that the leachate was present in deep wells under and immediately downgradient from the landfill.

The conservative constituent bromide was attenuated by dispersion, advection, and diffusion within 5,600 ft downgradient from the landfill, whereas the nonconservative constituents, calcium and sulfate, were attenuated by physical, chemical, and microbiological processes within much shorter distances. Thus, even if pumping at the proposed well field would draw water from the landfill area, attenuation would probably reduce the high concentrations of most leachate constituents by the time the water reached the airport site. Because the 7.5- and 10-Mgal/d simulated pumpings did not significantly affect the ground-water flow system in the landfill area, the current position of the leachate plume probably will not change. However, because the 20-Mgal/d pumping did slightly affect the flow system near the landfill, some new areas of leachate-influenced ground water may result from this pumping.

REFERENCES

Bennett, G. D., 1976, Introduction to ground-water hydraulics, a programmed text for self-instruction: U.S. Geological Survey Techniques of Water-Resources Investigations, book 3, chap. B2, 172 p.

Boggess, D. H., 1975, Effects of a landfill on ground-water quality: U.S. Geological Survey Open-File Report 75-594, 40 p.

Brown, R. H., 1963, Estimating the transmissibility of an artesian aquifer from the specific capacity of a well, in Bentall, Ray, Methods of determining permeability, transmissibility, and drawdown: U.S. Geological Survey Water-Supply Paper 1536-I, p. 336-338.

Burger, A. M., Keller, S. J., and Wayne, W. J., 1966, Map showing bedrock topography of northern Indiana: Indiana Department of Natural Resources, Geological Survey Division.

Clark, T. P., and Piskin, R. 1977, Chemical quality and indicator parameters for monitoring landfill leachate in Illinois: Environmental Geology, v. 1, p. 329-339.

Duncan, D. B., 1955, Multiple range and multiple F tests: Biometrics, v. 11, p. 1-42.

Elkhart Chamber of Commerce, 1979, Pertinent Information--Elkhart, Indiana: Elkhart, Ind., Elkhart Chamber of Commerce, 4 p.

Freeze, R. A., and Cherry, J. A., 1979, Groundwater: Englewood Cliffs, New Jersey, Prentice Hall, Inc., 604 p.

Gillies, D. C., 1976, Availability of ground water near Carmel, Hamilton County, Indiana: U.S. Geological Survey Water-Resources Investigations 76-46, 27 p.

Goerlitz, D. F., and Brown, Eugene, 1972, Methods for analysis of organic substances in water: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A3, 40 p.

Helwig, J. T., and Council, K. A., eds., 1979, SAS Users' Guide, 1979 edition: Raleigh, N.C., SAS Institute, Inc., 494 p.

Horner, R. G., 1976, Statistical summaries of Indiana streamflow data: U.S. Geological Survey Water-Resources Investigations 35-75, 526 p.

Hughes, J. L., Eccles, L. A., and Malcolm, R. L., 1974, Dissolved organic carbon (DOC), an index of organic contamination in ground water near Barstow, Calif.: Ground Water, v. 12, no. 5, p. 283-290.

Jenne, E. A., 1968, Controls on Mn, Fe, Co, Ni, Cu, and Zn concentrations in soils and water. The significant role of hydrous Mn and Fe oxides, in Trace Inorganics: Washington, D.C., American Chemical Society, Advances in Chemistry Series 73, p. 337-387.

Jester, W. A., Raupach, D. C., and Haaser, F. G., 1977, A comparison of the bromide ion neutron activatable tracer with tritiated water as ground-water tracers, in Third , 1977, International Conference on Nuclear Methods in Environmental and Energy Research, University of Missouri, Columbia, Mo., October 1977: U.S. Department of Energy, p. 253-264. Available from National Technical Information Service, Springfield, Va. (order no. CONF-771072).

Jester, W. A., and Uhler, K. A., 1974, Identification and evaluation of water tracers amenable to post-sampling neutron activation analysis: Pennsylvania State University Institute for Research on Land and Water Resources, Research Publication no. 85, 92 p.

Johnson, G. H., and Keller, S. J., 1972, Geologic map of the 1° x 2° Fort Wayne Quadrangle, Indiana, Ohio, and Michigan, showing bedrock and unconsolidated deposits: Indiana Department of Natural Resources, Geological Survey Division.

Keith, L. H., and Telliard, W. A., 1979, Priority pollutants I. A perspective view: Environmental Science and Technology, v. 13, no. 4, p. 416-423.

Kimmel, G. E., and Braids, O. C., 1980, Leachate plumes in ground water from Babylon and Islip landfills, Long Island, New York: U.S. Geological Survey Professional Paper 1085, 38 p.

Leenheer, J. A., Malcolm, R. L., McKinley, P. W., and Eccles, L. A., 1974, Occurrence of dissolved organic carbon in selected ground-water samples in the United States: Journal of Research of the U.S. Geological Survey, v. 2, no. 3, p. 361-369.

Malcolm, R. L., and Leenheer, J. A., 1973, The usefulness of organic carbon parameters in water-quality investigations, in Institute of Environmental Sciences Proceedings, Anaheim, Calif., April 1973: Mt. Prospect, Ill., Institute of Environmental Sciences, p. 336-340.

Marie, J. R., 1975, Hydraulic characteristics and water-supply potential of the aquifers in the vicinity of the wastewater treatment plant, South Bend, Indiana, U.S. Geological Water-Resources Investigation 49-74, 26 p.

Meyer, William, Reussow, J. P., and Gillies, D. C., 1975, Availability of ground water in Marion County, Indiana, with a section on water quality by W. J. Shampine: U.S. Geological Survey Open-File Report 75-312, 87 p.

National Oceanic and Atmospheric Administration, 1978, Climatological data, Indiana: Raleigh, N.C., National Oceanic and Atmospheric Administration, Environmental Data and Information Service, Annual Summary 1978, v. 83, no. 13, 13 p.

National Research Council, 1980, Drinking Water and Health, v. 3: Washington, D.C., National Academy Press, 415 p.

Norris, S. E., 1962, Permeability of glacial till, in Geological Survey Research 1962: U.S. Geological Survey Professional Paper 450-E, p. E150-E151.

Plummer, L. N., Jones, B. F., and Truesdell, A. H., 1976, WATEQF--A Fortran IV version of WATEQ, a computer program for calculating chemical equilibrium of natural waters: U.S. Geological Survey Water-Resources Investigations 76-13, 63 p.

Schneider, A. F., and Keller, S. J., 1970, Geologic map of the 1°x2° Chicago Quadrangle, Indiana, Illinois, and Michigan, showing bedrock and unconsolidated deposits: Indiana Department of Natural Resources, Geological Survey Division.

Skougstad, M. W., and others, eds., 1979, Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A1, 626 p.

Stallman, R. W., 1965, Effects of water table conditions on water level changes near pumping wells: Water Resources Research, v. 1, no. 2, p. 295-312.

Stumm, W., and Morgan, J. J., 1970, Aquatic chemistry: New York, Wiley-Interscience, 583 p.

Theis, C. V., 1963, Estimating the transmissibility of a water-table aquifer from the specific capacity of a well, in Bentall, Ray, Methods of determining permeability, transmissibility, and drawdowns: U.S. Geological Survey Water-Supply Paper 1536-I, p. 332-336.

Trescott, P. C., 1975, Documentation of finite-difference model for simulation of three-dimensional ground-water flow: U.S. Geological Survey Open-File Report 75-438, 129 p.

U.S. Environmental Protection Agency, 1975, National interim primary drinking water regulations: Federal Register, v. 40, no. 248, p. 59566-59588.

______1976, Quality criteria for water: Washington, D.C., U.S. Government Printing Office, EPA-440/9-76-023, p. 347-349.

______ 1977, Procedures manual for ground-water monitoring at solid waste disposal facilities: U.S. Government Printing Office, EPA-530/SW-611, 269 p.

U.S. Environmental Protection Agency, 1979a, National secondary drinking water regulations: Federal Register, v. 44, no. 140, p. 52195-42202.

______ 1979b, National interim primary drinking water regulations; control of trihalomethanes in drinking water: Federal Register, v. 44, no. 231, p. 68624-68707.

______ 1979c, Part III--Guidelines for establishing test procedures for the analysis of pollutants; proposed regulations: Federal Register, v. 44, no. 233, p. 69463-69575.

Wood, W. W., 1976, Guidelines for collection and field analysis of groundwater samples for selected unstable constituents: U.S. Geological Survey Techniques of Water-Resources Investigations, book 1, chap. D2, 24 p.

Zar, J. H., 1974, Biostatistical analysis: Englewood Cliffs, N.J., Prentice-Hall, Inc., 620 p.

Zenone, C., Donaldson, D. E., and Grunwaldt, J. J., 1975, Ground-water quality beneath solid-waste disposal sites at Anchorage, Alaska: Ground Water, v. 13, no. 2, p. 182-190.