INDEX

EXHIBIT A 2000
Equus Beds Recharge Project: public meeting list

EXHIBIT B 2001
Equus Beds Recharge Project: Federal and State Agency meeting list

EXHIBIT C 2002
Dr. Herman Bouwer letter to David Stous – August 10, 2000

EXHIBIT D 2003
Changes in Ground-Water Levels and Storage in the Wichita Well Field Area, South - Central Kansas, 1940 – 98; Water – Resources Investigation Report 98-4141
USGS, Lawrence, Kansas 1998

EXHIBIT E 2004
Status of Ground-Water Levels and Storage Volume in the Equus Beds Aquifer Near Wichita, Kansas, January 2000 – January 2003; Water – Resources Investigations Report 03-4298
USGS, Lawrence, Kansas, 2004

EXHIBIT F 2005
Baseline Water quality and Preliminary Effects of Artificial Recharge on Ground Water, Soutb – Central Kansas, 1995-98; Water Resources Investigations Report 99-4250
USGS, Lawrence, Kansas, 1999

EXHIBIT G 2006
Effects of Artificial Recharge on Water Quality in the Equus Beds Aquifer, South – Central Kansas, 1995-2000; USGS Fact Sheet 096-01
USGS, Lawrence, Kansas, November 2001

EXHIBIT H 2007
Hydrologic and Chemical Interaction of the Arkansas River and the Equus Beds Aquifer Between Hutchinson and Wichita, South- Central Kansas; Water – Resources Investigations Report 95-4191 USGS, Lawrence, Kansas, 1996

EXHIBIT I 2008
Burrton IGUCA Remediation Study, by Burns & McDonnell, 1997
Prepared for Kansas Corporation Commission

EXHIBIT J 2009
Arkansas River Water Management Improvement Study; Modeling of Chloride Transport in the Equus Beds Aquifer Technical Report
U. S. Bureau of Reclamation Report, November 1993

EXHIBIT K 2010
Hydrobiological Monitoring Plan for the Integrated Local Water Supply Project
Prepared for the City of Wichita, by Burns & McDonnell, October 2004

EXHIBIT A

Equus Beds Recharge Public Meetings

- Dec. 2, 1993 – Halstead H.S.
- July 26, 1994 – Halstead H.S.
- Sept. 26, 1995 – Halstead H.S.
- July 9, 1996 – GMD Board
- Oct. 28, 1996 – Halstead H.S.
- Opened Bids on Demo. Project- Dec. 13, 1996
- Feb. 4, 1997, Sedgwick Planning Commission
- Feb. 10, 1997 - Halstead City Council
- March 10, 1997 – Halstead City Council
- May 22, 1997 – Halstead Chamber of Commerce
- October 11, 1997 - Public Tour and Lunch (used Farm Bureau mailing list)
- October 20, 1997 – ILWSP EIS Public Scoping Meeting, Wichita
- October 21, 1997 – ILWSP EIS Public Scoping Meeting, Cheney
- October 22, 1997 – ILWSP EIS Public Scoping Meeting, Halstead
- April 21, 1998 – Halstead H.S.
- July 13, 1998 – Halstead City Council
- Dec. 2, 1998 – Halstead H.S.
- Nov. 22, 1999 – Sedgwick H.S.
- Nov. 2, 2000 – Halstead H.S.
- Feb. 4, 2002 – Halstead Lions Club
- April 23, 2002 – Public Meeting on Draft ILWSP EIS, Halstead HS
- April 24, 2002 – Public Meeting on Draft ILWSP EIS, Wichita City Hall
- August 8, 2002 – Clarke Dixon's
- August 28, 2002 – Bentley City Hall
- Nov. 26, 2002 – Halstead H.S.- GMD Information Fair
- Feb. 27, 2003 – GMD Board Meeting

EXHIBIT B

Equus Beds Recharge Project
Federal and State Agency Meetings

Update Meetings:

- October 6, 1994 · Topeka
- August 23, 1995 – Topeka
- April 17, 1996 – Topeka
- September 12, 1996 – Topeka
- December 3, 1997 – Topeka

EIS:

- November 1997 – Wichita
- October 1997 – Wichita
- July 1998 – Wichita
- May 1998 – Wichita
- April 1999 – Wichita
- December 1999 – Wichita

HBMP:

- November 14, 2003 – Wichita
- December 2, 2003 – Wichita
- January 20, 2004 – Wichita
- May 12, 2004 – Wichita

EIS COOPERATING AGENCIES

U.S. Bureau of Reclamation
U.S. Fish and Wildlife Service
U.S. Environmental Protection Agency
U.S. Geological Survey
Groundwater Management District No. 2

Kansas Water Office
Kansas Department of Health & Environment
Kansas Department of Wildlife & Parks
Kansas Department of Agriculture, Division of Water Resources

EXHIBIT C

HERMAN BOUWER, Ph.D.,P.E.
GROUNDWATER CONSULTANT

338 La Diosa Drive
Tempe, Arizona 85282
(602) 379-4356
(480) 967-0236
FAX (602) 379-4355
E-mail: boutemcons@aol.com

Artificial recharge
Wastewater reuse
Water conservation
Flow in vadose zone
Water quality for irrigation
Surface-subsurface water relations
Pollution by agricultural chemicals

August 10, 2000

David H. Stous, P.E., P.G.
Chief Water Supply Hydrogeologist
Burns & McDonnell
9400 Ward Parkway
Kansas City, Missouri 64114

Dear David:

I enjoyed our trip to the Equus Beds Aquifer Recharge, Storage and Recovery Project of the City of Wichita on 8 and 9 August. The strong technical points of this project are the characterization of the hydrogeology, the pre-treatment of the water through bank filtration or in-plant treatment (coagulation, sedimentation and PAC addition), and the use of various recharge methods (basins, wells, trench). Also, the whole project has "good engineering" written all over it!

Sincerely,

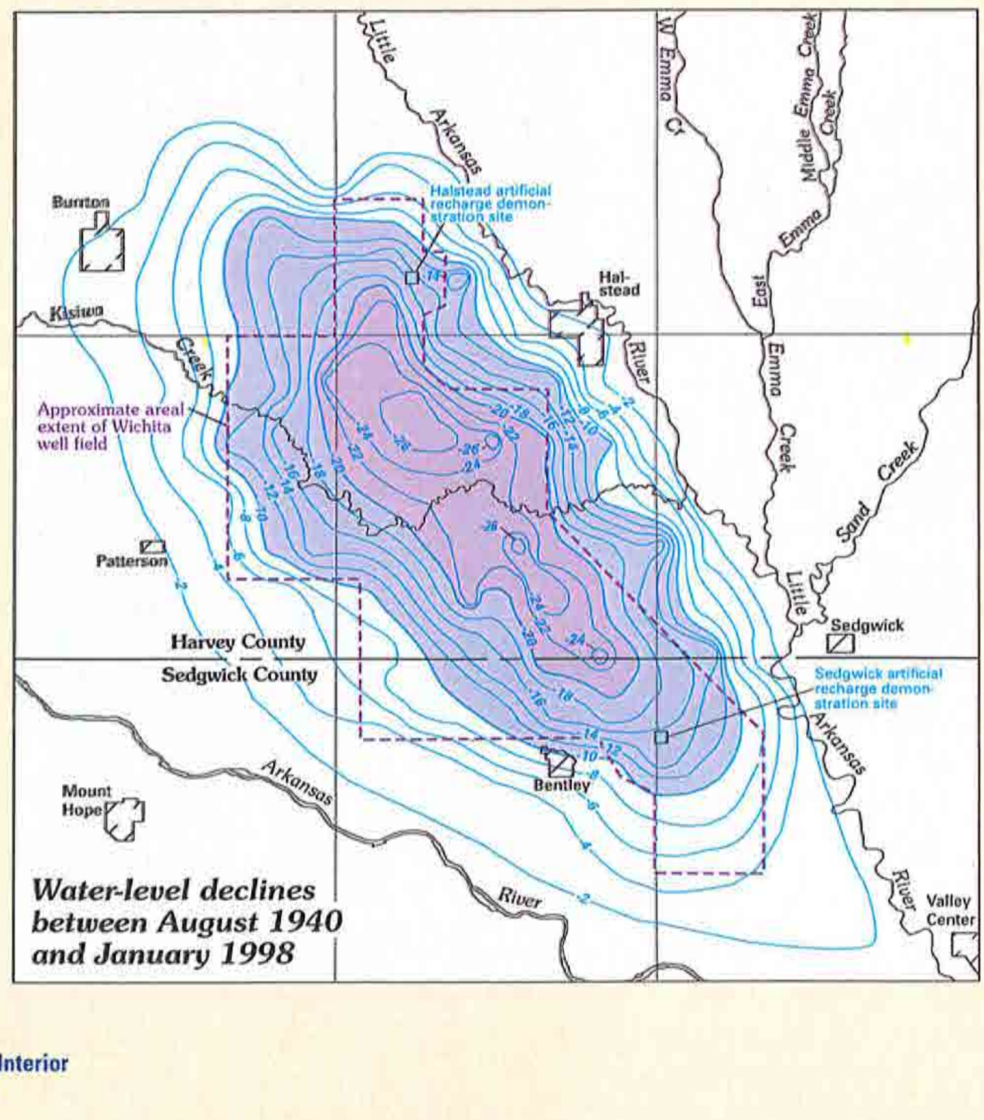
Herman Bouwer

EXHIBIT D

Prepared in cooperation with the
CITY OF WICHITA, KANSAS

Changes in Ground-Water Levels and Storage in the Wichita Well Field Area, South-Central Kansas, 1940–98

Water-Resources Investigations Report 98–4141



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

002003

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

Prepared in cooperation with the
CITY OF WICHITA, KANSAS

Changes in Ground-Water Levels and Storage in the Wichita Well Field Area, South-Central Kansas, 1940–98

By Walter R. Aucott and Nathan C. Myers

Water-Resources Investigations Report 98–4141

Lawrence, Kansas
1998

U.S. Department of the Interior

Bruce Babbitt, Secretary

U.S. Geological Survey

Thomas J. Casadevall, Acting Director

For additional information write to:

District Chief
U.S. Geological Survey
4821 Quail Crest Place
Lawrence, KS 66049–3839

Copies of this report can be purchased from:

U.S. Geological Survey
Information Services
Box 25286
Federal Center
Denver, CO 80225–0826

CONTENTS

Abstract 1
Introduction 1
  Background 1
  Purpose and Scope 3
  Description of Study Area 3
  Previous Studies 3
Geology and Ground Water 4
Changes in Ground-Water Levels and Storage 4
Summary 19
Selected References 20

FIGURES

1. Maps showing location of study area and Wichita well field 2
2. Graph showing mean annual, 1940–96 annual mean, and 5-year moving average precipitation at Hutchinson, Mount Hope, Newton, Sedgwick, and Wichita, Kansas 3
3. Generalized geologic section 5
4. Map showing water-level altitudes for August 1940 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 6
5. Graph showing municipal water use by the city of Wichita from Wichita well field and Cheney Reservoir and agricultural ground-water use in study area, 1940–96 7
6. Hydrographs for selected wells and *Equus* Beds aquifer storage-volume depletions in study area, 1938–97 8

7–10. Maps showing water-level altitudes for:
  7. January 1957 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 9
  8. January 1970 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 10
  9. January 1993 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 11
  10. January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 12

11–15. Maps showing water-level change from:
  11. August 1940 to January 1957 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 13
  12. August 1940 to January 1970 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 14
  13. August 1940 to January 1993 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 15
  14. August 1940 to January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 16
  15. January 1993 to January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field 17

TABLE

1. Storage-volume changes in the *Equus* Beds aquifer in the Wichita well field area, August 1940 to January 1998 18

CONVERSION FACTORS, ABBREVIATIONS, AND VERTICAL DATUM

Multiply	By	To obtain
acre-foot (acre-ft)	1,233	cubic meter
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year
degree Fahrenheit (°F)	([1])	degree Celsius (°C)
foot (ft)	0.3048	meter
inch (in.)	25.4	millimeter
inch per year (in/yr)	25.4	millimeter per year
mile (mi)	1.609	kilometer
square mile (mi^2)	2.590	square kilometer

[1] Temperature can be converted to degrees Celsius (°C) or degrees Fahrenheit (°F) by the equations:

$$°C = 5/9\ (°F-32)$$
$$°F = 9/5\ (°C) + 32.$$

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

Changes in Ground-Water Levels and Storage in the Wichita Well Field Area, South-Central Kansas, 1940–98

By Walter R. Aucott and Nathan C. Myers

Abstract

The Wichita well field was developed in the *Equus* Beds aquifer northwest of Wichita, Kansas, to supply water to the city. On September 1, 1940, pumping began from 25 wells in the well field. City pumpage increased from then until the early 1950's and between the late 1970's and early 1990's. Since then, city withdrawals from the well field have decreased as withdrawals increased from Cheney Reservoir, the other major water-supply source for the city. Nearby agricultural withdrawals increased substantially in the 1970's and 1980's and, in the 1990's, was similar in magnitude to city withdrawals in the study area although more seasonal and variable in response to changing climatic conditions.

Ground-water withdrawals in the vicinity of the well field caused a large area of water-level declines to develop in the *Equus* Beds aquifer. Water levels declined from 1940 through the 1950's drought, stabilized in the 1960's and 1970's, continued to decline between the late 1970's and the 1988–92 drought, and reached their maximum to date of as much as 40 feet or more during 1991–93. Loss of ground water in storage since August 1940 followed a pattern similar to water-level declines, with a maximum loss of storage of 255,000 acre-feet reached in January 1993. Water-level declines encompassed an area of about 190 square miles at their maximum in January 1993 and extended from the Arkansas River to the Little Arkansas River in the vicinity of Halstead and Sedgwick. Ground-water levels have since recovered more than 10 feet in some areas and aquifer storage replenished by 79,000 acre-feet between 1993 and 1998 primarily as a result of decreased city withdrawals.

INTRODUCTION

Background

The Wichita well field was developed in the *Equus* Beds aquifer to supply water to the city of Wichita in south-central Kansas (fig. 1). On September 1, 1940, pumping began from 25 wells in the well field (Stramel, 1956). By 1959, there were 55 wells in use in the well field (Stramel, 1967). Ground-water pumping from the well field has caused water levels to decline over a large area. Much of the water-level decline occurred from 1940 to early 1957 (Stramel, 1967). Ground-water withdrawals for irrigation in the Wichita well field area also increased substantially in the 1970's and 1980's and contributed to the water-level decline (Myers and others, 1996). Although most of the water-level declines can be attributed to ground-water withdrawals, climatic conditions and thus recharge to the *Equus* Beds aquifer also have affected ground-water levels.

In 1965, the city of Wichita began using water from Cheney Reservoir (Stramel, 1967) in addition to water from the *Equus* Beds aquifer. Since 1995 (Warren and others, 1995), the city of Wichita, in cooperation with Equus Beds Groundwater Management District No. 2, the Bureau of Reclamation, U.S. Geo-

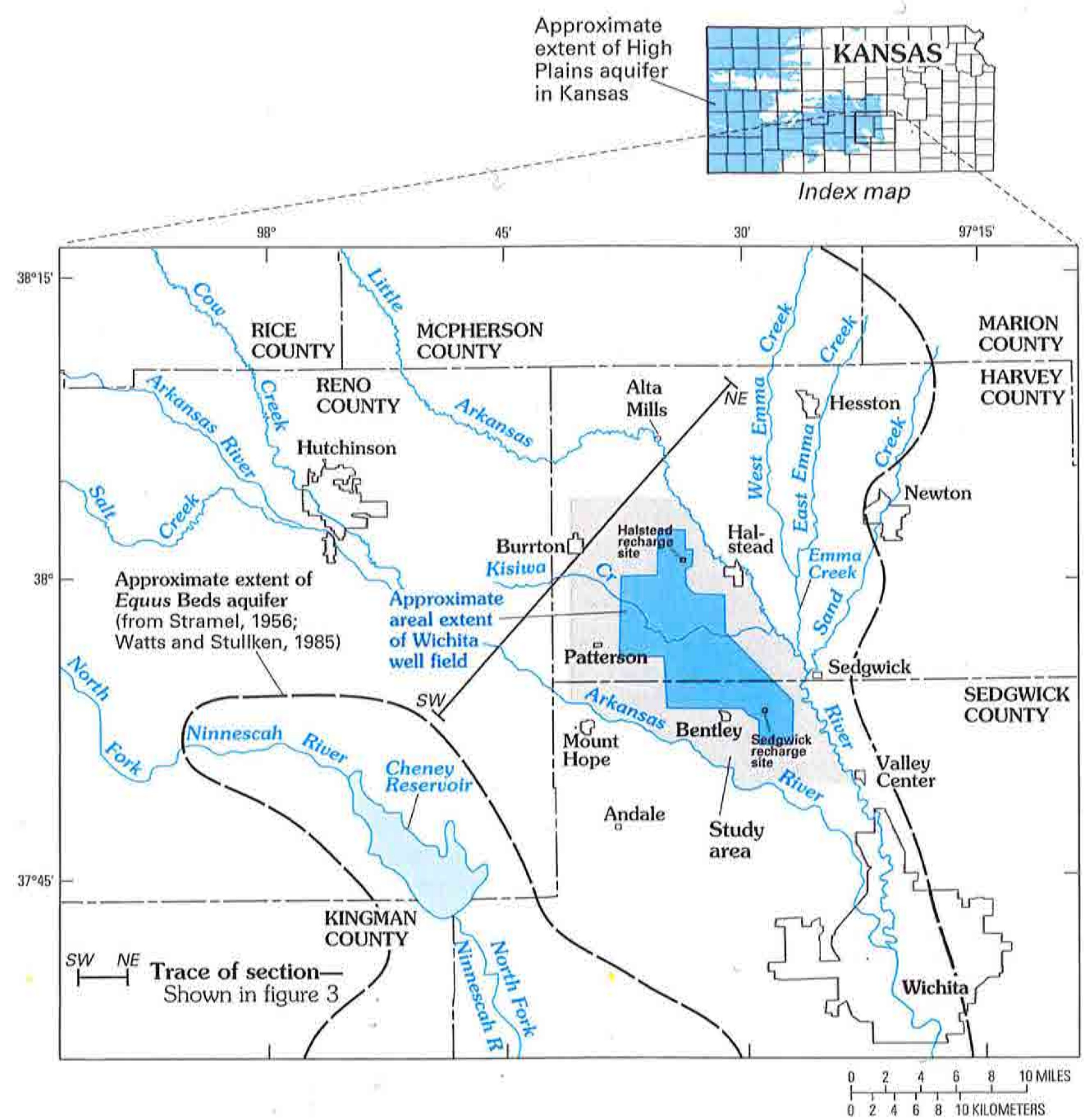


Figure 1. Location of study area and Wichita well field.

logical Survey, U.S. Environmental Protection Agency, Kansas State agencies, Burns and McDonnell Engineering Consultants, and Mid-Kansas Engineering Consultants, has been investigating the possibility of artificial ground-water recharge in the well field to meet future needs and to protect the aquifer from saltwater intrusion from natural and anthropogenic sources to the west. Because of the social and economic importance of ground-water resources and because of the changes that artificial recharge is expected to bring to the aquifer, the city of Wichita entered into a cooperative agreement with the U.S. Geological Survey (USGS) to document historical hydrologic conditions, their changes and causes, in the Wichita well field area (study area shown in fig. 1); to develop a baseline condition for evaluating the effects of artificial recharge on ground-water levels in the aquifer; and to annually review changes in the hydrologic system. The USGS and the city of Wichita have worked cooperatively for many years in evaluating the ground-water system and the interaction with streams in the area to further the understanding of the entire hydrologic system and to provide information to improve local decisionmaking.

Purpose and Scope

The purpose of this report is to present changes in ground-water levels and storage in the Wichita well field area from August 1940 to January 1998. Maps of ground-water levels and water-level changes were selected for significant periods in time. Ground-water levels presented as hydrographs were selected to show water-level fluctuations in various parts of the well field area.

Description of Study Area

The study area (fig. 1) includes 165 mi^2 and is located in Harvey and Sedgwick Counties, northwest of Wichita, Kansas. It is bounded to the southwest by the Arkansas River and to the northeast by the Little Arkansas River. The Wichita well field covers 55 mi^2 and is located within the study area.

The study area lies in the Arkansas River section of the Central Lowlands physiographic province (Schoewe, 1949). There is very little topographic relief in the study area. For the most part, the land surface slopes gently toward the major streams in the area.

South-central Kansas has a continental climate and is characterized by large variations in seasonal temperatures, moderate precipitation, and windy conditions. Seasonal temperatures range from daily averages of 30.6 ^{o}F in January to 79.6 ^{o}F in July for 1961–90 (National Oceanic and Atmospheric Administration, 1996). The mean annual precipitation at weather stations near the study area (at Hutchinson, Mount Hope, Newton, Sedgwick, and Wichita) is 31.06 in. for 1940–96 (National Oceanic and Atmospheric Administration, 1997) (fig. 2). Most of this precipitation occurs during spring and summer.

Previous Studies

Water-level data have been collected by the city of Wichita in the study area quarterly since 1940 and are on file with the USGS in Lawrence, Kansas. Water-level data have been collected in the *Equus* Beds aquifer by Equus Beds Groundwater Management District No. 2, and water-level change maps have been published by the district (Equus Beds Groundwater Management District No. 2, 1995). Annual water-level

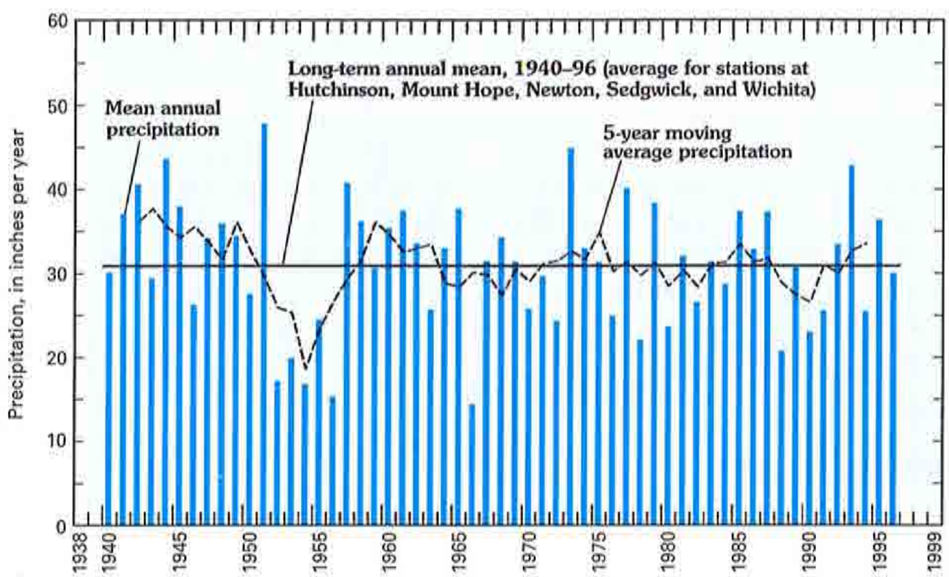


Figure 2. Mean annual, 1940–96 annual mean, and 5-year moving average precipitation at Hutchinson, Mount Hope, Newton, Sedgwick, and Wichita, Kansas (source of data: National Oceanic and Atmospheric Administration, 1997)

data for the High Plains aquifer (fig. 1), which includes the *Equus* Beds aquifer, have been collected by the Kansas Department of Agriculture (Division of Water Resources), USGS, and the Kansas Geological Survey, and are on file with the USGS in Lawrence, Kansas, and have been compiled by the Kansas Geological Survey (Schloss and others, 1997) and mapped by McGuire and Sharpe (1997).

Williams and Lohman (1949, plate 33) show potentiometric-surface maps for the Wichita well field area for June 1, 1940, through October 1, 1944. Water levels from June 1940 were used as the basis for calculating water-level declines in observation wells in the Wichita well field area. Williams and Lohman (1949, plate 34) also show water-level decline maps for the Wichita well field area for April 15, 1941, through October 1, 1944.

Stramel (1956) calculated the net decrease in ground-water storage in the Wichita well field area, which is the net result of natural discharge, discharge by pumping, and natural recharge. Assuming a specific-yield value of 0.20, Stramel (1956) calculated that the net storage decrease from August 30, 1940, to January 1, 1944, was 33,390 acre-ft; to January 1, 1948, was 49,800 acre-ft; to January 1, 1952, was 50,200 acre-ft; and to January 1, 1955, was 111,000 acre-ft.

Stramel (1967) calculated storage decreases for eight dates from December 31, 1957, through December 31, 1965. Stramel's (1967) maximum net storage decrease was 114,225 acre-ft calculated for December 31, 1964, although he did not compute a storage decrease for the December 31, 1956, map, which had the largest water-level declines up to that time. A net storage decrease 1 year later was calculated at 95,800 acre-ft, and this smaller net storage decrease was attributed to the use of water from Cheney Reservoir to supplement water from the Wichita well field (Stramel, 1967). Ross and others (1997) noted an increase in water levels in the *Equus* Beds aquifer from 1993–97 and attributed them largely to decreased withdrawals from the Wichita well field.

GEOLOGY AND GROUND WATER

Quaternary deposits, primarily alluvial, occur throughout the study area. These alluvial deposits, known as the *Equus* beds, are as much as 250 ft thick in the study area (fig. 3) (Lane and Miller, 1965a). The *Equus* beds consist primarily of sand and gravel interbedded with clay or silt but locally may consist primarily of clay with thin sand and gravel layers (Lane and Miller, 1965a; Myers and others, 1996). The middle part of the deposits generally has more fine-grained material than the lower and upper parts (Lane and Miller, 1965b; Myers and others, 1996).

The Wellington Formation underlies the Quaternary alluvial deposits and is about 700 ft thick (Bayne, 1956). The Wellington Formation consists of three members—the lower anhydrite member, about 200 ft thick; the Hutchinson Salt Member, about 300 ft thick; and the upper shale member, about 200 ft thick (Bayne, 1956). Dissolution of the Hutchinson Salt Member has resulted in subsidence of the overlying upper shale member, formation of low areas in the bedrock surface, and accumulation of alluvial deposits that now comprise the *Equus* beds (fig. 3) (Myers and others, 1996).

The *Equus* Beds aquifer is the easternmost extension of the High Plains aquifer in Kansas (fig. 1). The extent of the *Equus* Beds aquifer is delineated in figure 1 as defined by Stramel (1956) and in the vicinity of Cheney Reservoir by a more recent regional study (Watts and Stullken, 1985). Watts and Stullken (1985) limit the areal extent of the aquifer north and west of Wichita but still include the *Equus* Beds aquifer in all of the study area. The *Equus* beds are an important source of ground water because of the generally shallow depth to the water table, the large saturated thickness, and the generally good quality of water. Near the Arkansas River, the water table may be as little as 10 ft below land surface. Farther from the river and near the Little Arkansas River, the water table is at a greater depth below land surface, depending on the altitude of land surface and the amount of water-level decline that has been caused by ground-water withdrawals. The maximum saturated thickness of the *Equus* Beds aquifer within the study area, almost 250 ft, is near the course of the Arkansas River and corresponds to the lowest areas of the underlying bedrock surface (fig. 3). The Wellington Formation acts as a confining unit underlying the *Equus* Beds aquifer.

CHANGES IN GROUND-WATER LEVELS AND STORAGE

Prior to pumpage from the Wichita well field in September 1940, near-predevelopment conditions existed for the *Equus* Beds aquifer in the study area

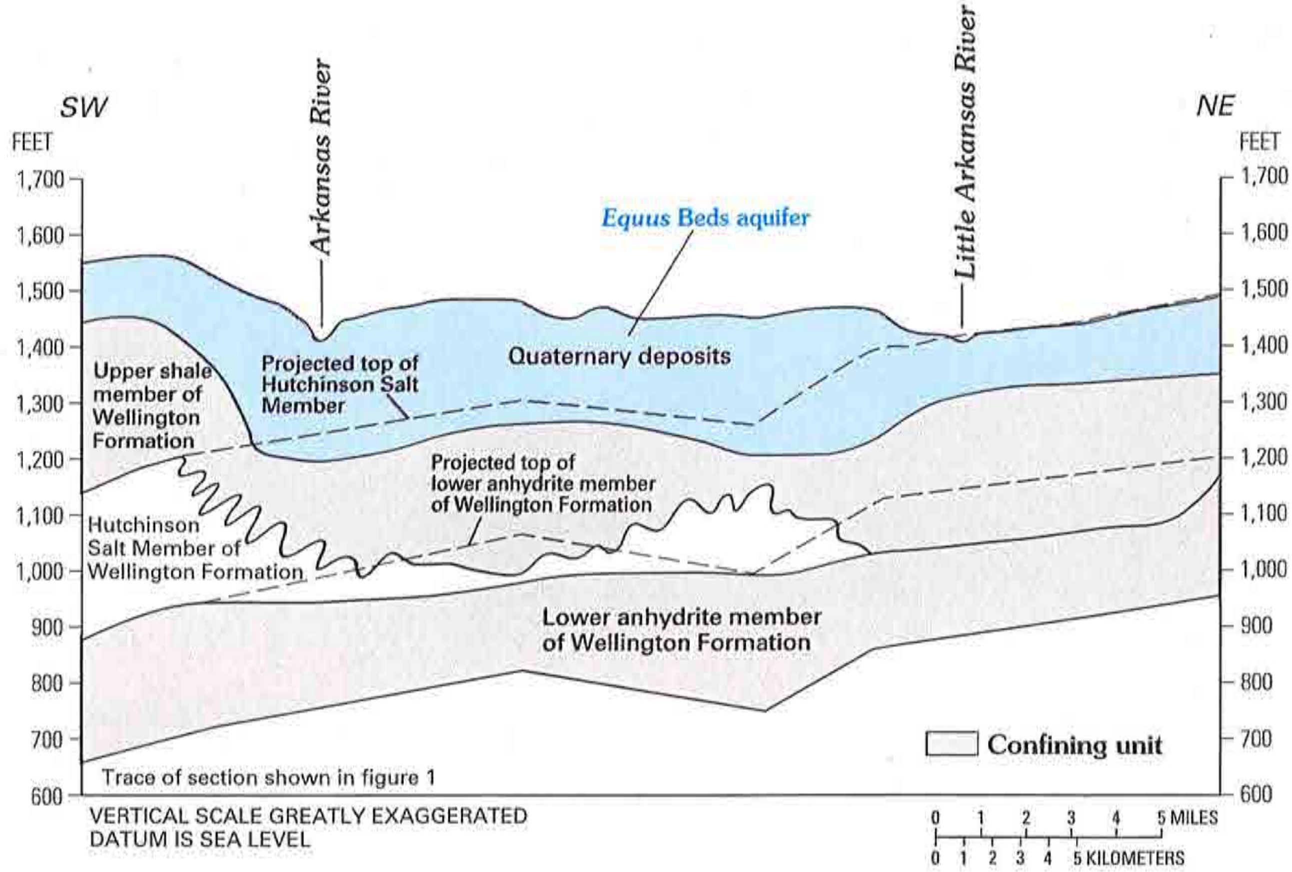


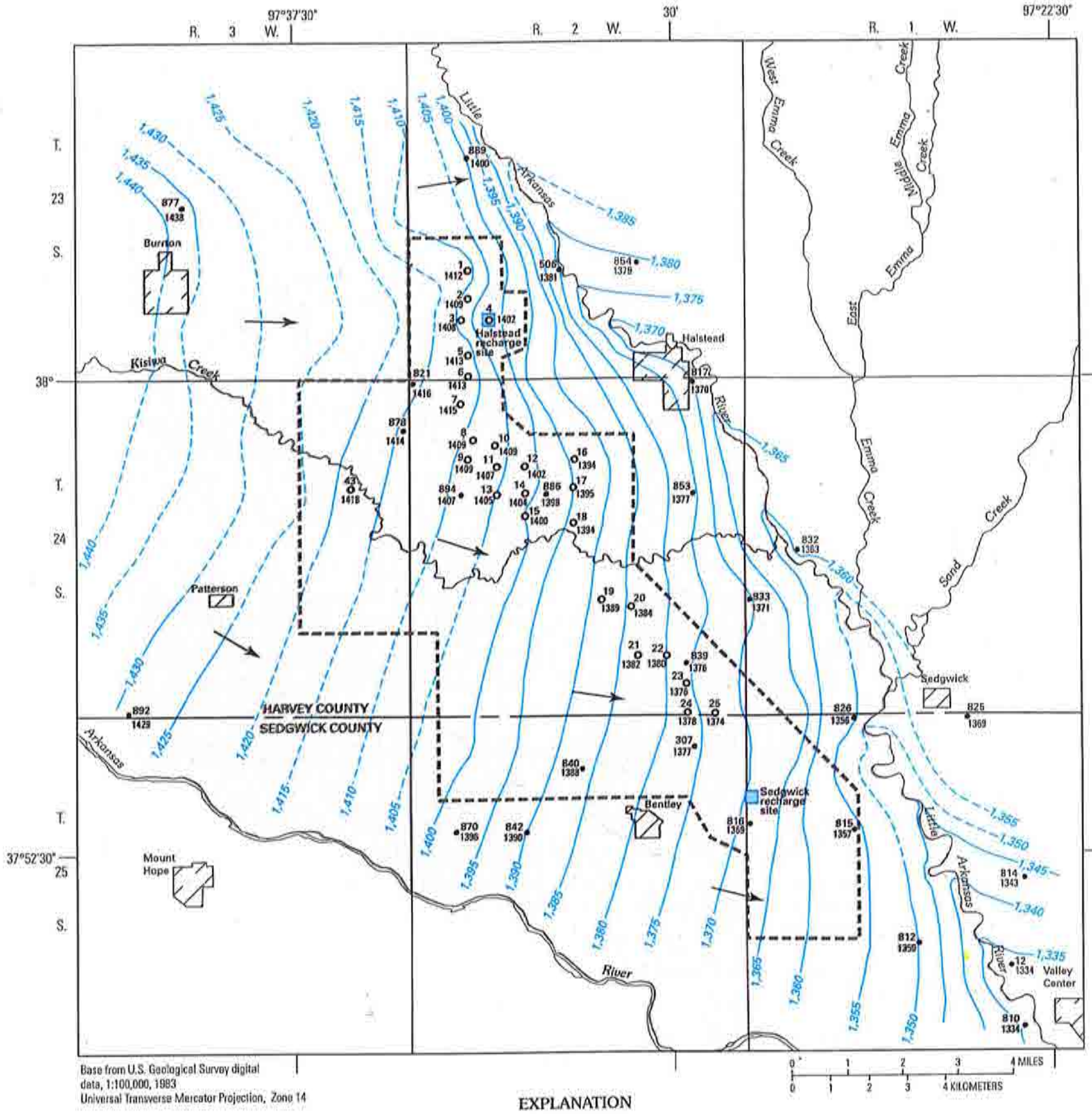
Figure 3. Generalized geologic section (from Leonard and Kleinschmidt, 1976; Myers and others, 1996).

(Williams and Lohman, 1949). Ground water flowed generally west to east and discharged to the Little Arkansas River prior to development, as indicated in figure 4. Water levels in about 50 observation wells measured in August 1940 were used to construct this map; most of these wells were associated with the Wichita well field.

Ground-water-level declines can result from pumpage and from decreased recharge resulting from less-than-average precipitation as well as other factors. Figure 2 indicates annual mean and 5-year moving average precipitation data for long-term stations in the vicinity of the study area. Droughts, such as occurred in 1952–56 and 1988–92, decrease the amount of recharge available and increase the demand for and thus withdrawals of ground water, resulting in increased water-level declines.

Since 1940, ground-water withdrawals for the city of Wichita, and later for agricultural uses, have become significant (fig. 5). City of Wichita withdrawals, which began in September 1940 in the study area, increased steadily into the early 1950's and then were relatively constant until the mid-1970's. City withdrawals increased sporadically from the late 1970's through the early 1990's in response to increased demand and droughts and have since decreased due to the increasing reliance of the city on Cheney Reservoir as a water-supply source. Agricultural withdrawals were relatively small until the early 1970's but have increased subtanstially since (Myers and others, 1996). Agricultural water-use amounts reported prior to the early 1990's are not plotted in figure 5 because of ongoing data-verification considerations.

Extensive information is available to describe hydrologic conditions in the study area. Water-level data have been collected periodically from more than 100 wells by city of Wichita personnel. Data collection began just prior to the beginning of city withdrawals in September 1940, and as well-field development proceeded, water levels in additional wells were measured. Measuring frequency varies from well to well, but most are measured quarterly or annually. Standard ground-water-level measurement techniques similar to USGS methods (Stallman, 1971) have been used by



EXPLANATION

- Study area
- —1,375— Water-level contour—Shows altitude of water level, August 1940. Dashed where approximately located. Contour interval 5 feet. Datum is sea level
- Approximate areal extent of Wichita well field
- Drainage ditch
- Approximate direction of ground-water flow
- Artificial recharge demonstration site
- 1 / 1412 Observation well in vicinity of city of Wichita supply well
- 854 / 1379 Other observation well

Well number shown above symbol; water-level altitude, in feet above sea level, shown below.

Water-level measurements are made by city of Wichita personnel.

Figure 4. Water-level altitudes for August 1940 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

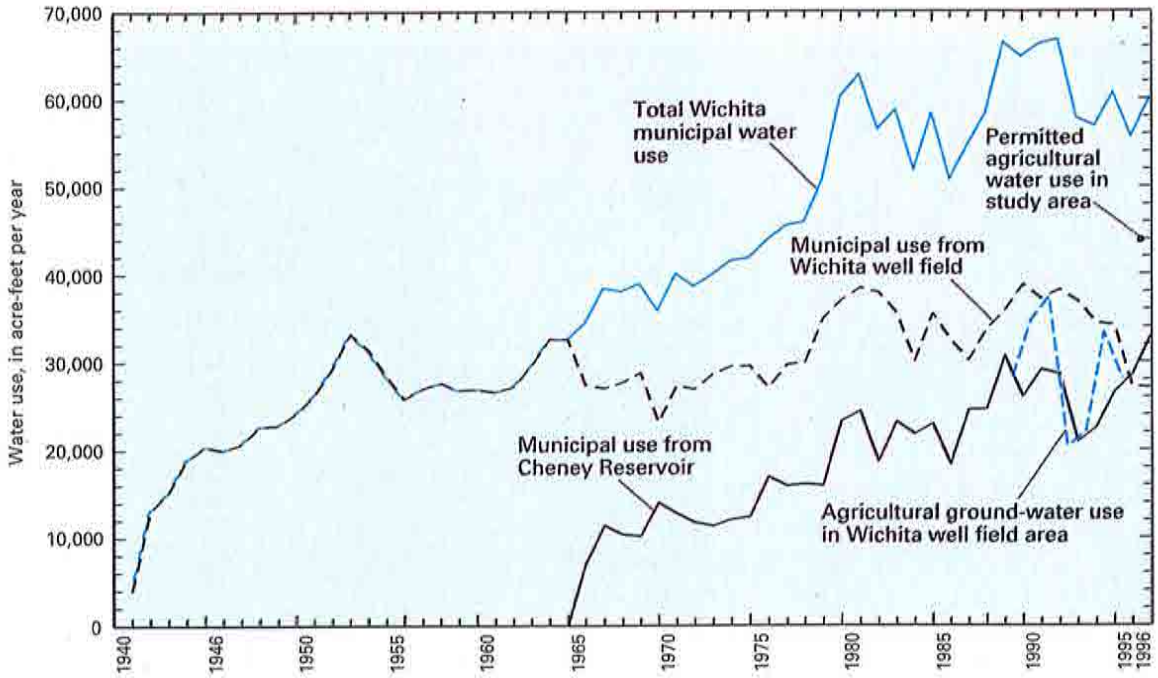


Figure 5. Municipal water use by the city of Wichita from the Wichita well field and Cheney Reservoir and agricultural ground-water use in study area, 1940–96. Data obtained from Gerald T. Blain (City of Wichita, written commun., 1997) and the Kansas Department of Agriculture, Division of Water Resources, and the Kansas Water Office (1990–96).

city personnel. Data are stored by the city in paper and electronic form and by the USGS in electronic form.

The available water-level data were used to plot hydrographs for selected wells for 1938–97 and to map water levels and water-level changes for a variety of time periods. The wells selected (fig. 6) are distributed throughout the study area as indicated in figure 7. The hydrographs for these selected wells reflect a history of hydrologic conditions and responses to ground-water withdrawals within the well field and in surrounding areas. The hydrograph for well 886 is a representative descriptor of water-level changes near the historic center of the city's well field pumpage, whereas the hydrograph for well 307 presents a view of the effects in the southern part of the well field. Hydrographs for wells P29, 104, 810, 3004, and 3039 present hydrologic conditions in areas away from the city well field. The hydrograph for well 104 is particularly noteworthy in its depiction of the effects of irrigation withdrawals.

Water-level altitude maps were constructed for January 1957, 1970, 1993, and 1998 (figs. 7–10). Water-level change maps were constructed to describe changes between August 1940 (predevelopment) and each of these times (figs. 11–14) and for the period January 1993 to January 1998 (fig. 15). These years were selected as representative of broader time periods and trends as depicted in the hydrograph for well 886 (fig. 6) and as discussed later in this report. The month of January was selected for each year to provide comparable conditions with a minimal effect from seasonal factors. The water-level change maps were constructed using August 1940 measurements where they existed, primarily near city wells 1–25, and a few measurements from September 1940 for some wells in the western part of the study area. Where no 1940 measurements existed, values were interpolated from the August 1940 water-level contour map (fig. 4).

Table 1 indicates changes in aquifer storage volume between August 1940 and January 1957, 1970, 1993, and 1998 and between January 1993 and January 1998. Changes in storage were determined from areas inside water-level-change contours for these selected time periods and were computed as changes in storage volume inside the study area boundary and inside the well field boundary. Storage changes were computed using the specific-yield value (0.2) used by Stramel (1956). Water-level declines at individual wells such as observation well 886 are important for indicating changes at a specific time and are suggestive of the effects at that point, such as dewatered shal-

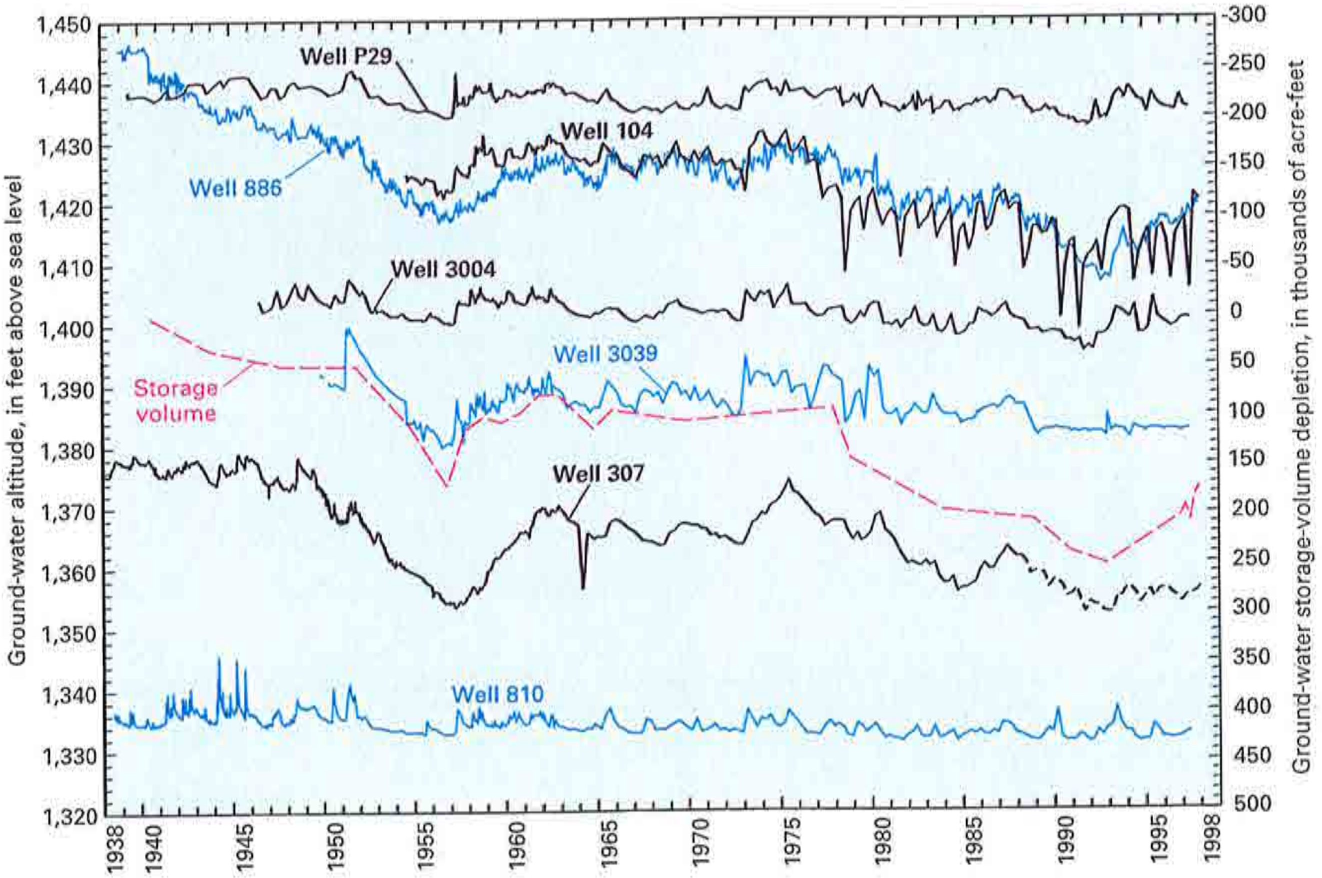
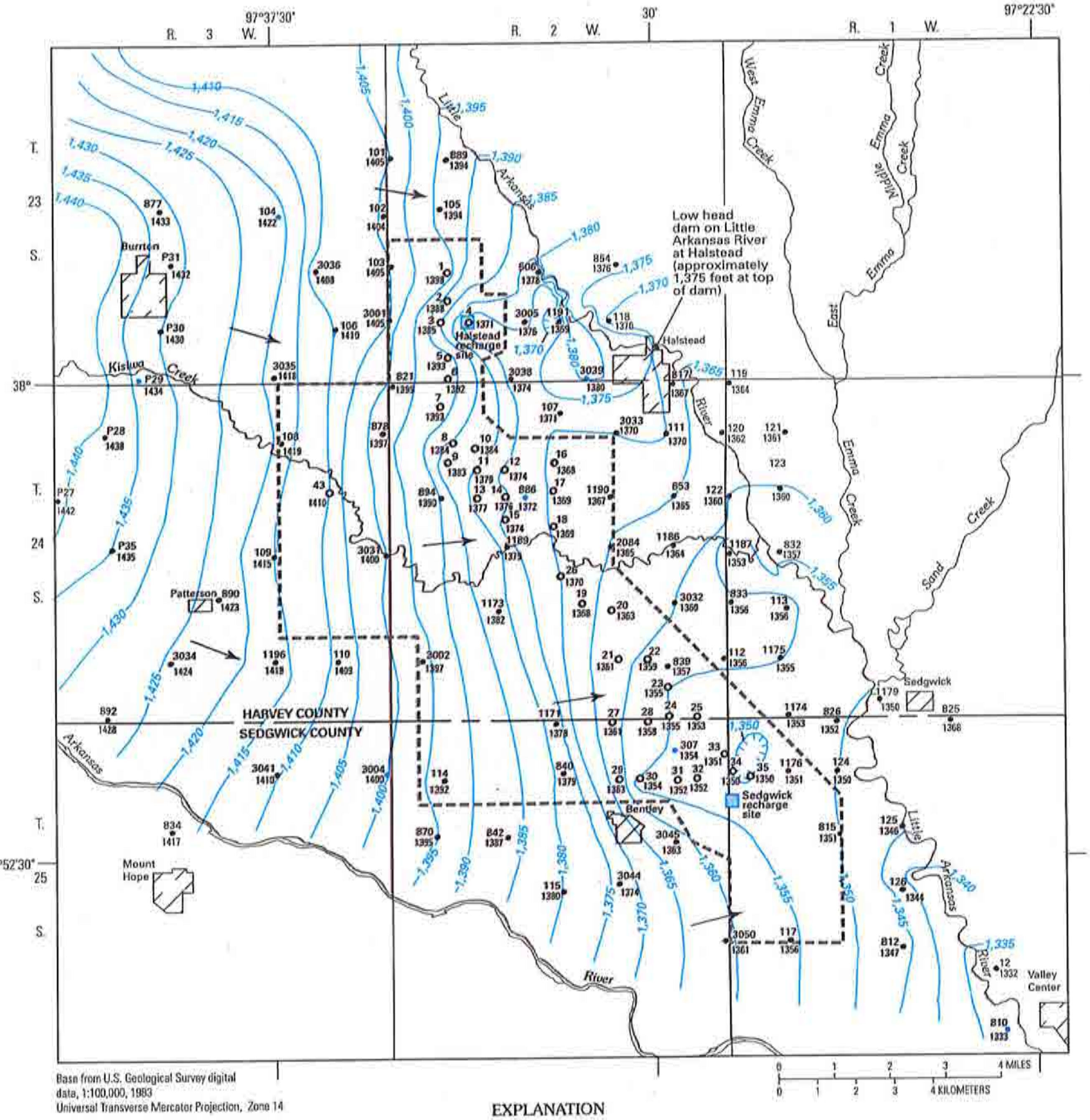


Figure 6. Hydrographs for selected wells and *Equus* Beds aquifer storage-volume depletions in study area, 1938–97 (source of data: water levels collected by city of Wichita personnel; data on file with U.S. Geological Survey in Lawrence, Kansas.) Location of wells shown in figure 7.

low wells or increased pumping costs to lift water from greater depths. Changes in storage are a better measure of the overall effect on the aquifer and represent a decrease in the ground-water resource available for use.

The period from September 1940 to early 1957 was a time of water-level decline resulting from the beginning of withdrawals from the city well field and culminating in the drought of the mid-1950's (fig. 2). City well field withdrawals increased throughout most of this period. Water levels had declined by more than 20 ft by 1957 in large parts of the well field as indicated by figure 11 and hydrographs for wells 886 and 307 in figure 6. The elongated northwest-southeast shape of the area of water-level declines generally encompasses the locations of the city pumping wells. The maximum water-level decline of 30.48 ft was noted in the observation well near city well 4 (fig. 11). Some water-level declines, generally less than 5 ft, even occurred in areas far from the city withdrawals toward the end of the drought as indicated by figure 11 and hydrographs of wells P29, 104, 810, and 3004 (fig. 6). By January 1957, a loss of 171,000 acre-ft of water in storage in the *Equus* Beds aquifer had occurred within the study area since August 1940 (table 1). The general flow direction in the *Equus* Beds aquifer in the study area at the culmination of the 1950's drought, however, remained generally west to east as it had been prior to development (August 1940) (fig. 7).

The period from 1958 to 1977 was generally characterized by relatively constant ground-water withdrawals, near-normal precipitation, and stable water levels with some recovery of water levels from the lows reached during the mid-1950's drought (wells 886 and 307, fig. 6), at least partly due to wetter than normal years immediately following the drought (fig. 2). The water-level change map for August 1940 to January 1970 (fig. 12) indicates again larger water-level declines in the east-central (older) part of the



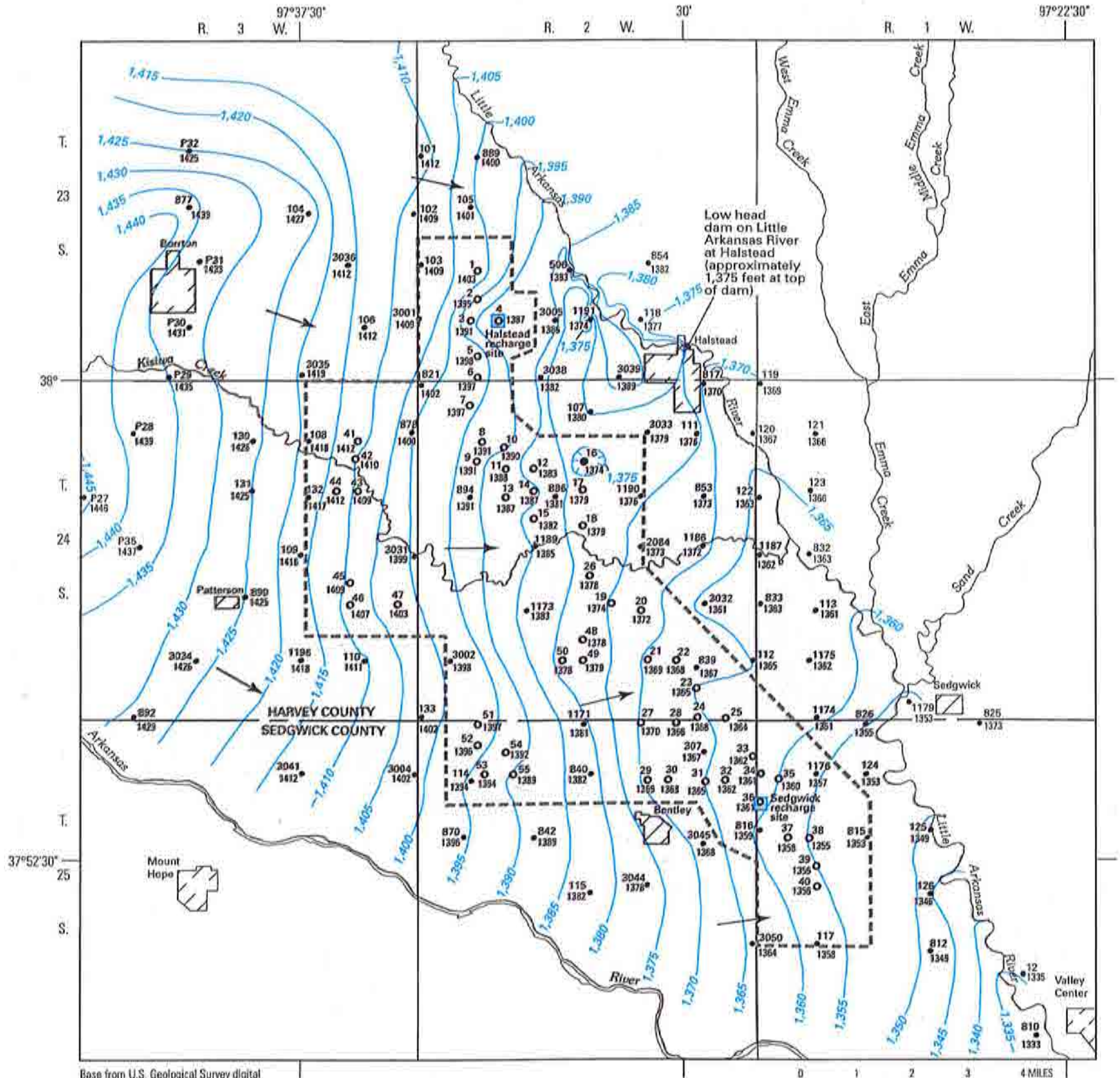
EXPLANATION

- Study area
- Water-level contour—Shows altitude of water level, January 1957. Contour interval 5 feet. Datum is sea level
- Approximate areal extent of Wichita well field
- Drainage ditch
- Approximate direction of ground-water flow
- Artificial recharge demonstration site
- Observation well in vicinity of city of Wichita supply well
- Other observation well
- Well with hydrograph in figure 6

Well number shown above symbol; water-level altitude, in feet above sea level, shown below.

Water-level measurements are made by city of Wichita personnel.

Figure 7. Water-level altitudes for January 1957 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.



EXPLANATION

- Study area
- —1,375— Water-level contour—Shows altitude of water level, January 1970. Contour interval 5 feet. Datum is sea level
- Approximate areal extent of Wichita well field
- Drainage ditch
- Approximate direction of ground-water flow
- Artificial recharge demonstration site
- Observation well in vicinity of city of Wichita supply well
- Other observation well

Well number shown above symbol; water-level altitude, in feet above sea level, shown below.

Water-level measurements are made by city of Wichita personnel.

Figure 8. Water-level altitudes for January 1970 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

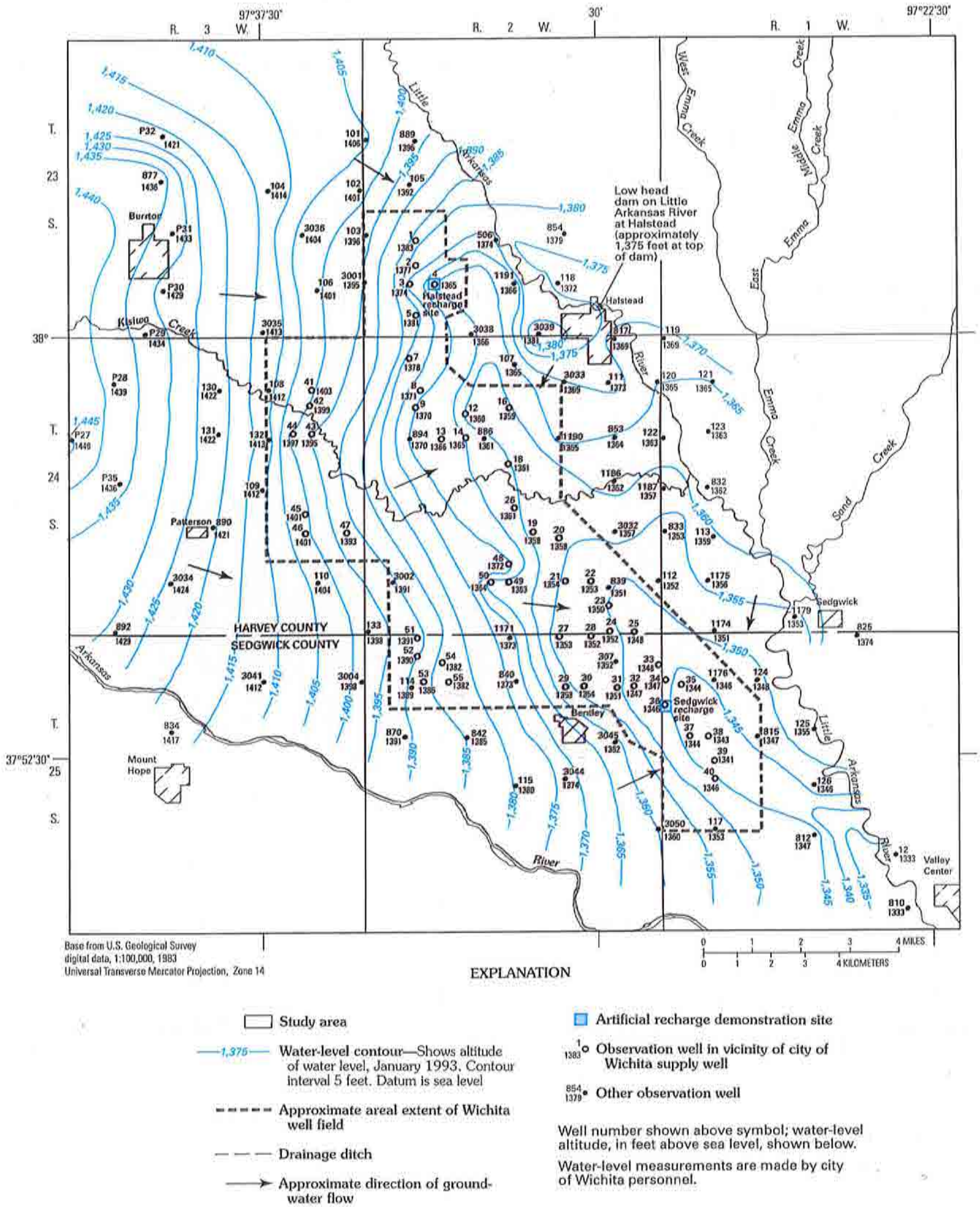
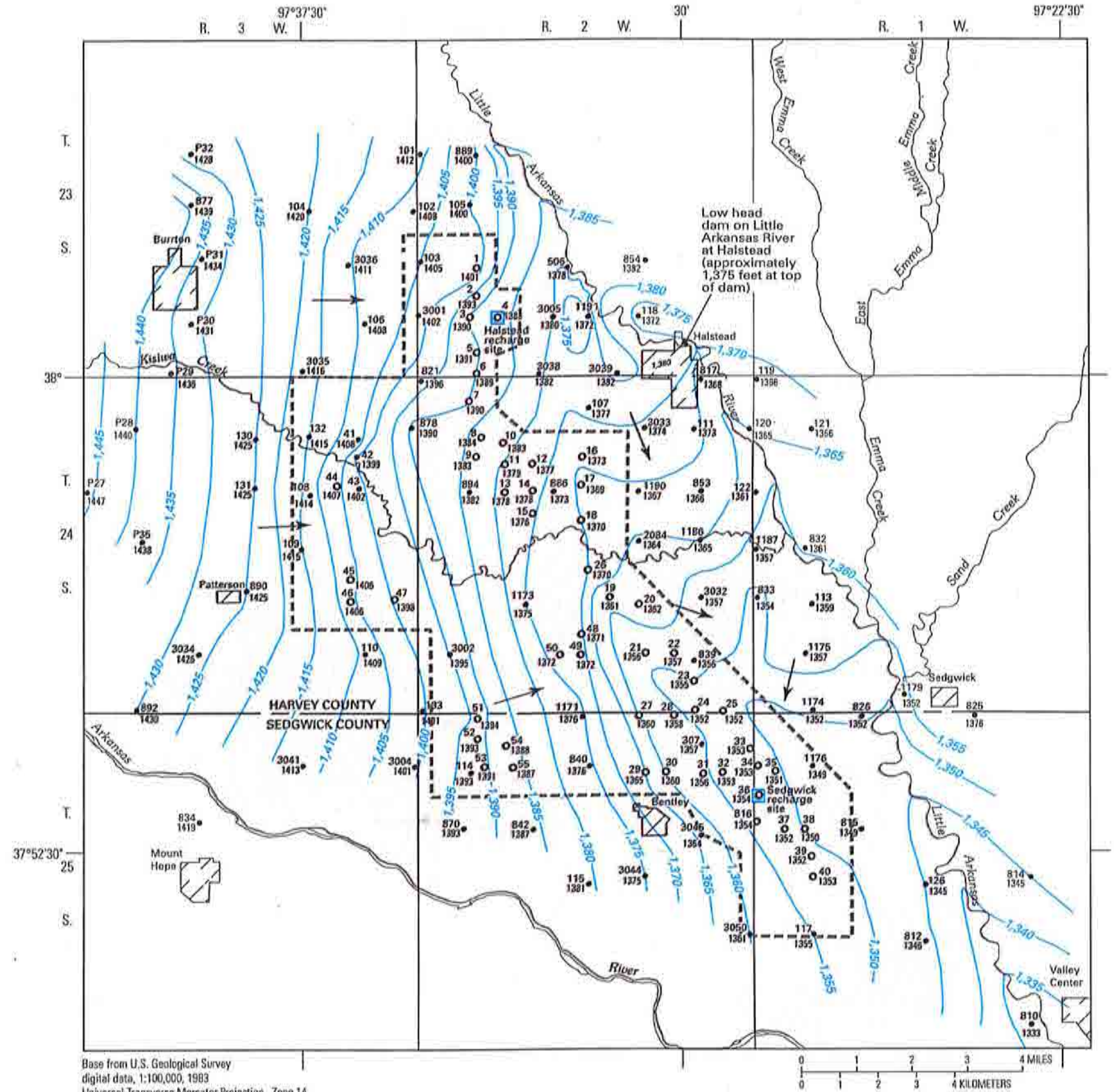


Figure 9. Water-level altitudes for January 1993 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.



EXPLANATION

- Study area
- —1,375— Water-level contour—Shows altitude of water level, January 1998. Contour interval 5 feet. Datum is sea level
- - - - - Approximate areal extent of Wichita well field
- — — — Drainage ditch
- ——➤ Approximate direction of ground-water flow
- Artificial recharge demonstration site
- 1 / 1401 o Observation well in vicinity of city of Wichita supply well
- 854 / 1382 • Other observation well

Well number shown above symbol; water-level altitude, in feet above sea level, shown below.

Water-level measurements are made by city of Wichita personnel.

Figure 10. Water-level altitudes for January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

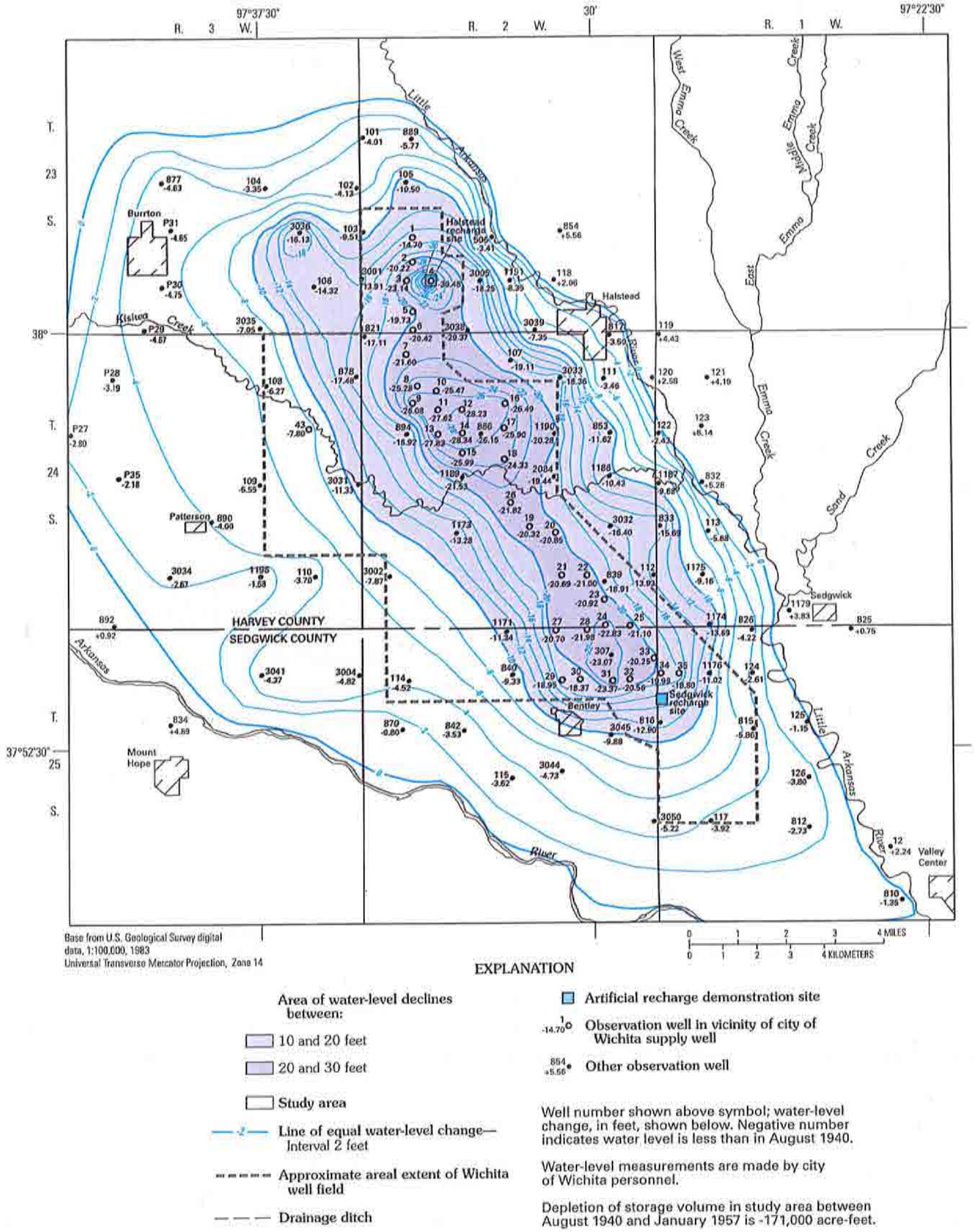
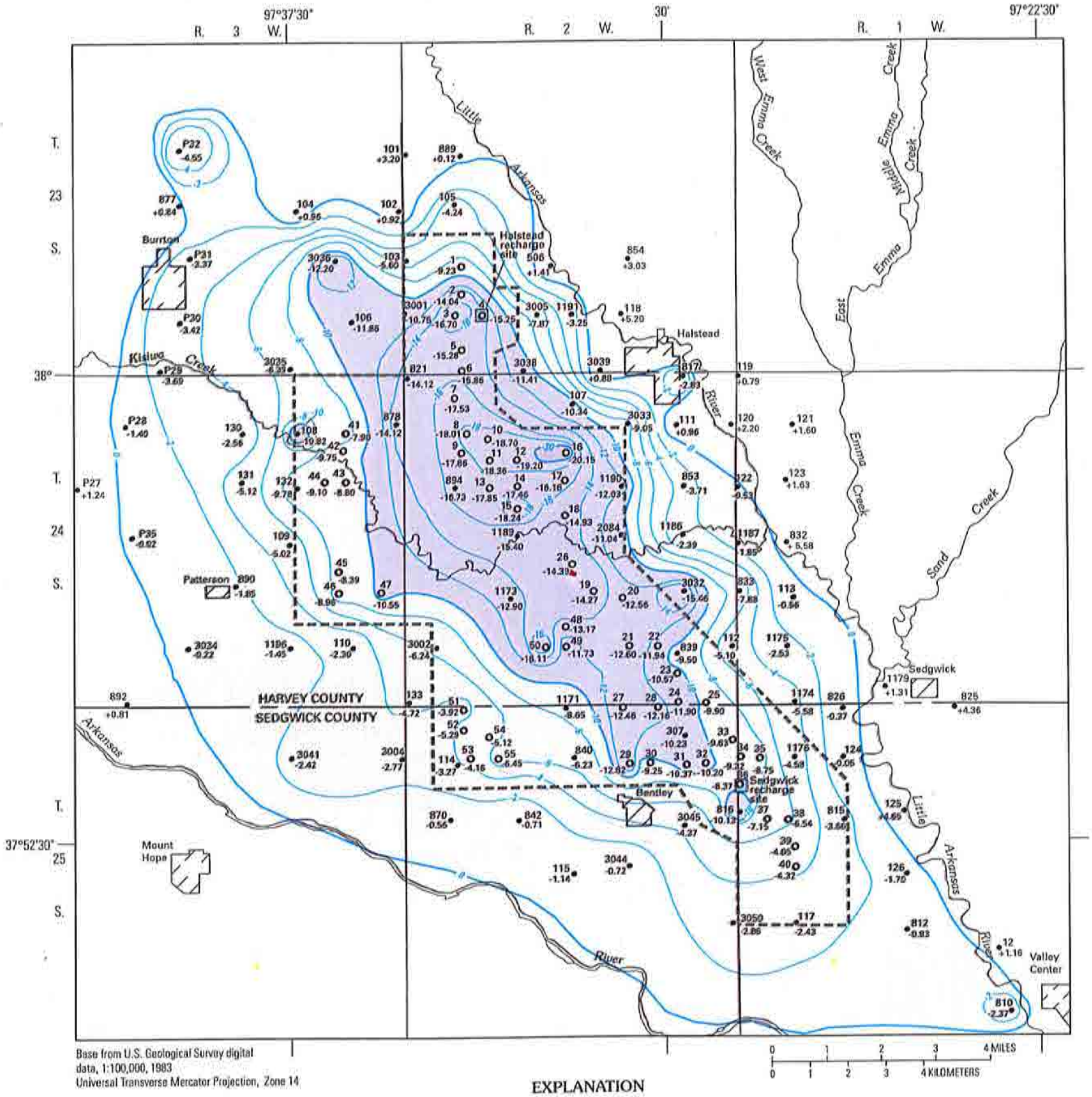


Figure 11. Water-level change from August 1940 to January 1957 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.



EXPLANATION

Area of water-level declines between:

- 10 and 20 feet
- 20 and 30 feet

Study area

—2— Line of equal water-level change—Interval 2 feet

- - - - Approximate areal extent of Wichita well field

— — — Drainage ditch

Artificial recharge demonstration site

1 / -9.23 o Observation well in vicinity of city of Wichita supply well

854 / +3.03 • Other observation well

Well number shown above symbol; water-level change, in feet, shown below. Negative number indicates water level is less than in August 1940.

Water-level measurements are made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and January 1970 is -108,000 acre-feet.

Figure 12. Water-level change from August 1940 to January 1970 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

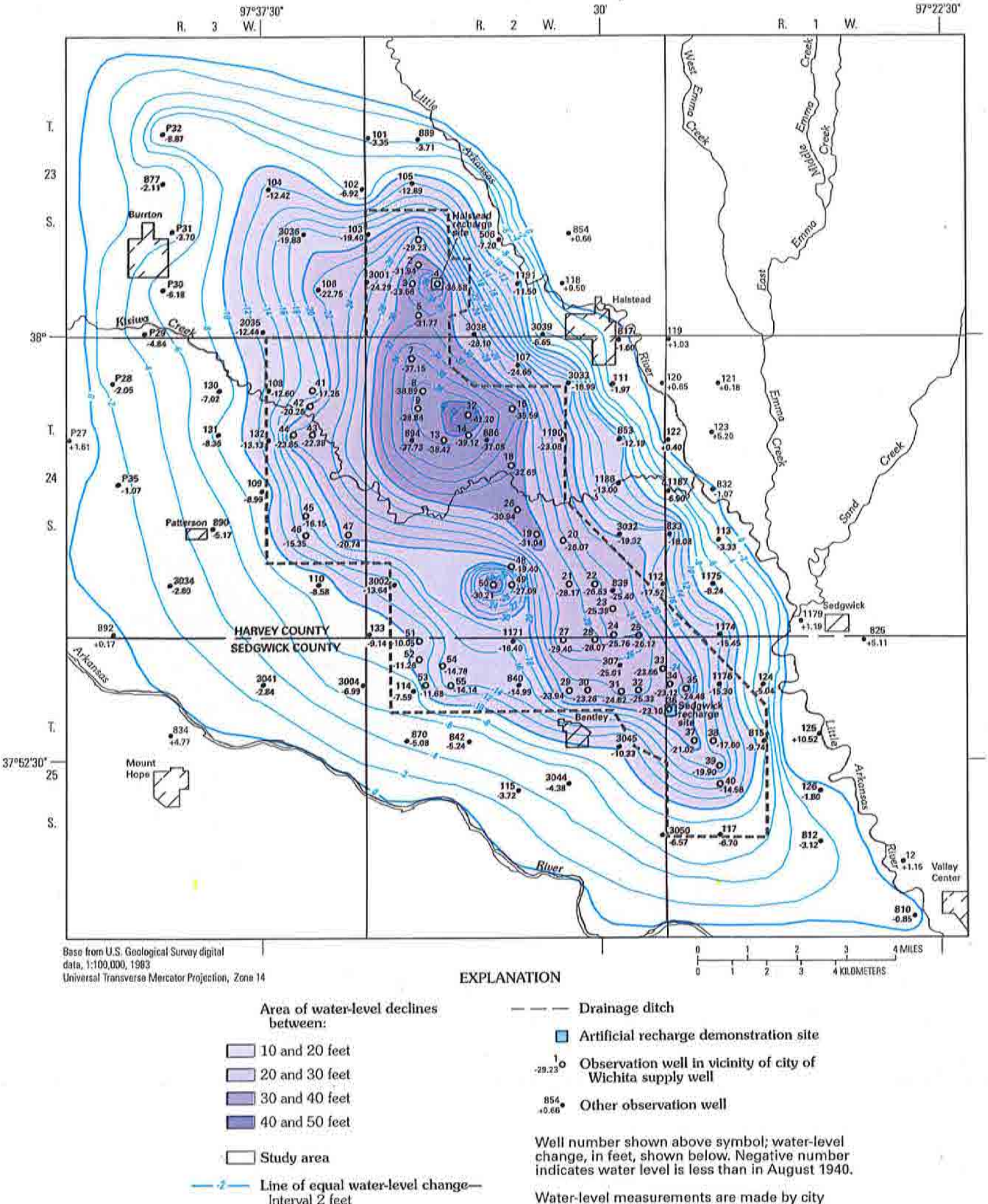


Figure 13. Water-level change from August 1940 to January 1993 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

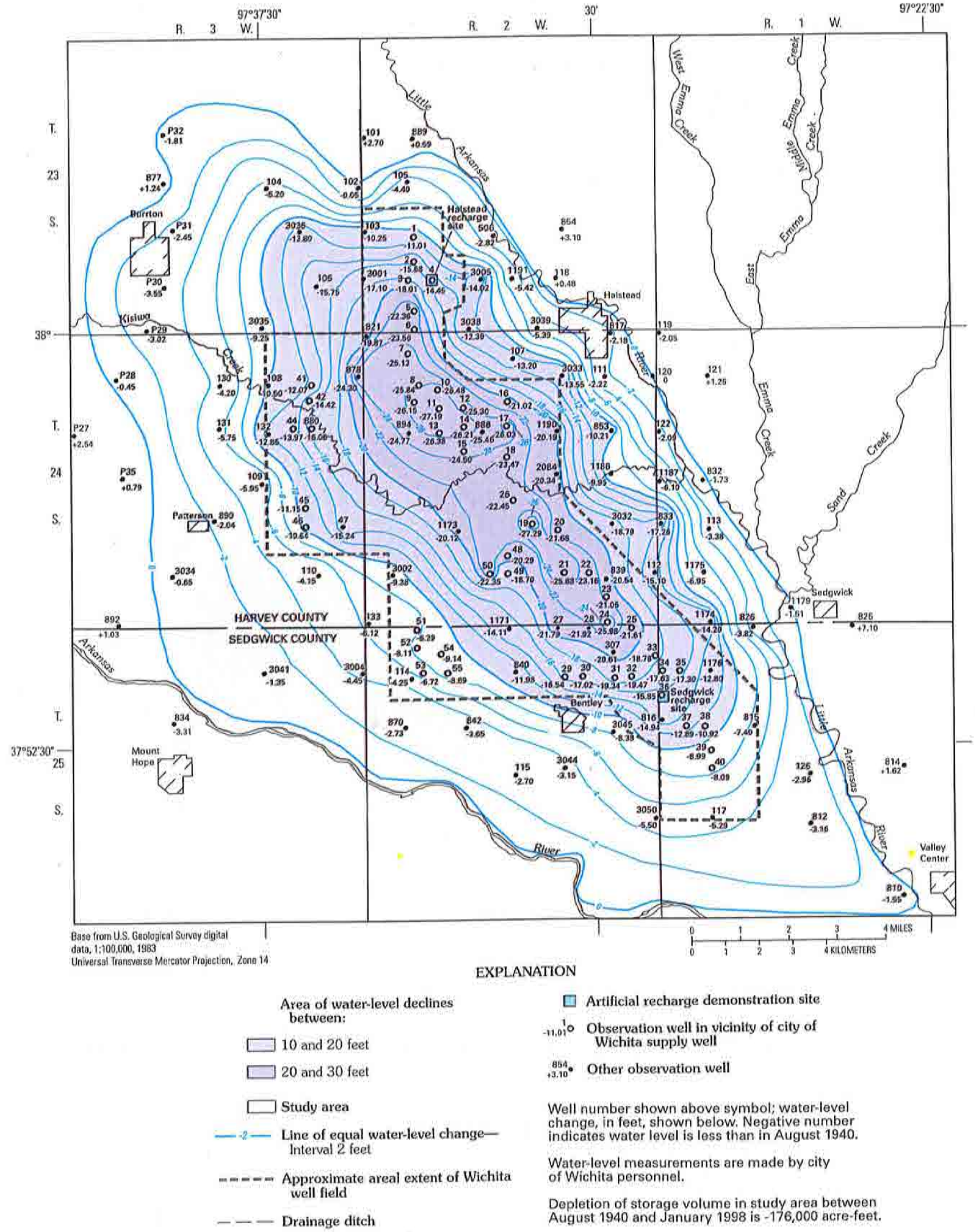
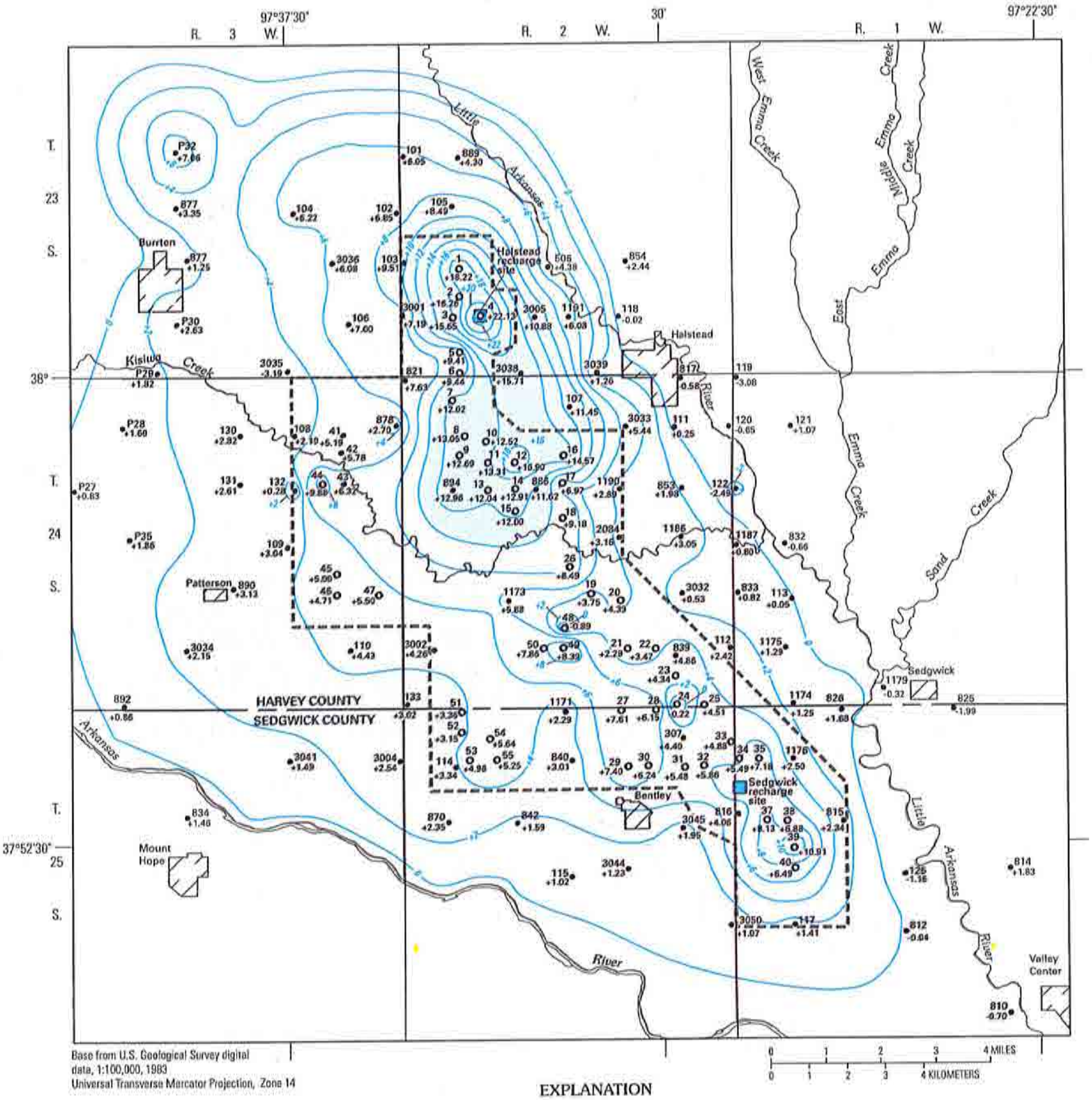


Figure 14. Water-level change from August 1940 to January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.



EXPLANATION

- Area of water-level recovery greater than 10 feet
- Study area
- +2 Line of equal water-level change—Interval 2 feet
- Approximate areal extent of Wichita well field
- Drainage ditch
- Artificial recharge demonstration site
- 1 o +18.22 Observation well in vicinity of city of Wichita supply well
- 854 • +2.44 Other observation well

Well number shown above symbol; water-level change, in feet, shown below. Negative number indicates water level is less than in January 1993.

Water-level measurements are made by city of Wichita personnel.

Replenishment of storage volume in study area between January 1993 and January 1998 is 79,000 acre-feet.

Figure 15. Water-level change from the January 1993 to January 1998 in the *Equus* Beds aquifer in the vicinity of the Wichita well field.

Table 1. Storage-volume changes in the *Equus* Beds aquifer in the Wichita well field area, August 1940 to January 1998

[Data on file with U.S. Geological Survey, Lawrence, Kansas]

Time period	Storage-volume change within study area, in acre-feet	Storage-volume change within well field area, in acre-feet
From August 1940 to January 1957	-171,000	- 93,700
From August 1940 to January 1970	-108,000	- 68,500
From August 1940 to January 1993	-255,000	-154,000
From August 1940 to January 1998	-176,000	-110,000
From January 1993 to January 1998	+ 79,000	+ 44,000

well field but lesser declines in most areas than in 1957. The maximum water-level decline from predevelopment to January 1970 was 20.15 ft near city well 16, which was the only well on the 1970 map where water-level declines exceeded 20 ft, whereas on the 1957 map (fig. 11), 34 wells had declines of more than 20 ft. The loss in aquifer storage of 108,000 acre-ft between August 1940 and January 1970 for the study area represents a 37-percent replenishment of storage from January 1957 lows. In 1970, the general flow direction in the *Equus* Beds aquifer in the study area (fig. 8) remained generally west to east as it had been in 1957 and prior to development.

The period from 1978 to 1993 produced another phase of ground-water-level declines (fig. 6) resulting from increased city well field withdrawals, greatly increased agricultural withdrawals (fig. 5), and the drought of 1988–92 (fig. 2). Many wells in the study area had their lowest water levels of record between 1991–93 at the climax of the drought. The largest water-level declines again occurred in the east-central part of the well field, particularly near city well 12 where a maximum water-level decline between August 1940 and January 1993 of 42.20 ft (fig. 13) was determined. Water-level declines also became more pronounced in areas farther from the original pumping wells in the east-central part of the well field. This was due to increased pumping in the western parts of the well field and increased agricultural withdrawals in the study area that by the early 1990's were similar in magnitude to city withdrawals from the well field (fig. 5). The effects of increased agricultural withdrawals during the late 1970's is apparent in the water-level declines and increased annual variations in water levels in well 104 (fig. 6). At their lowest in January 1993, water-level declines resulting from city and agricultural withdrawals encompassed an area of about 190 mi^2, extending from the Arkansas River to the Little Arkansas River in the vicinity of Halstead and Sedgwick. The largest ground-water-storage depletion recorded for the study area occurred in January 1993 at 255,000 acre-ft (table 1). By January 1993, water-level declines in the east-central part of the well field area had altered the water-level surface sufficiently between that area and the Little Arkansas River to change the water-level gradient and direction of flow from west to east prior to the 1988–92 drought, to northeast to southwest from the river toward the well field in that area (fig. 9). The drought was ended by greater-than-average precipitation and flooding during the spring and summer of 1993.

The period from of 1993 to 1998 has been characterized by near-average precipitation since the 1993 flooding (fig. 2), accompanied by decreased city well field withdrawals from the *Equus* Beds aquifer (fig. 5), and variable but near-average agricultural withdrawals compared to the previous years (fig. 5). These factors have resulted in some recovery in water levels from the record lows related to the drought of 1988–92. Water-level recoveries have been greatest in the east-central part of the well field and exceeded 10 ft in many places (figs. 6 and 15) where water-level declines had been largest. Even with the 1988–92 drought and the greatest recoveries occurring in the older pumping areas in the east-central part of the well field, the largest water-level declines since August 1940 remain in the older pumping areas (fig. 14). An observation well near city well 19 had the largest water-level decline, 27.29 ft, between August 1940 and January 1998. The loss of aquifer storage of 176,000 acre-ft between August 1940 and January 1998 in the study area represents a 31-percent replenishment of storage from the January 1993 low of record and is similar to the loss of storage at the low point of the 1950's drought. Water-level gradients and direction of flow in the area between the east-central part of the well field and the Little Arkansas River (fig. 10) are no longer from the river to the well field, as was the case during January 1993, but the gradients and direction present prior to development and up through at least the mid-1970's also have not been reestablished. In other parts of the study area, the direction of flow remained generally from west to east as had been the case since predevelopment.

The city of Wichita is investigating the potential for artificial recharge at two sites near Halstead and Sedgwick (fig. 1). Operation of the Halstead recharge demonstration site (fig. 1) began in September 1997. Water levels in a well near the Halstead recharge site (observation well near city supply well 4) increased more than 2 ft between July and October 1997 measurements, whereas water levels in near wells declined, indicating some effects from the recharge site (Aucott and others, 1998). A 20-ft rise in water level was noted in an observation well closer to the recharge site shortly after recharge began (David Stous, Burns and McDonnell Engineering Consultants, written commun., 1997).

The future of ground-water-level changes and the availability of water from the *Equus* Beds aquifer in the study area is dependent on the wise management of the resource. The historical record demonstrates the effects of city and agricultural withdrawals, artificial recharge, and droughts on water levels and the availability of ground water in storage. These factors need to be balanced and managed efficiently to optimize use of this resource and preserve it for future generations. The Equus Beds Groundwater Management District No. 2 was formed in 1975 to manage ground-water supplies in the study area. The District works with municipal and agricultural users to manage the aquifer using the "aquifer safe-yield principle," which limits ground-water withdrawals to annual ground-water recharge, and a "ground-water quality principle" as noted in Equus Beds Groundwater Management District No. 2 (1995).

SUMMARY

Quaternary alluvial deposits, known as the *Equus* beds, are as much as 250 ft thick in the study area northwest of Wichita, Kansas, and consist primarily of sand and gravel interbedded with clay or silt. The Wellington Formation underlies the Quaternary deposits, forming the bedrock confining unit below these deposits. Dissolution of the Hutchinson Salt Member, the middle member of the Wellington Formation, has resulted in subsidence of the overlying upper shale member of the Wellington Formation, the formation of low areas in the bedrock surface, and the accumulation of the alluvial deposits that now comprise the *Equus* beds.

The Wichita well field was developed in the *Equus* Beds aquifer northwest of Wichita, Kansas, to supply water to the city. In August 1940, ground-water levels in the study area were in a near-predevelopment condition. On September 1, 1940, the city began pumping from 25 wells in the well field, and by 1959 there were 55 wells in use in the well field. City ground-water withdrawals from the *Equus* Beds aquifer increased steadily from the beginning of pumpage until the early 1950's. Agricultural withdrawals were relatively small until the 1970's. Water levels declined by more than 20 ft in much of the city well field area between August 1940 and January 1957 in response to the increasing city withdrawals and the mid-1950's drought. During the late 1950's, water levels recovered somewhat from their mid-1950's lows and were relatively stable through the mid-1970's in response to relatively constant city withdrawals and normal climatic conditions. City withdrawals increased sporadically from the late 1970's through the early 1990's in response to increased demand and the 1988–92 drought. Agricultural withdrawals in the study area increased substantially beginning in the late 1970's, and by the end of the 1988–92 drought, were nearly equal to city withdrawals from the well field. In response to increased withdrawals and drought conditions, water levels in most wells declined to lows of record during 1991–93, declining as much as 40 ft or more. Ground-water storage depletion reached a maximum of record of 255,000 acre-ft in January 1993. Water-level declines encompassed an area of about 190 mi^2 at their maximum in January 1993 and extended from the Arkansas River to the Little Arkansas River in the vicinity of Halstead and Sedgwick. Since flooding in 1993, city withdrawals have decreased due to the increasing reliance of the city on Cheney Reservoir as a water-supply source, while average agricultural withdrawals were similar to withdrawals in the early 1990's, and precipitation was near average. Ground-water levels have since recovered more than 10 ft in some areas and aquifer storage replenished by 79,000 acre-ft between 1993 and 1998 primarily as a result of decreased city withdrawals.

The future availability of water from the *Equus* Beds aquifer in the study area is dependent on the wise management and long-term balance of city and agricultural withdrawals and artificial recharge especially during droughts. The Equus Beds Groundwater Management District No. 2 manages the *Equus* Beds aquifer in part by limiting ground-water withdrawals to annual ground-water recharge.

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EXHIBIT E

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CITY OF WICHITA, KANSAS

Status of Ground-Water Levels and Storage Volume in the *Equus* Beds Aquifer Near Wichita, Kansas, January 2000–January 2003

Water-Resources Investigations Report 03-4298

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

002004

Cover photograph—Ray Casanova, USGS, Wichita, Kansas, measuring an areal index well (IW–22) in Harvey County, July 11, 2002 (photograph taken by Trudy Bennett, USGS, Wichita, Kansas).

Status of Ground-Water Levels and Storage Volume in the *Equus* Beds Aquifer Near Wichita, Kansas, January 2000–January 2003

By Cristi V. Hansen and Walter R. Aucott

Prepared in cooperation with the
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Contents

Abstract 1
Introduction 1
  Purpose and Scope 3
  Methods 3
  Description of Study Area 3
  Previous Studies 3
Geology and Ground Water 5
Ground-Water-Level Changes 5
Storage-Volume Changes 31
Effects of Artificial Recharge 34
Summary 34
Selected References 35

Figures

1. Maps showing location of study area near Wichita, south-central Kansas 2
2. Graphs showing relation of precipitation, water use for agricultural irrigation and by city of Wichita for public supply, and water-level altitudes in observation walls 104 and 886 and *Equus* Beds aquifer storage-volume change in study area, 1938–January 2003 4
3. Generalized geologic section 6

4–7. Maps showing water-level altitudes in *Equus* Beds aquifer in study area for:
  4. August 1940 8
  5. October 1992 9
  6. April 2000 10
  7. January 2003 11

8–20. Maps showing water-level change in *Equus* Beds aquifer in study area:
  8. August 1940–January 2000 12
  9. August 1940–April 2000 13
  10. August 1940–July 2000 14
  11. August 1940–October 2000 15
  12. August 1940–January 2001 16
  13. August 1940–April 2001 17
  14. August 1940–July 2001 18
  15. August 1940–October 2001 19
  16. August 1940–January 2002 20
  17. August 1940–April 2002 21
  18. August 1940–July 2002 22
  19. August 1940–October 2002 23
  20. August 1940–January 2003 24

21. Map showing maximum seasonal water-level changes in *Equus* Beds aquifer in study area during January 2000–January 2003 26

22–25. Maps showing water-level change in *Equus* Beds aquifer in study area:
22. October 1992–January 2003 27
23. October 1992–April 2000 28
24. April 2000–January 2003 29
25. January 2000–January 2003 30
26. Graph showing storage-volume changes in *Equus* Beds aquifer in study area for significant periods during August 1940–January 2003 33

Table

1. Storage-volume changes in *Equus* Beds aquifer near Wichita, south-central Kansas, August 1940–January 2003 32

Conversion Factors, Abbreviations, and Datums

Multiply	By	To obtain
acre-foot (acre-ft)	1,233	cubic meter (m^3)
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year (m^3/yr)
degree Fahrenheit (°F)	([1])	degrees Celsius (°C)
foot (ft)	0.3048	meter (m)
inch (in.)	25.4	millimeter (mm)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
mile (mi)	1.609	kilometer (km)
square mile (mi^2)	2.590	square kilometer (km^2)

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

$$°F = (1.8 \times °C) + 32$$

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

$$°C = (°F - 32) / 1.8$$

Prior to April 2000, vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29). For April 2000 and after, vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88). NAVD 88 vertical coordinates are about 0.5 ft higher than NGVD 29 vertical coordinates in the part of south-central Kansas discussed in this report.

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

Status of Ground-Water Levels and Storage Volume in the *Equus* Beds Aquifer Near Wichita, Kansas, January 2000–January 2003

By Cristi V. Hansen and Walter R. Aucott

Abstract

The *Equus* Beds aquifer northwest of Wichita, Kansas, was developed to supply water to Wichita residents and for irrigation in south-central Kansas beginning on September 1, 1940. Ground-water pumping for city and agricultural use from the aquifer caused water levels to decline in a large part of the area. Irrigation pumpage in the area increased substantially during the 1970s and 1980s and accelerated water-level declines. A period of water-level rises associated with greater-than-average precipitation and decreased city pumpage from the study area began in 1993. An important factor in the decreased city pumpage was increased use of Cheney Reservoir as a water-supply source by the city of Wichita; as a result, city pumpage from the *Equus* Beds aquifer during 1993–2002 went from being greater than one-half to slightly less than one-third of Wichita's water usage. Since 1995, the city also has been investigating the use of artificial recharge in the study area to meet future water-supply needs and to protect the aquifer from the intrusion of saltwater from natural and human-related sources to the west.

During January 2003, the direction of ground-water flow in the *Equus* Beds aquifer in the area was generally from west to east, similar to predevelopment of the aquifer. The maximum water-level decline since 1940 for the period January 2000 to January 2003 was 29.54 feet in July 2002 at well 3 in the northern part of the area. Cumulative water-level changes from January 2000 to January 2003 typically were less than 4 feet with rises of less than 4 feet common in the central part of the area; however, declines of more than 4 feet occurred in the northwestern and southern parts of the area.

The recovery of water levels and aquifer storage volumes from record low levels in October 1992 generally continued to April 2000. The recovery of about 182,000 acre-feet of storage volume in the area from October 1992 to April 2000 represents about a 64-percent recovery of the storage depletion that occurred from August 1940 to October 1992. About 47 percent of this recovery was lost from April 2000 to October 2002 when storage volume in the area decreased by about 86,000 acre-feet. Major contributors to the decreases in water levels and storage volumes were reduced recharge associated with precipitation that was less than in the preceding 5 years and increased irrigation pumpage. The loss of storage probably would have been larger if the continued decrease in city pumpage, which is closely associated with the water-level rises in the central part of the study area, and increased city use of water from Cheney Reservoir had not occurred. The effect of artificial recharge on water levels and storage volume probably was masked by the generally larger decreases in city pumpage in the area.

Introduction

The Wichita well field in the *Equus* Beds aquifer in southwestern Harvey County and northwestern Sedgwick County was developed to supply water to residents of Wichita and for irrigation in the study area in south-central Kansas (fig. 1). On September 1, 1940, Wichita began pumping from 25 wells completed in the aquifer in the central part of the study area (Stramel, 1956) (central part of the study area shown in fig. 1), and by 1959, there were 55 wells in use by the city of Wichita (Stramel, 1967). Ground-water pumpage from the aquifer for city and agricultural use has caused water levels to decline in a large part of the study area. A substantial decline in water levels occurred from 1940 until the drought of the 1950s ended in early 1957 (Stramel, 1967). Ground-water pumpage for irrigation in the study area increased substantially during the 1970s and 1980s and accelerated water-level declines (Myers and others, 1996; Aucott and Myers, 1998). Most of the water-level declines can be attributed to ground-water pumpage; however, climatic conditions (and thus recharge to the *Equus* Beds aquifer) also have affected water levels.

The *Equus* Beds Groundwater Management District No. 2 was formed in 1975 as part of the effort to balance the factors affecting water levels in the *Equus* Beds aquifer, to efficiently manage and optimize the use of water from the aquifer, and to preserve the aquifer for future generations. The District works with municipal and agricultural users to manage pumpage from the aquifer using the "aquifer safe-yield principle," which limits ground-water pumpage to the annual amount of ground-water

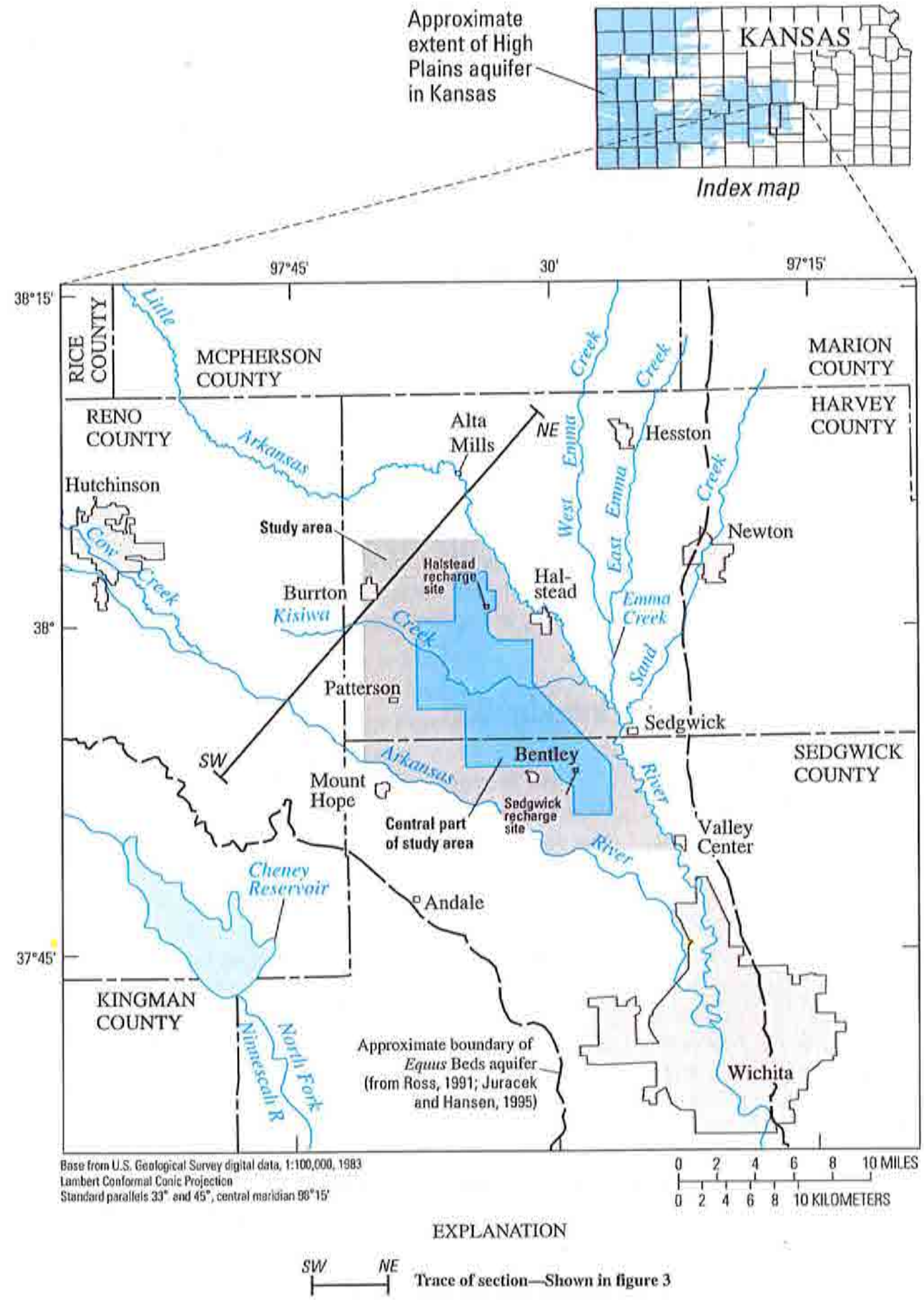


Figure 1. Location of study area near Wichita, south-central Kansas (modified from Aucott and Myers, 1998).

recharge as noted in the management program of *Equus* Beds Groundwater Management District No. 2 (1995).

In 1965, the city of Wichita began using water from Cheney Reservoir (Stramel, 1967) in addition to water from the *Equus* Beds aquifer. Since 1995 (Warren and others, 1995), the city of Wichita, in cooperation with *Equus* Beds Groundwater Management District No. 2 (Halstead, Kansas), Bureau of Reclamation (U.S. Department of the Interior), U.S. Geological Survey (USGS), U.S. Environmental Protection Agency, various Kansas State agencies, Burns and McDonnell Engineering Consultants (Kansas City, Missouri), and Mid-Kansas Engineering Consultants (Wichita, Kansas), has been investigating the potential for using artificial ground-water recharge in the study area to meet future water-supply needs and to protect the aquifer from the intrusion of saltwater from natural and human-related sources to the west. Because of the social and economic importance of ground-water resources and the potential changes that artificial recharge may bring to the aquifer, the city of Wichita conducted a cooperative study with the USGS to document changes in historical hydrologic and water-quality conditions and the probable causes of these changes in the study area, to develop a baseline condition for evaluating the effects of artificial recharge on ground-water levels in the aquifer, and to periodically review changes in the ground-water flow system.

The USGS and the city of Wichita have worked cooperatively since 1940 in evaluating the *Equus* Beds aquifer and its interaction with streams in the area to further the understanding of the entire hydrologic system and to provide information to aid local decisionmaking. The understanding gained from this cooperative study of the hydrologic system and the *Equus* Beds aquifer can contribute to the wise management of water resources where similar hydrologic conditions exist elsewhere. This report is prepared in cooperation with the city of Wichita.

Purpose and Scope

The purpose of this report is to describe ground-water-level and storage-volume changes in the *Equus* Beds aquifer northwest of Wichita during January 2000 to January 2003 as compared with predevelopment (1940) ground-water levels and to update historical information related to changes in the aquifer since 1940. Maps of ground-water-level measurements and water-level changes are presented. Two hydrographs of ground-water levels were selected to show historical water-level variations. Historical water-use and climate information also are presented. The information in this report can be used to monitor and improve understanding of the effects of climate, water use, and water-resource management practices on water supplies in the *Equus* Beds aquifer, an important source of water for the city of Wichita and the surrounding area.

Methods

Extensive information is available to describe hydrologic conditions in the study area. Water-level data have been collected periodically from more than 100 wells by city of Wichita personnel using standard water-level measurement techniques similar to USGS methods described in Stallman (1971). Data collection began just prior to the beginning of city pumpage from the aquifer in the study area in 1940; water levels in most wells have been measured at least quarterly. These data are stored by the city in paper and electronic form and by the USGS in electronic form.

During 2001 and 2002, 38 pairs of areal index wells were installed in and near the study area for the city of Wichita. Each pair of areal index wells consists of a well completed in the upper part of the aquifer and another well completed in the lower part of the aquifer. These wells were designed for use by the city to monitor the quality of water in the aquifer throughout the study area and to determine if there are any water-quality differences between the shallow and deep parts of the aquifer (Andrew C. Ziegler, U.S. Geological Survey, oral commun., September 2003). As water levels also can be measured in these wells, the wells were added to the water-level monitoring network in the study area in 2002. Water levels in these index wells are measured by the USGS and by *Equus* Beds Ground-water Management District No. 2 (GMD2). The data collected by the USGS are stored in the National Water Information System (NWIS) database and are available at the following URL: *http://ks.waterdata.usgs.gov/nwis/gw*

The data collected by GMD2 is stored in KGS's Water Information Storage and Retrieval Database (WIZARD) and are available at the following URL:
http://www.kgs.ku.edu/Magellan/WaterLevels/index.html

Description of Study Area

The study area (fig. 1) includes 165 mi^2 and is located in Harvey and Sedgwick Counties, northwest of Wichita, Kansas. The study area is in the Arkansas River section of the Central Lowlands physiographic province (Schoewe, 1949). There is little topographic relief in the study area. For the most part, the land surface slopes gently toward the major streams in the area. The study area is bounded on the southwest by the Arkansas River and on the northeast by the Little Arkansas River. The center or central part of the study area (fig. 1), which is referred to throughout this report, is the historic center of pumping in the study area and includes wells that supply water to the city of Wichita and for irrigation.

South-central Kansas has a continental climate that is characterized by large variations in seasonal temperatures, moderate precipitation, and windy conditions. In Wichita, Kansas, long-term daily average temperatures for 1971–2000 range from 30.2 °F in January to 81.0 °F in July (National Oceanic and Atmospheric Administration, 2002). The long-term annual mean precipitation for 1940–2002 at weather stations near the study area (at Hutchinson, Mount Hope, Newton, Sedgwick, and Wichita) is 31.18 in. (National Oceanic and Atmospheric Administration, 1998–2001b; Mary Knapp, State Climatologist, written commun., March 20, 2003) (fig. 2*A*). Most of this precipitation commonly occurs during spring and summer (May–September). Although mean annual precipitation in 2002 was near average, about one-fourth of the year's precipitation did not occur until the month of October (Mary Knapp, State Climatologist, Kansas State University, written commun., March 20, 2003)—after the growing season ended and in a month that normally receives less than one-tenth of the annual precipitation.

Previous Studies

Water-level data have been collected periodically by the city of Wichita in the study area since 1940 and are on file with the city and the USGS in Wichita and Lawrence, Kansas, respectively. Water-level data also have been collected by

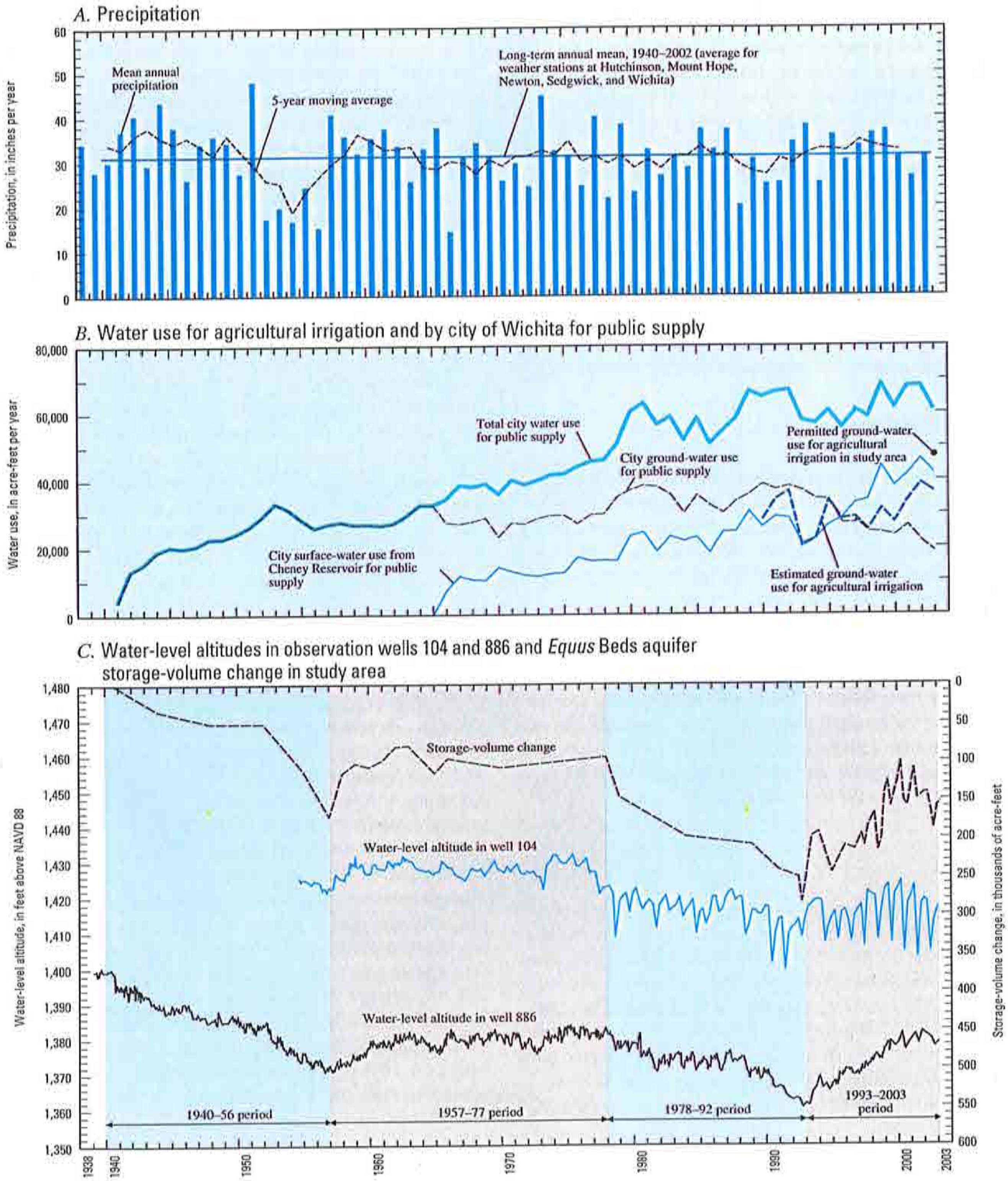


Figure 2. Relation of *(A)* precipitation, *(B)* water use for agricultural irrigation and by city of Wichita for public supply, and *(C)* water-level altitudes in observation wells 104 and 886 and *Equus* Beds aquifer storage-volume change in study area, 1938–January 2003 (modified from Aucott and others, 1998). Source: *(A)* precipitation data from National Oceanic and Atmospheric Administration (1998–2001b) and Mary Knapp (State Climatologist, Kansas State University, written commun., March 20, 2003); *(B)* water-use data from Stramel (1956, 1967), Gerald T. Blain (city of Wichita, written commun., 1997), Joan Kenny (U.S. Geological Survey, written commun., 2000 and 2003), Brownie Wilson (Kansas Water Office, written commun., 2000), and Kelly Emmons (Kansas Department of Agriculture, Division of Water Resources, written commun., 2003); *(C)* water-level altitude data from Stramel (1956, 1967) and from data collected by city of Wichita, *Equus* Beds Groundwater Management District No. 2, and on file with U.S. Geological Survey, Lawrence, Kansas. Location of observation wells is shown in figures 4–25. Storage-volume changes from Stramel (1956, 1967), Aucott and Myers (1998), Aucott and others (1998), Hansen and Aucott (2001), and data on file with U.S. Geological Survey in Lawrence, Kansas.

Equus Beds Groundwater Management District No. 2 since 1978 from wells completed in the *Equus* Beds aquifer (*Equus* Beds Groundwater Management District No. 2, 1995). Annual water-level data for the High Plains aquifer (fig. 1), which includes the *Equus* Beds aquifer, have been collected since 1937 by the Kansas Department of Agriculture (Division of Water Resources), USGS, and the Kansas Geological Survey (KGS). The data on file with the USGS in Lawrence, Kansas, also are stored in the NWIS database and are available at URL *http://ks.waterdata.usgs.gov/nwis/gw;* data on file with the KGS are stored in their WIZARD database (Kansas Geological Survey, 2002). Historical and near-real-time data and reports associated with the Equus Beds Ground-Water Demonstration Recharge Project (Ziegler and others, 1999) are available at URL
http://ks.water.usgs.gov/Kansas/studies/equus/

Williams and Lohman (1949) and Stramel (1956, 1967) have published water levels and water-level-altitude and decline maps for the study area. Ross and others (1997) noted water-level rises in the *Equus* Beds aquifer from 1993 to 1997 and attributed them largely to decreases in withdrawals by the city of Wichita. Aucott and Myers (1998), Aucott and others (1998), and Hansen and Aucott (2001) published water-level decline maps for the study area and discussed the changes in storage volume for noteworthy past and recent periods of time. Myers and others (1996) evaluated the hydrologic interaction between the Arkansas River and the *Equus* Beds aquifer in the study area. Water-level data for the *Equus* Beds and High Plains aquifers have been compiled and mapped recently in Kansas by Olea and Davis (2002) and Woods and Sophocleous (2002) and regionally by McGuire and Sharpe (1997), McGuire and Fischer (1999), and McGuire (2001).

Geology and Ground Water

Quaternary deposits occur throughout the study area primarily as alluvial deposits. These alluvial deposits, known locally as the *Equus* beds, are as much as 250 ft thick in the study area (fig. 3). The *Equus* beds consist primarily of sand and gravel interbedded with clay or silt but locally may consist primarily of clay with thin sand and gravel layers (Lane and Miller, 1965a; Myers and others, 1996). The middle part of the deposits generally has more fine-grained material than the lower and upper parts (Lane and Miller, 1965b; Myers and others, 1996).

The Wellington Formation of Permian age underlies the Quaternary deposits in the study area and forms the bedrock confining unit below these deposits. The Wellington Formation is about 700 ft thick (Bayne, 1956) and consists of three members—the lower anhydrite member, about 200 ft thick; the Hutchinson Salt Member, about 300 ft thick; and the upper shale member, about 200 ft thick (Myers and others, 1996). Dissolution of the Hutchinson Salt Member has resulted in subsidence of the overlying upper shale member, formation of low areas in the bedrock surface, and concurrent accumulation of alluvial deposits that now compose the *Equus* Beds aquifer (fig. 3) (Myers and others, 1996).

The *Equus* Beds aquifer is the easternmost extension of the High Plains aquifer in Kansas (Stullken and others, 1985; Hansen and Aucott, 2001). The *Equus* beds are an important source of ground water because of the generally shallow depth to the water table, the large saturated thickness, and the generally good water quality. Near the Arkansas River, the water table may be as little as 10 ft below land surface. Farther from the Arkansas River and near the Little Arkansas River, the water table is at a greater depth, depending on the altitude of the land surface and the amount of water-level decline that has been caused by ground-water withdrawals. The maximum saturated thickness of the *Equus* Beds aquifer within the study area, almost 250 ft, is near the Arkansas River and corresponds to the lowest areas of the underlying bedrock surface (fig. 3).

Ground-Water-Level Changes

Ground-water-level declines can result from pumpage, decreased recharge resulting from less-than-average precipitation, and other factors. Droughts, such as occurred during 1952–56 and 1988–92 (fig. 2*A*), tend to decrease the amount of recharge available and increase the demand for and thus pumpage of ground water (fig. 2*B*), resulting in increased water-level declines (fig. 2*C*). Periods of greater-than-average rainfall, such as occurred in 1957–62 (fig. 2*A*), tend to increase the amount of recharge available and decrease the demand for and thus pumpage of ground water (fig. 2*B*), resulting in water-level rises (fig. 2*C*). If these water-level declines or rises are large enough, they may locally alter the direction of ground-water flow.

Aucott and Myers (1998) identified four noteworthy periods of water-level change (fig. 2*C*): 1940–56, the initial water-level decline period when pumpage began in the study area, which includes a phase of accelerated declines in the mid-1950s coinciding with drought conditions; 1957–77, a period of general equilibrium with relatively stable city pumpage and water levels and increasing irrigation pumpage that became significant in the late 1970s; 1978–92, another period of water-level declines and increased city and irrigation pumpage due to increased demands and drought conditions; and 1993 to 1998, a period of water-level rises associated with generally greater-than-average precipitation and decreased city pumpage. The first three periods have been well documented by Aucott and Myers (1998) and will not be described in this report. According to Hansen and Aucott (2001), the fourth period—the period of water-level rises seen by Aucott and Myers (1998) during 1993 to 1998—did not end in 1998 but rather continued during 1998 to 2000.

Description of noteworthy periods of water-level change in the study area is facilitated by the use of hydrographs of water levels in observation wells 104 and 886 (fig. 2*C*). The hydrograph of well 104 serves as a representative descriptor of agricultural irrigation effects near the northern edge of the study

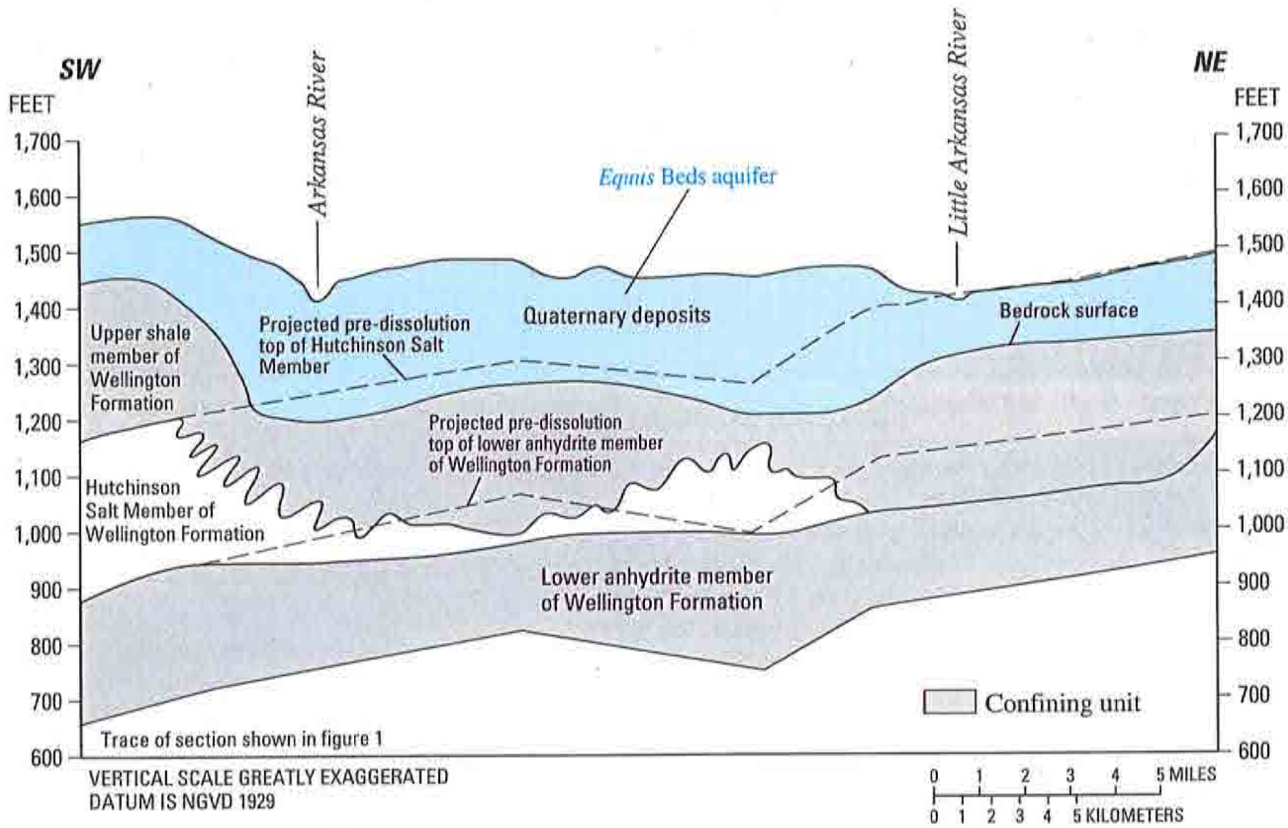


Figure 3. Generalized geologic section (from Leonard and Kleinschmidt, 1976; Myers and others, 1996).

area; the hydrograph of well 886 serves as a representative descriptor of historical water-level changes in an area of maximum water-level decline near the historic center of pumping by the city of Wichita in the central part of the study area.

In 1993, the period of general water-level rise—shown by the hydrographs of water levels in both wells 104 and 886 (fig. 2*C*)—began with greater-than-average precipitation (fig. 2*A*). An important factor in the water-level rise in well 886 was decreased city pumpage from the aquifer that accompanied increased city use of Cheney Reservoir as a water-supply source (Ross and others, 1997) (fig. 2*B*). As a result, city pumpage from the aquifer in the study area went from being greater than one-half to slightly less than one-third of Wichita's water usage during 1993–2002 (fig. 2*B*). This shifting of water sources was a part of the city of Wichita's Integrated Local Water Supply Plan (Warren and others, 1995).

Generally greater-than-average precipitation and thus increased recharge since 1993 and through 2000 may account for part of the rise in water levels seen in wells 104 and 886 during this period. The resulting water levels in 2000 were similar to levels measured in well 886 in the late 1970s (fig. 2*C*). Following 2000 and until January 2003, water levels in wells 104 and 886 generally declined or remained relatively stable (fig. 2*C*). Whether this represents the end of the period of general water-level rise that began in 1993 is not clear at this time (2003). The lack of continued water-level rises in 2001 and 2002 probably is due to decreases in precipitation (fig. 2*A*) and increases in irrigation pumpage (fig. 2*B*). Precipitation generally was less during 2000–02 than during the 5 preceding years (fig. 2*A*), and irrigation pumpage in the study area, which was less than or similar to city pumpage during 1989–97 and slightly greater than city pumpage during 1999–2000, was almost double city pumpage in 2002 (fig. 2*B*). The consistently large seasonal water-level variations in well 104 probably are due to irrigation pumpage. Irrigation water-use amounts reported prior to 1989 are not plotted in figure 2*B* because of incomplete reporting of water-use data before 1989 (Lane Letourneau, Kansas Department of Agriculture, Division of Water Resources, oral commun., August 2, 2000). Estimated ground-water use for irrigation in the study area in 2002 was less than what is permitted by the State of Kansas (fig. 2*B*); thus, increased irrigation water use in the study area could become a factor during dry years.

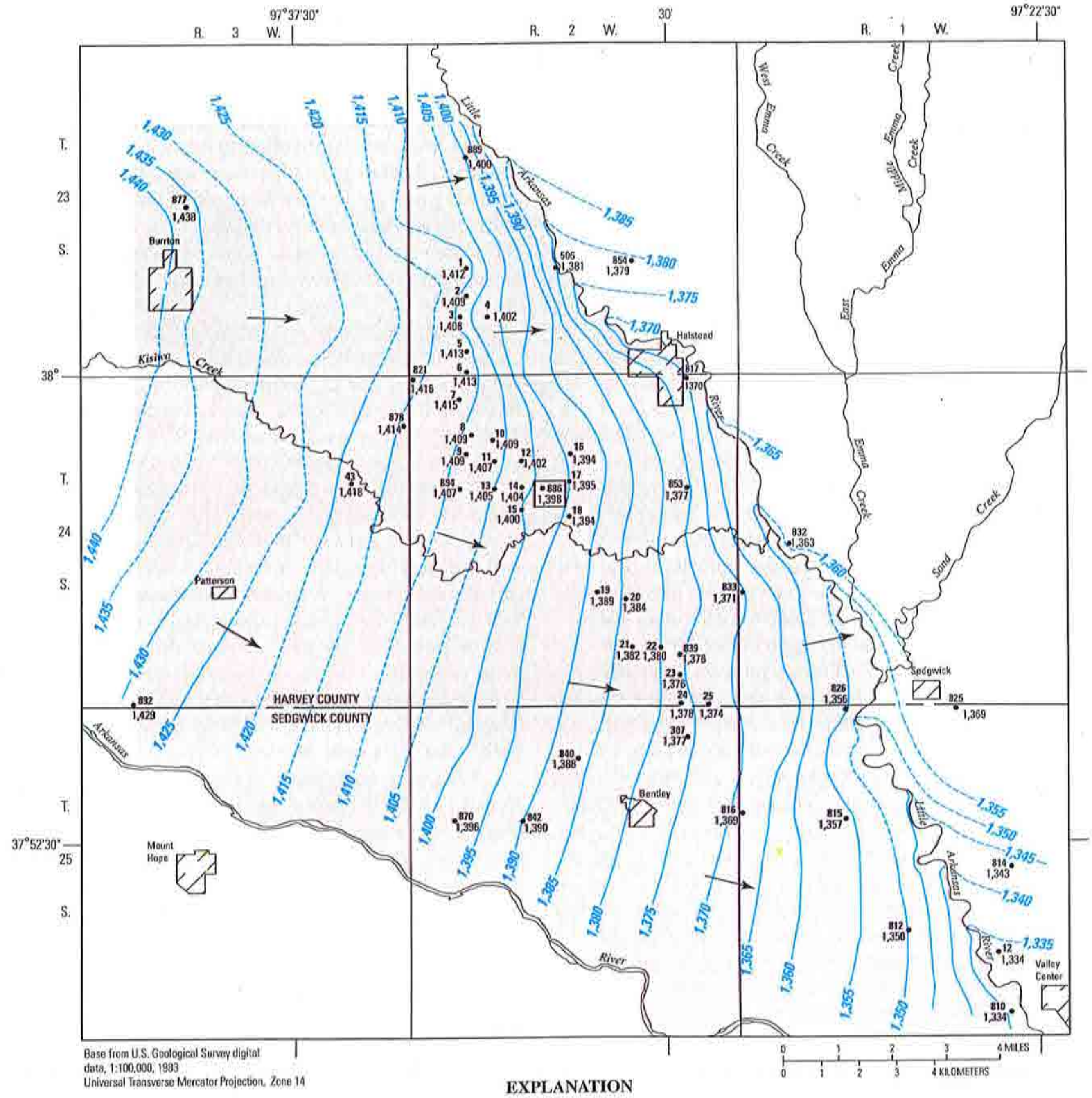
The use of hydrographs along with the use of water-level-altitude and water-level-change maps and tables and graphs of changes in storage volume can provide a more complete picture of changes in hydrologic conditions than the use of just one of these graphic tools. Hydrographs of individual wells are important for indicating changes at a specific time and can be used to infer the effects of water-level changes at that point. Such effects could include dewatered shallow wells or increased pumping costs to lift water from greater depths. Water-level-altitude maps show the gradient and direction of ground-water flow over a large area at a particular time. A single water-level-altitude map cannot indicate the distribution and extent of areas affected by water-level declines or rises. However, water-level-change maps can be used to illustrate the areal distribution and extent of water-level declines and rises. Tables and graphs showing changes in storage volume, which are derived from water-level-change maps and represent a decrease (or increase) in the ground-water resource available for use, are a good measure of the cumulative effects of pumping and climatic conditions on the aquifer.

Water-level-altitude maps for August 1940, October 1992, April 2000, and January 2003 (figs. 4, 5, 6, and 7) were constructed from available water-level data from wells to illustrate water-level conditions in the study area at selected times. Figures 6 and 7 include average daily surface-water-level altitude measurements computed for selected days from data automatically collected by equipment at U.S. Geological Survey gaging stations on the Little Arkansas River. No gaging stations on the Arkansas River are in the study area. Figure 7 also includes water-level-altitude measurements from the areal index wells. In most cases the measured water levels and computed water-level altitudes and declines for each pair of shallow and deep areal index wells were similar to each other and to nearby wells, indicating that the aquifer is well connected hydraulically in those areas. Where significant differences occur between water levels in an areal-index-well pair, the shallow and deep parts of the aquifer are less well connected hydraulically due to semiconfined conditions. In these areas, the water-level altitudes and declines were used from the well of the areal-index-well pair that indicated a better hydraulic connection to the part of the aquifer to which nearby wells are open than did the other well in the pair. Water-level altitudes and declines from deep areal index wells IW–1, IW–2, and IW–4 were used because they were similar to those of nearby wells indicating that the aquifer was better connected hydraulically between these deep areal index wells and the nearby wells than between the shallow areal index wells and the nearby wells. In all other areal-index-well pairs, the water-level altitudes and declines in the shallow wells were used because they indicated that the aquifer was as well or better connected hydraulically between the shallow areal index wells and the nearby wells than between the deep areal index wells and the nearby wells.

Figures 4, 5, 6, and 7, respectively, illustrate conditions during predevelopment (August 1940), record low water levels in October 1992, maximum recovery to date following the record low water levels (April 2000), and current conditions (January 2003). Prior to pumpage from the *Equus* Beds aquifer in 1940, near-predevelopment conditions existed in the study area (Williams and Lohman, 1949; Aucott and Myers, 1998). The August 1940 water-level-altitude map from Stramel (1956) that was modified by Aucott and Myers (1998) (fig. 4) shows that ground water flowed generally from west to east and discharged to the Little Arkansas River. Water-level-altitude maps for August 1940 and January 1955 (Stramel, 1956); for January 1957, January 1970, January 1993, and January 1998 (Aucott and Myers, 1998); for January 1997 (Aucott and others, 1998); for October 1992 and January 2000 (Hansen and Aucott, 2001); and for April 2000 and January 2003 (figs. 6 and 7) indicate that, following development, ground-water flow remained from west to east, but that between the central part of the study area and the Little Arkansas River and in the vicinity of Halstead and Sedgwick, the flow generally became more southerly and more parallel to the river. Flow in the aquifer in the central part of the study area was less southerly during January 2003 than in April 2000 (compare figs. 6 and 7), which probably was due to the continued decrease in city pumpage in the study area during this period.

Water-level change maps were constructed from available water-level data to show changes between August 1940 (predevelopment) and quarter-year intervals from January 2000 to January 2003 (figs. 8–20). In constructing these maps, if a 1940 water-level measurement did not exist for a well in the study area, one was interpolated from the August 1940 water-level altitude map (fig. 4). In figures 18, 19, and 20 where substantial differences between water-level-decline values shown for an areal index well and a nearby *Equus* beds historic observation well indicated improbable hydrologic conditions, the most probable value was used, and the value associated with the other well was ignored when constructing the lines of equal water-level decline. Some of these substantial differences may be due to the variability of the aquifer material over short distances or to water-level changes caused by pumping or precipitation that occurred between when the areal index wells and the rest of the wells in the study were measured.

The shapes of the water-level change contours since August 1940 for the period January 2000 to January 2003 (figs. 8–20) are similar to those published for recent years (Aucott and Myers, 1998; Aucott and others, 1998; Hansen and Aucott, 2001). Comparison of figures 8–20 shows the annual cycle of water-level declines and rises that generally occurs in the study area. Typically, the largest water-level declines occur during summer or fall when agricultural irrigation and city pumpage are greatest. This is shown most distinctly by the expansion of areas with water-level declines of 20 or more feet since 1940 during 2000–02 in the months of July and October (figs. 10, 11, 14, 15, 18, and 19) as compared to the areas with declines of 20 or more feet during 2000–03 in the months of January and April (figs. 8, 9, 12, 13, 16, 17, and 20). The maximum water-level decline since August 1940 for the period January 2000 to January 2003 was 29.54 ft in July 2002 at well 3 in the northern part of the study area (fig. 18). As vegetation and human water use decrease following the summer months, so does agricultural irrigation and city pumpage, resulting in water-level rises that can continue into the following spring. The maps of water-level changes since August 1940 for the period January 2000 to January 2003 show these water-level rises most obviously as the decrease in the size or disappearance of the areas with declines of 20 ft or more during January and April (figs. 8, 9, 12, 13, 16, 17, and 20). The maximum water-level rise since August 1940 for the period January 2000 to



EXPLANATION

Study area

—1,375— Water-level contour—Shows altitude of water level, August 1940. Dashed where approximately located. Contour interval 5 feet. Datum is NGVD 1929

→ Approximate direction of ground-water flow

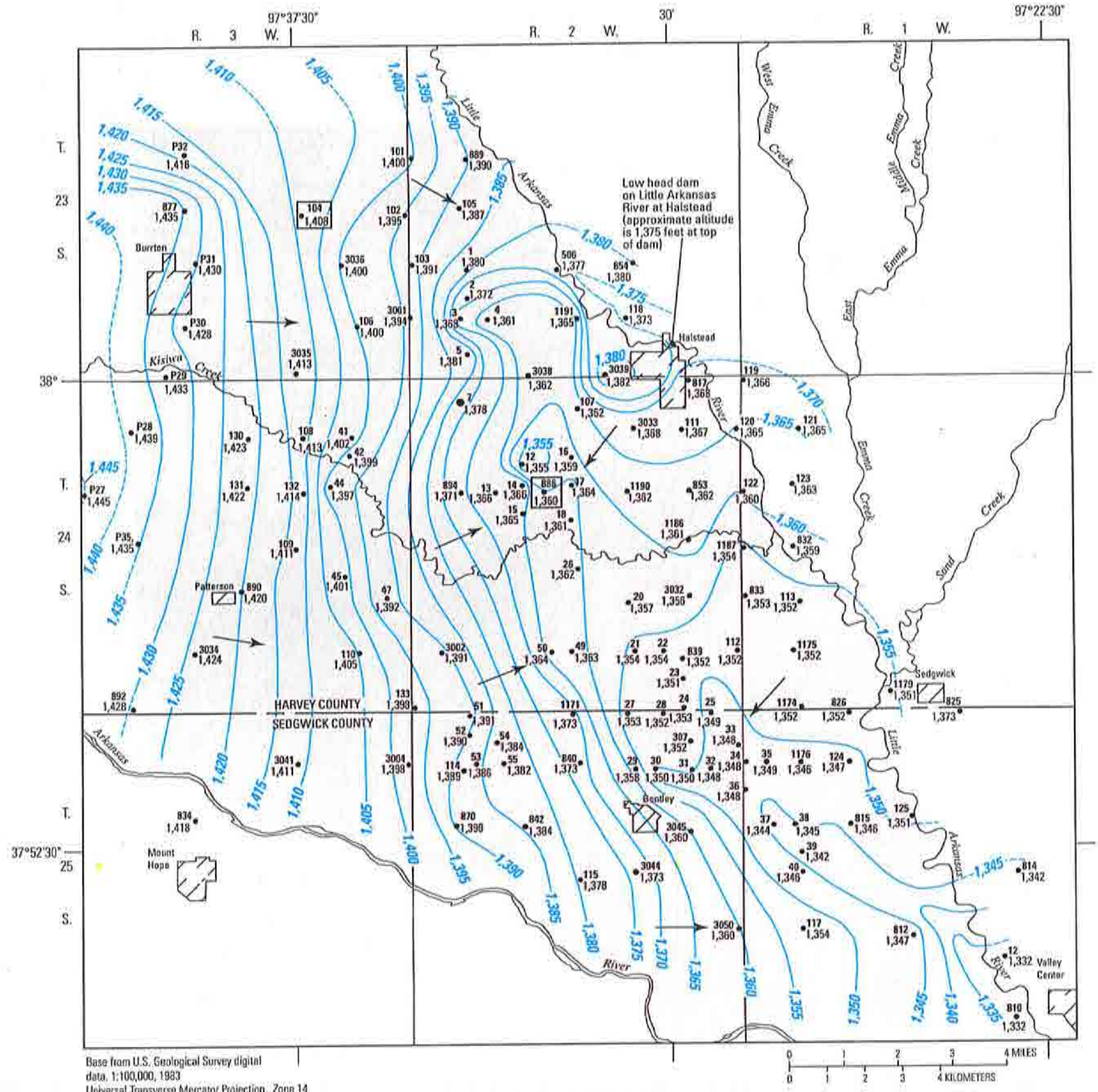
886 / 1,398 Well hydrograph shown in figure 2

1,412 (1) • *Equus* beds historic observation well

Upper number is well identification number. Lower number is water-level altitude, in feet above NGVD 29.

Water-level measurements were made by city of Wichita personnel.

Figure 4. Water-level altitudes in *Equus* Beds aquifer in study area for August 1940 (modified from Stramel, 1956, and Aucott and Myers, 1998).



EXPLANATION

Study area

—1,375— Water-level contour—Shows altitude of water level, October 1992. Dashed where approximately located. Contour interval 5 feet. Datum is NGVD 29

→ Approximate direction of ground-water flow

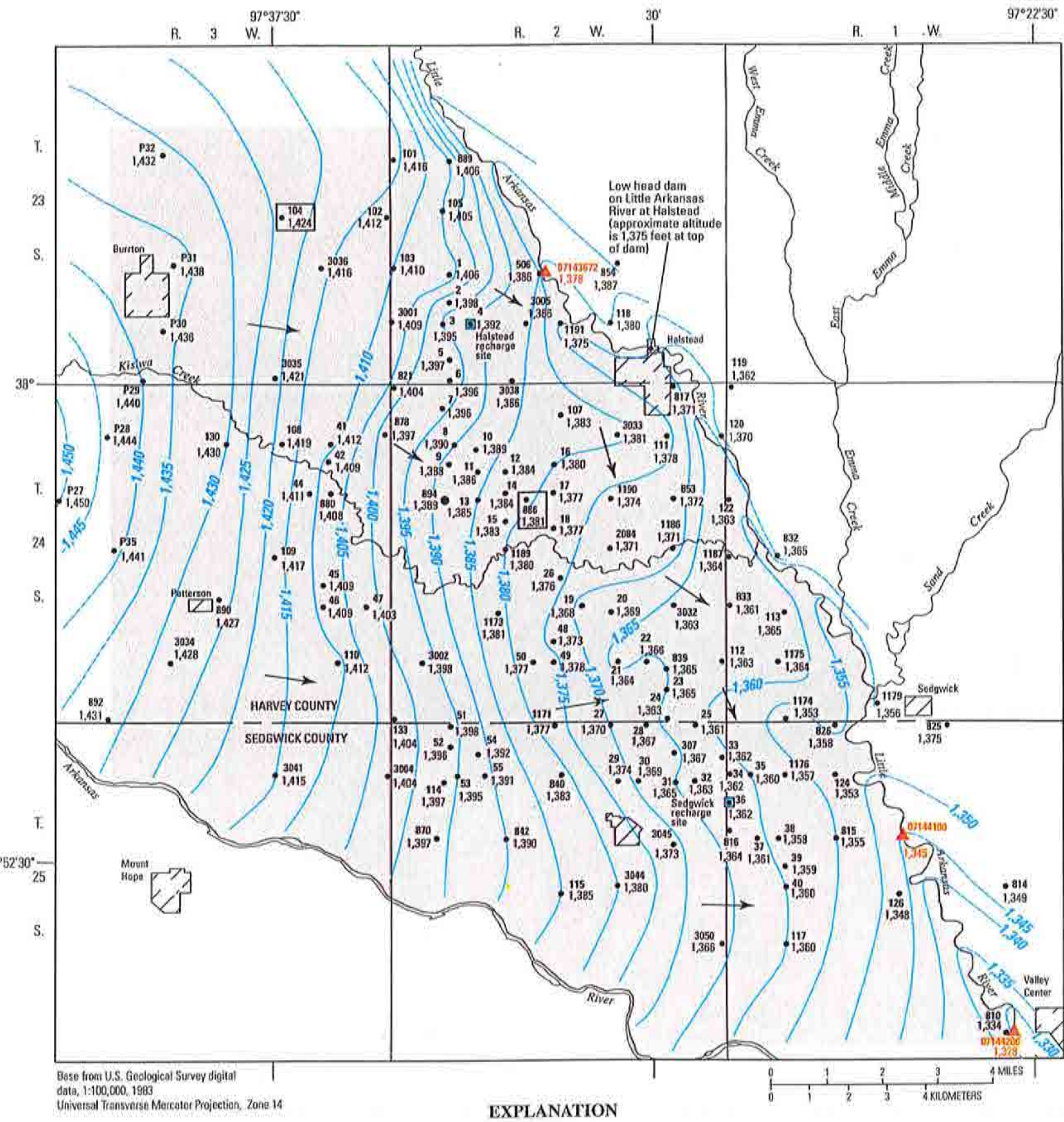
886 / 1,360 Well hydrograph shown in figure 2

1 / 1,380 *Equus* beds historic observation well

Upper number is well identification number. Lower number is water-level altitude, in feet above NGVD 29.

Water-level measurements were made by city of Wichita personnel.

Figure 5. Water-level altitudes in *Equus* Beds aquifer in study area for October 1992 (modified from Hansen and Aucott, 2001).



EXPLANATION

Study area

1,415 **Water-level contour**—Shows altitude of water level, April 2000. Dashed where approximately located. Contour interval 5 feet. Datum is NAVD 88

Approximate direction of ground-water flow

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

07143672 1,378 **Gaging station**—Upper number is station number; lower number is water-level altitude, in feet above NAVD 88

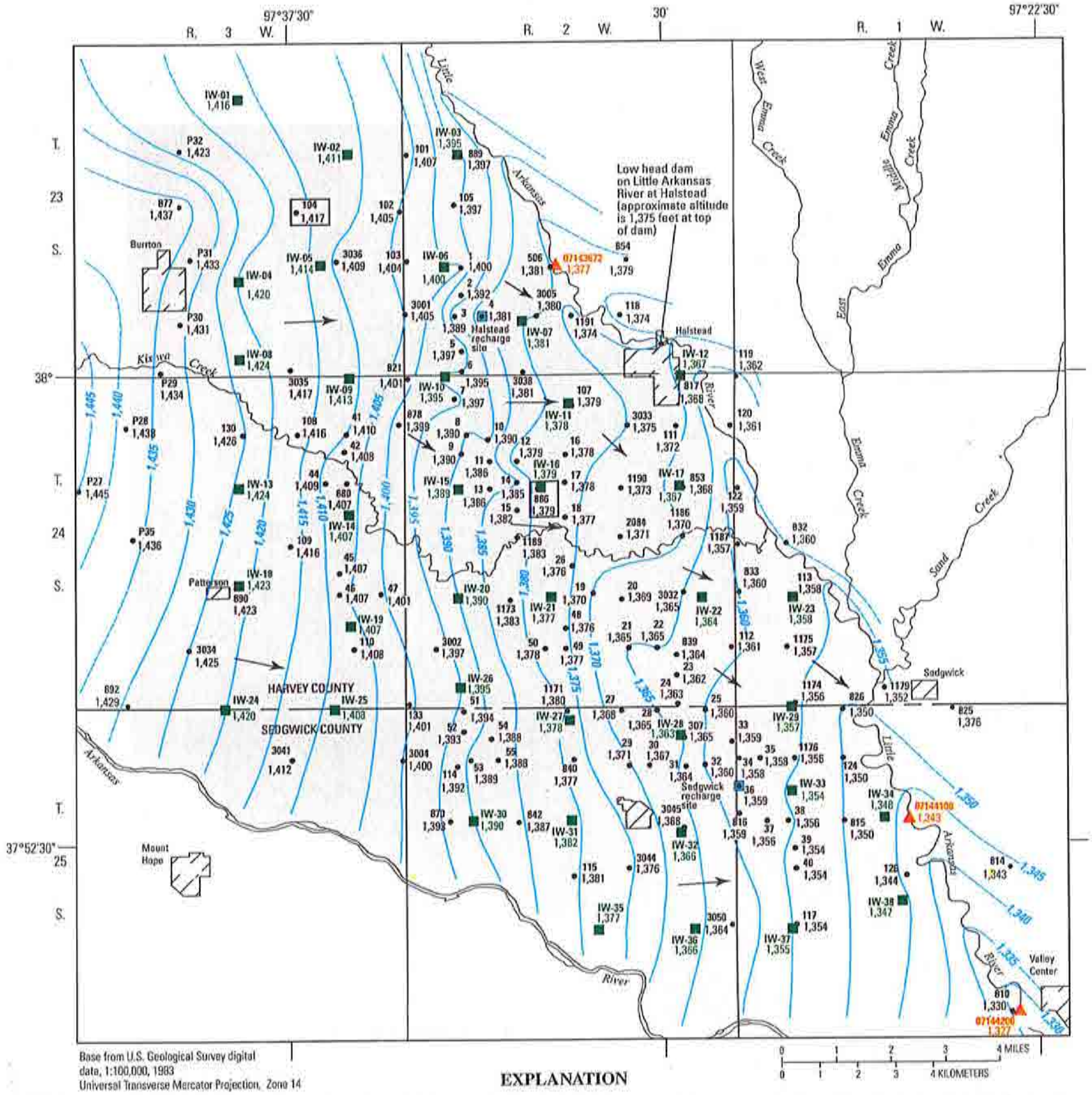
886 1,381 **Well hydrograph shown in figure 2**

3050 1,366 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level altitude, in feet above NAVD 88.

Water-level measurements in wells were made by city of Wichita personnel and *Equus* Beds Groundwater Management District No. 2 personnel. Water-level measurements at gaging stations were collected by U.S. Geological Survey.

Figure 6. Water-level altitudes in *Equus* Beds aquifer in study area for April 2000.



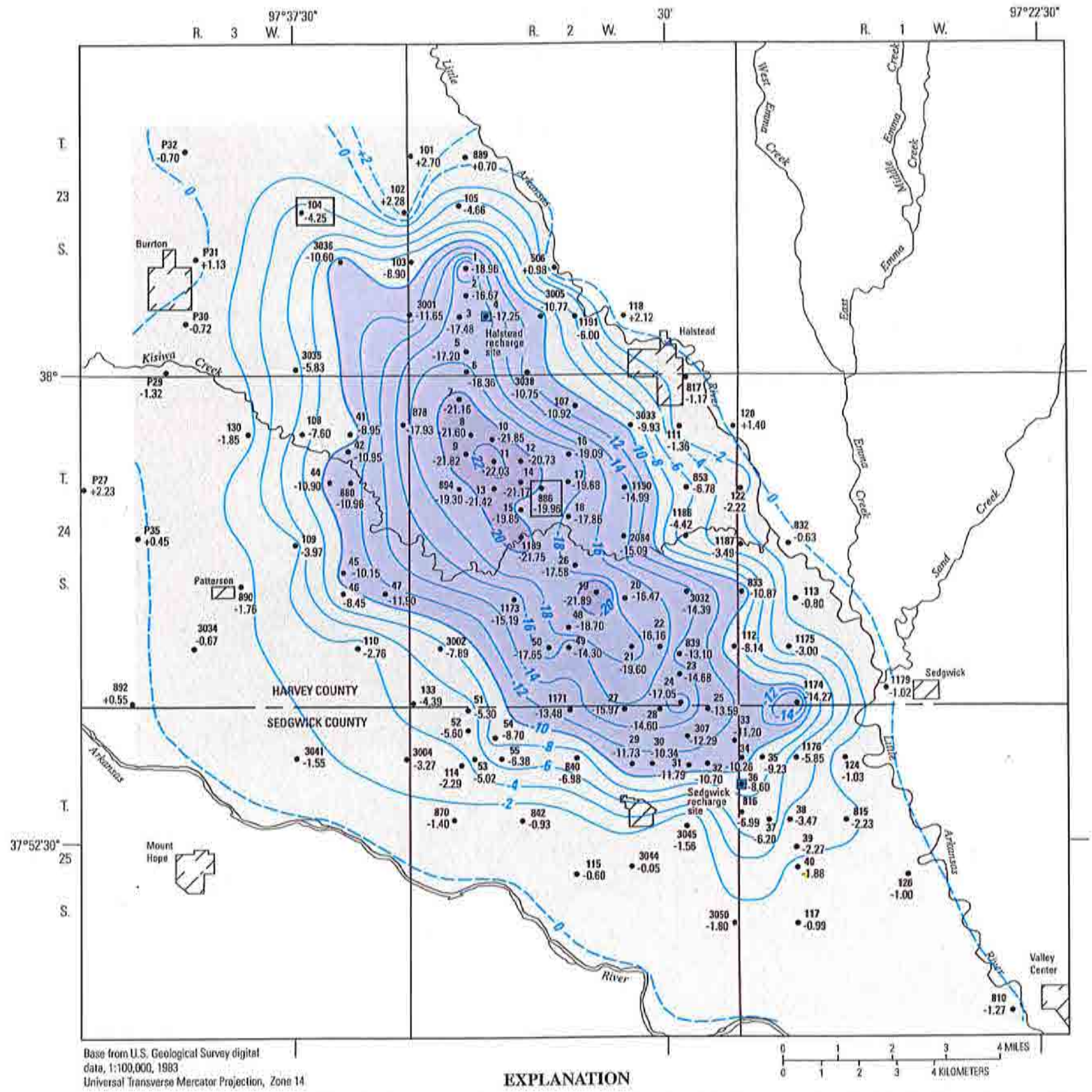
EXPLANATION

- Study area
- 1,415 — **Water-level contour**—Shows altitude of water level, January 2003. Dashed where approximately located. Contour interval 5 feet. Datum is NAVD 88
- → **Approximate direction of ground-water flow**
- **Artificial recharge demonstration site**—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000
- 07144200 / 1,327 ▲ **Gaging station**—Upper number is station number; lower number is water-level altitude, in feet above NAVD 88
- 886 / 1,379 • **Well hydrograph shown in figure 2**
- 3050 / 1,364 • ***Equus* beds historic observation well**
- IW-04 / 1,420 ■ ***Equus* beds areal index well**

Upper number is well identification number. Lower number is water-level altitude, in feet above NAVD 88.

Water-level measurements in wells were made by city of Wichita personnel and *Equus* Beds Groundwater Management District No. 2 personnel. Water-level measurements at gaging stations were collected by U.S. Geological Survey.

Figure 7. Water-level altitudes in *Equus* Beds aquifer in study area for January 2003.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—*-10*— **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -19.96 **Well hydrograph shown in figure 2**

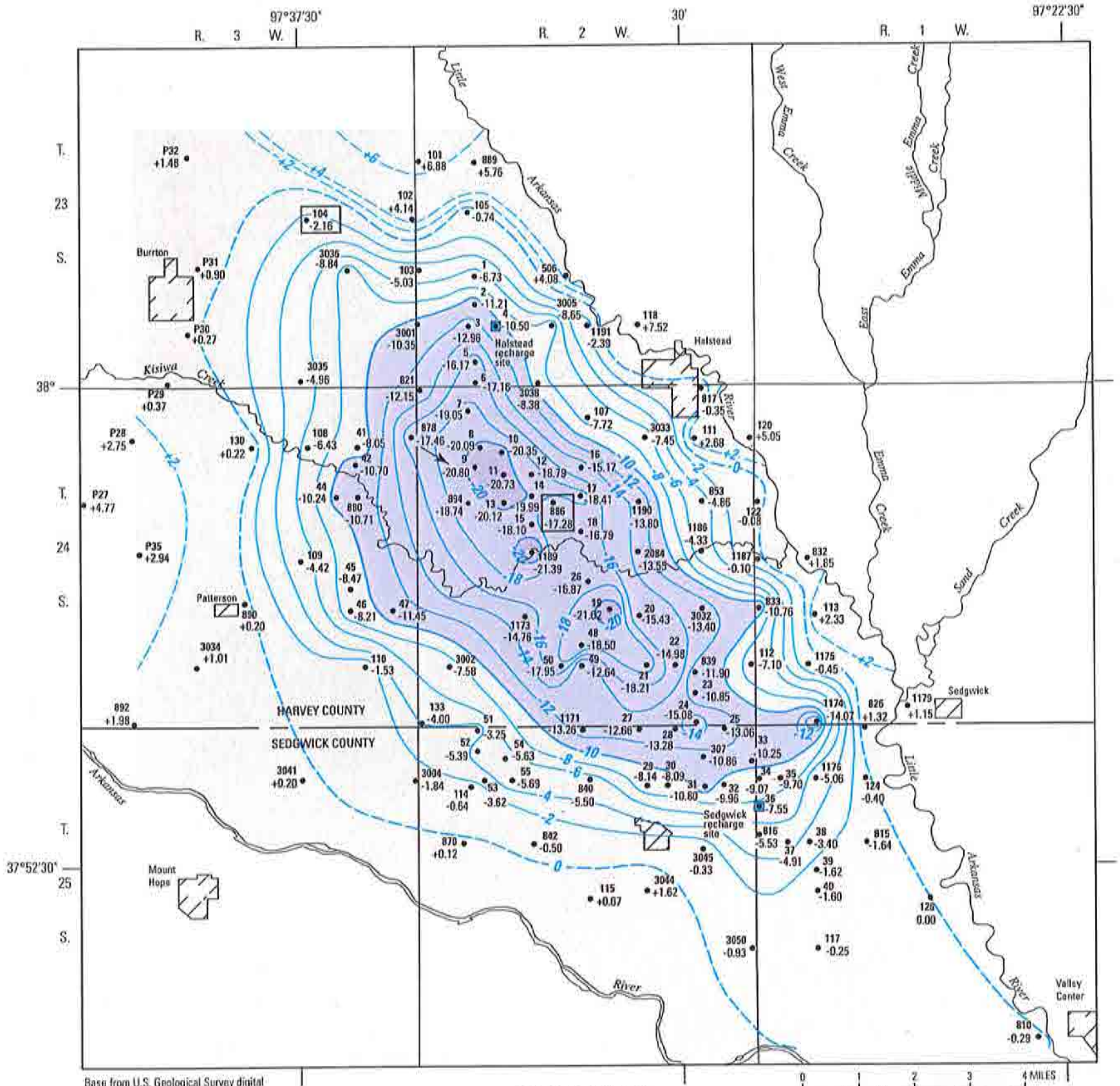
1 -18.96 *Equus* **beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and January 2000 was 126,000 acre-feet.

Figure 8. Water-level change in *Equus* Beds aquifer in study area, August 1940–January 2000 (from Hansen and Aucott, 2001).



Base from U.S. Geological Survey digital data, 1:100,000, 1983
Universal Transverse Mercator Projection, Zone 14

EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

-10 — **Line of equal change of water level—** Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site— Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -17.28 **Well hydrograph shown in figure 2**

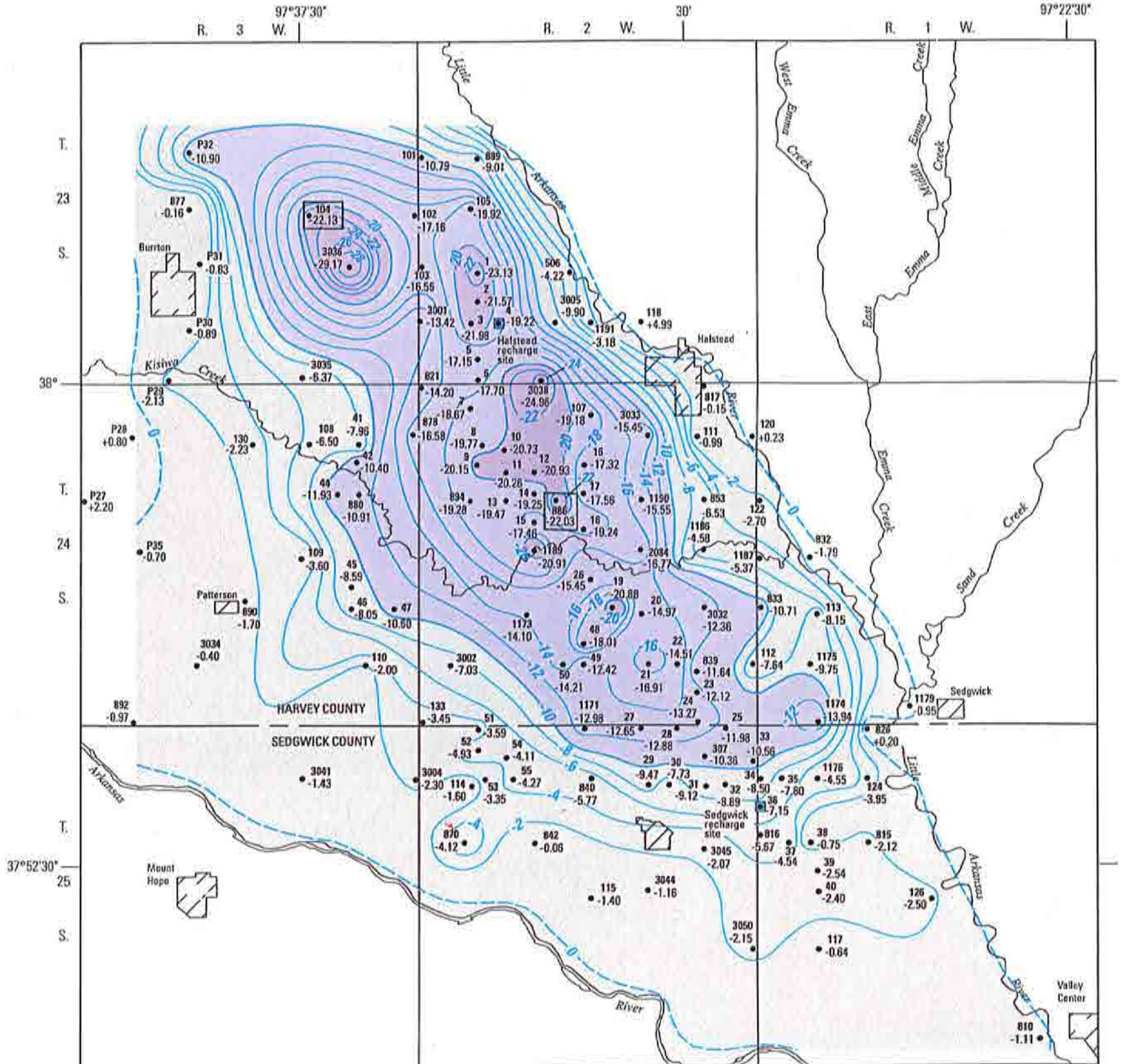
1 -6.73 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and April 2000 was 101,000 acre-feet.

Figure 9. Water-level change in *Equus* Beds aquifer in study area, August 1940–April 2000.



Base from U.S. Geological Survey digital data, 1:100,000, 1983
Universal Transverse Mercator Projection, Zone 14

0 1 2 3 4 MILES
0 1 2 3 4 KILOMETERS

EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

— -10 — **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -22.03 **Well hydrograph shown in figure 2**

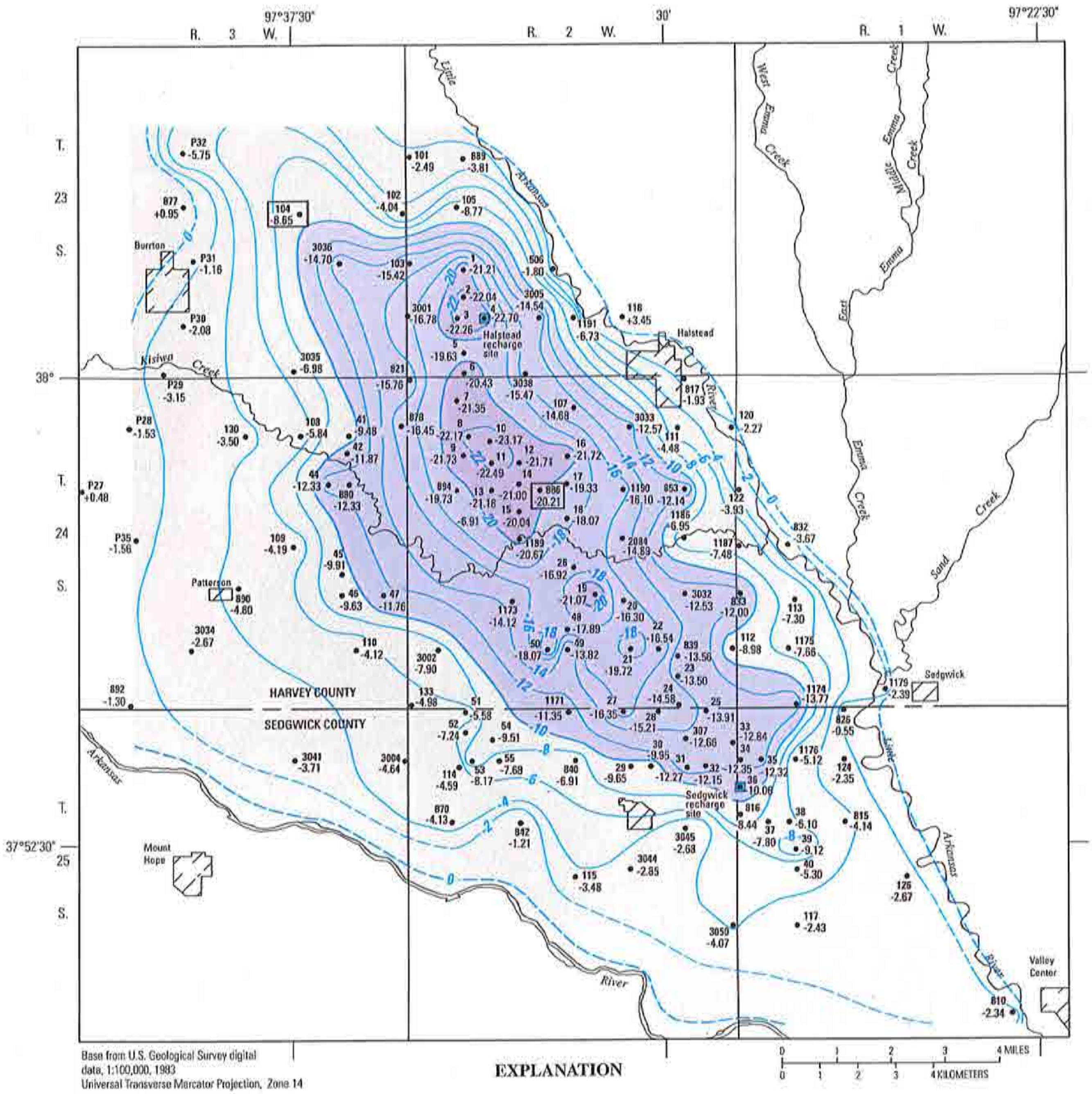
1 -23.13 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and July 2000 was 152,000 acre-feet.

Figure 10. Water-level change in *Equus* Beds aquifer in study area, August 1940–July 2000.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—*10*— **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / -20.21 **Well hydrograph shown in figure 2**

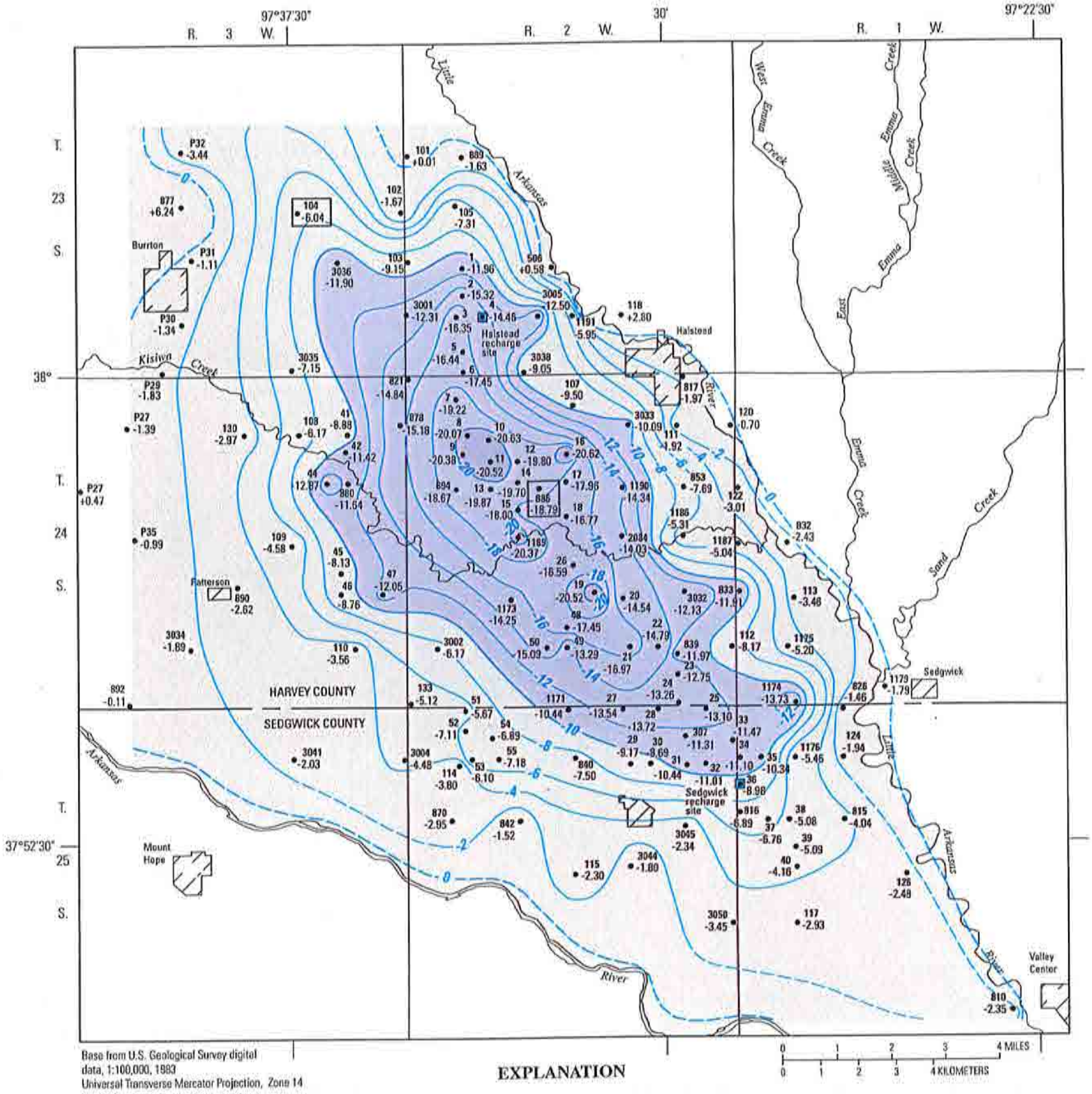
1 / -21.21 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and October 2000 was 159,000 acre-feet.

Figure 11. Water-level change in *Equus* Beds aquifer in study area, August 1940–October 2000.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

-10 **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / -18.79 **Well hydrograph shown in figure 2**

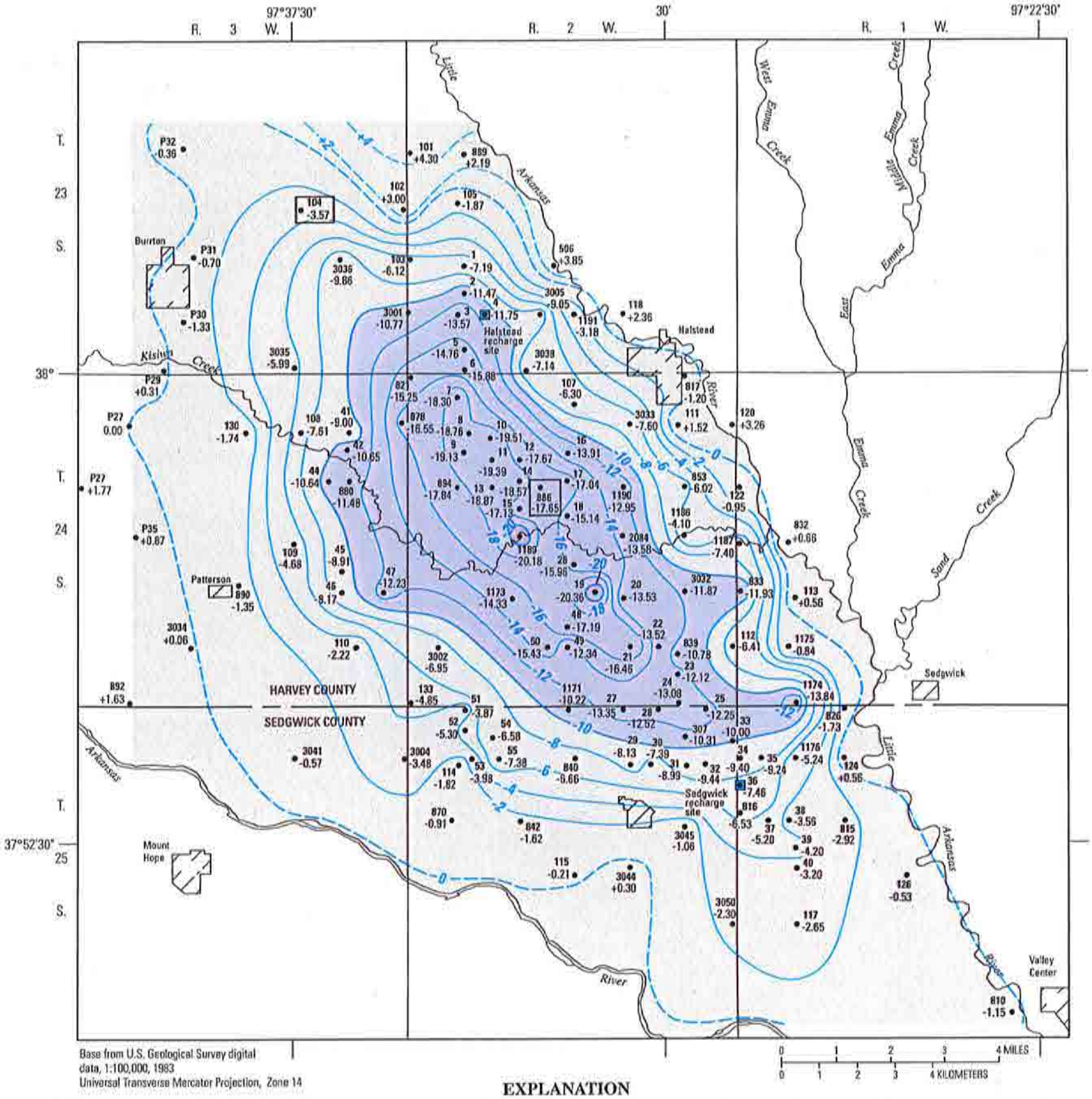
1 / -11.96 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and January 2001 was 134,000 acre-feet.

Figure 12. Water-level change in *Equus* Beds aquifer in study area, August 1940–January 2001.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—10— — Line of equal change of water level—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / -17.65 **Well hydrograph shown in figure 2**

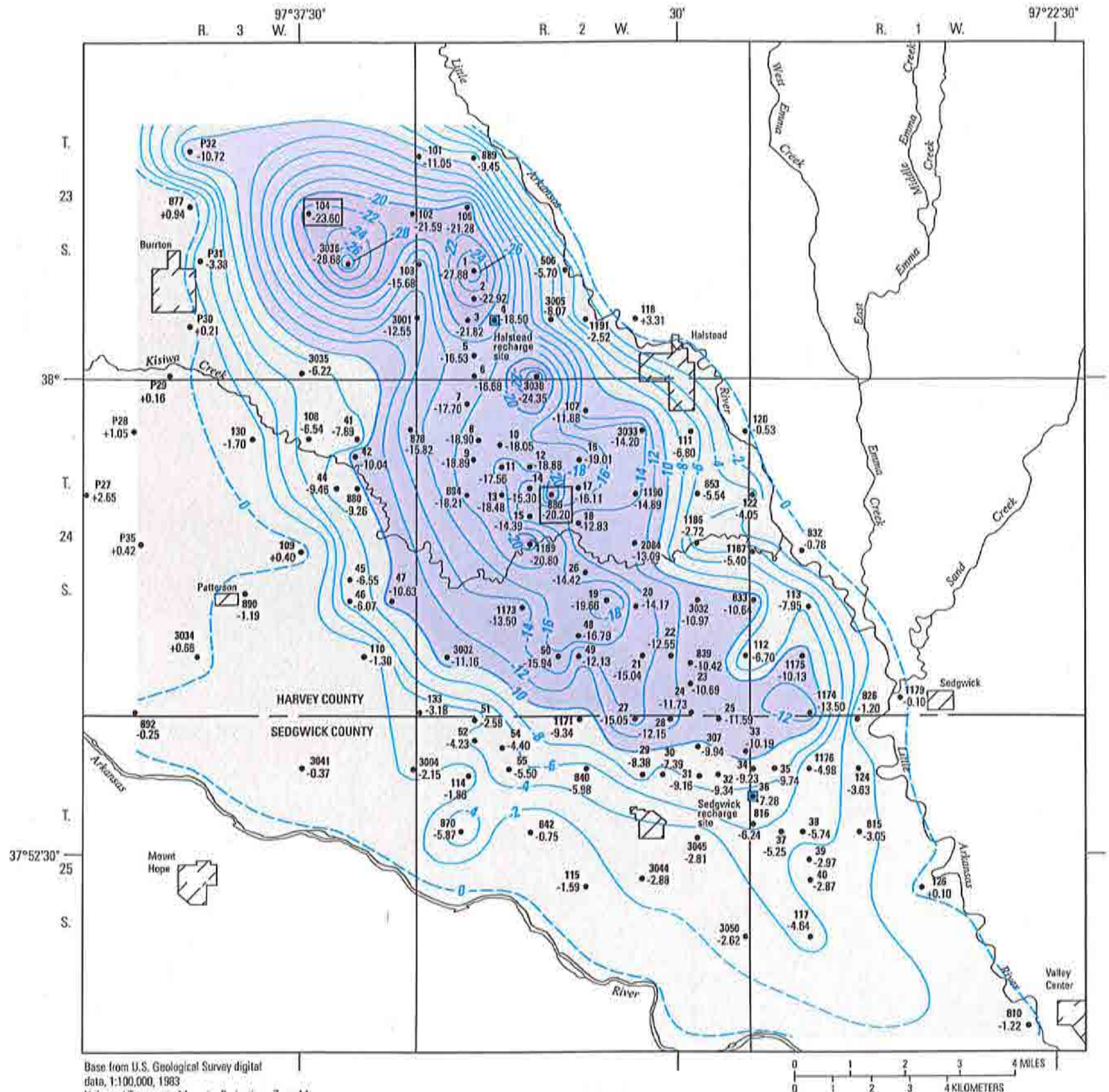
1 / -7.19 *Equus* **beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and April 2001 was 110,000 acre-feet.

Figure 13. Water-level change in *Equus* Beds aquifer in study area, August 1940–April 2001.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

-10 Line of equal change of water level—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -20.20 Well hydrograph shown in figure 2

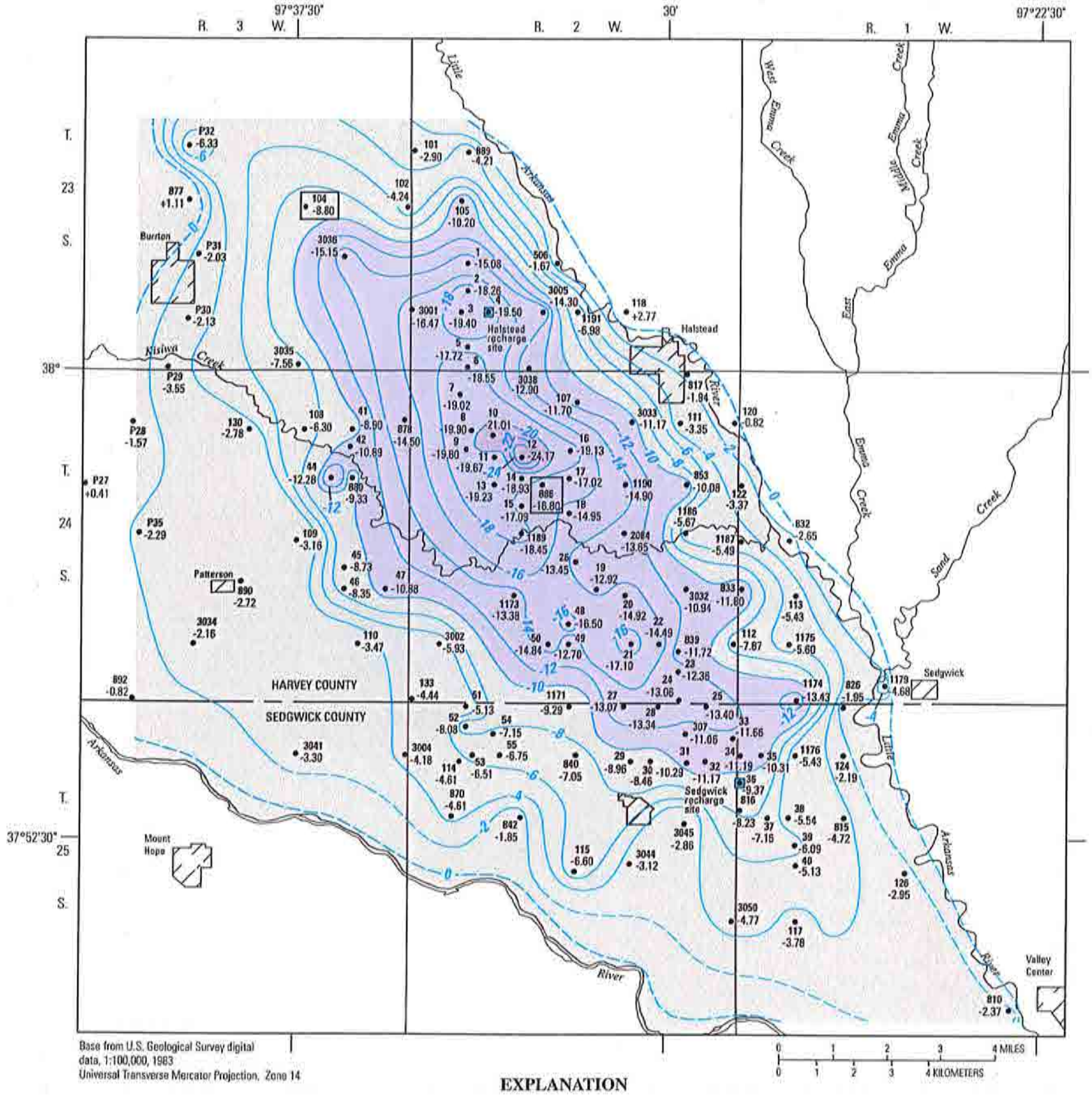
1 -27.88 *Equus* beds historic observation well

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and July 2001 was 149,000 acre-feet.

Figure 14. Water-level change in *Equus* Beds aquifer in study area, August 1940–July 2001.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—— -10 —— Line of equal change of water level— Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site— Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -16.80 **Well hydrograph shown in figure 2**

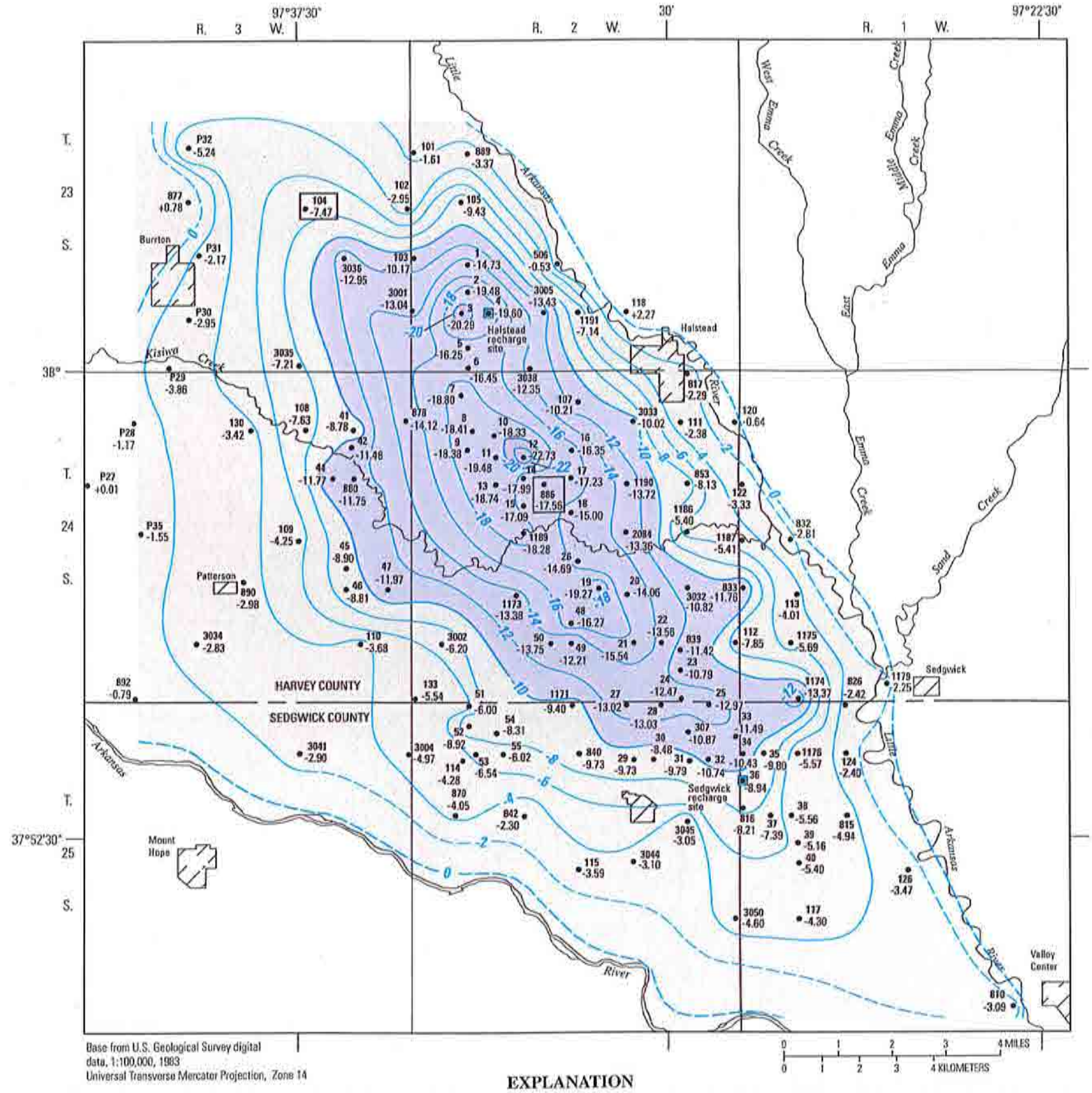
1 -15.08 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and October 2001 was 146,000 acre-feet.

Figure 15. Water-level change in *Equus* Beds aquifer in study area, August 1940–October 2001.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

-10 — **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -17.56 **Well hydrograph shown in figure 2**

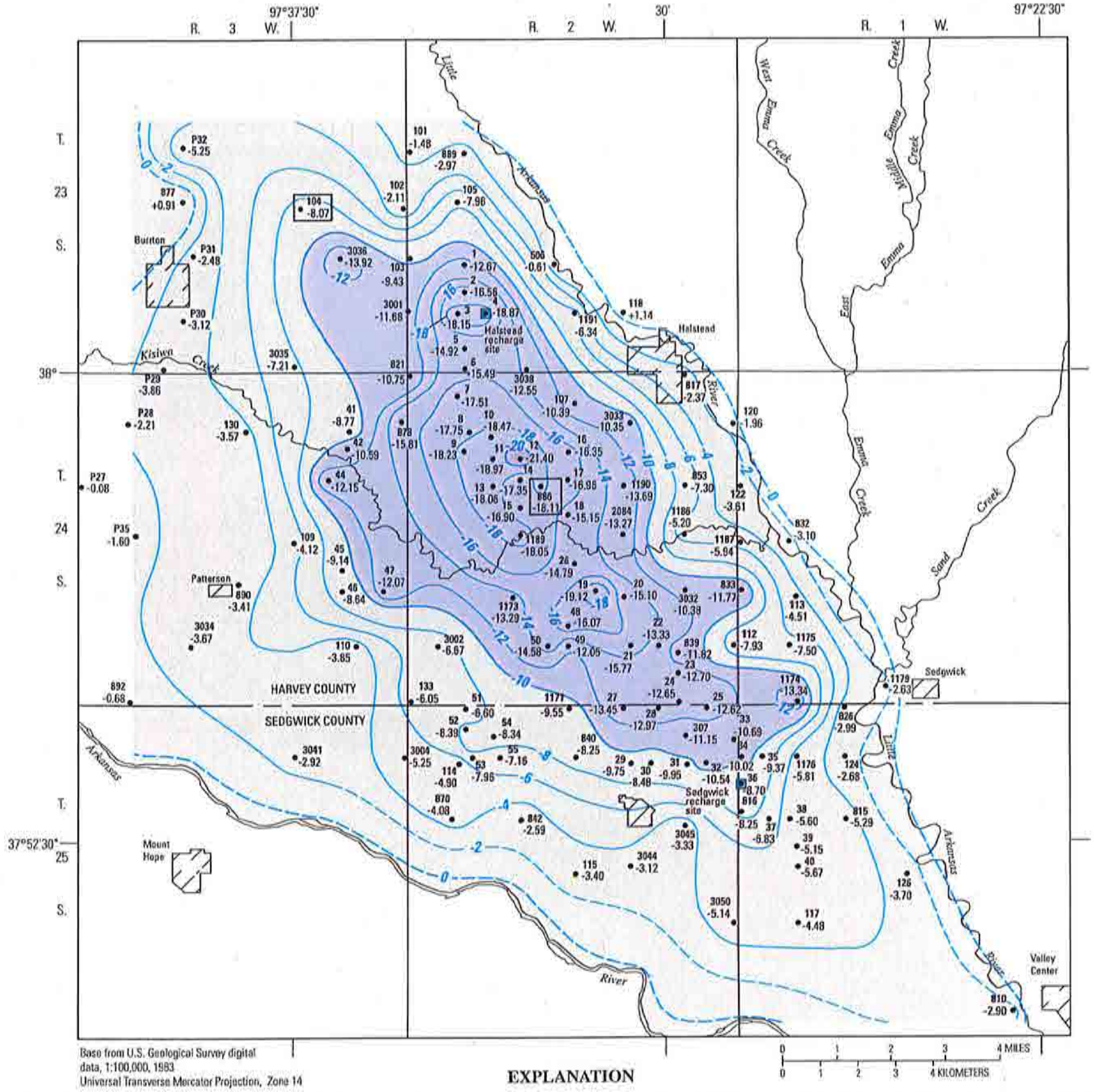
1 -14.73 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and January 2002 was 142,000 acre-feet.

Figure 16. Water-level change in *Equus* Beds aquifer in study area, August 1940–January 2002.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—10— — **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -18.11 **Well hydrograph shown in figure 2**

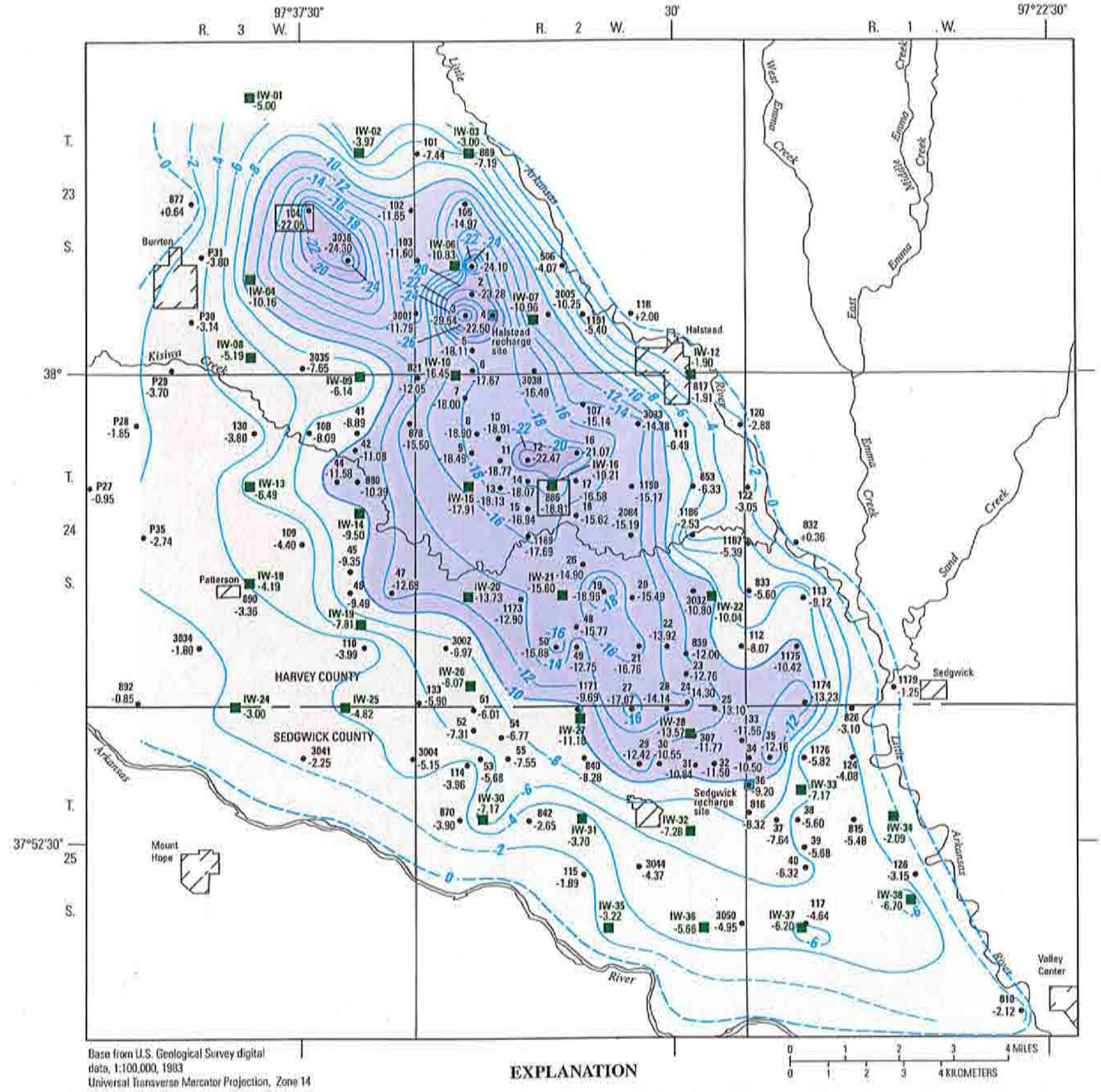
1 -12.67 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between August 1940 and April 2002 was 141,000 acre-feet.

Figure 17. Water-level change in *Equus* Beds aquifer in study area, August 1940–April 2002.



EXPLANATION

Study area

Area of water-level decline, in feet
- 10 – 20
- 20 – 30

-10 Line of equal change of water level—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / -18.81 Well hydrograph shown in figure 2

1 / -24.10 *Equus* beds historic observation well

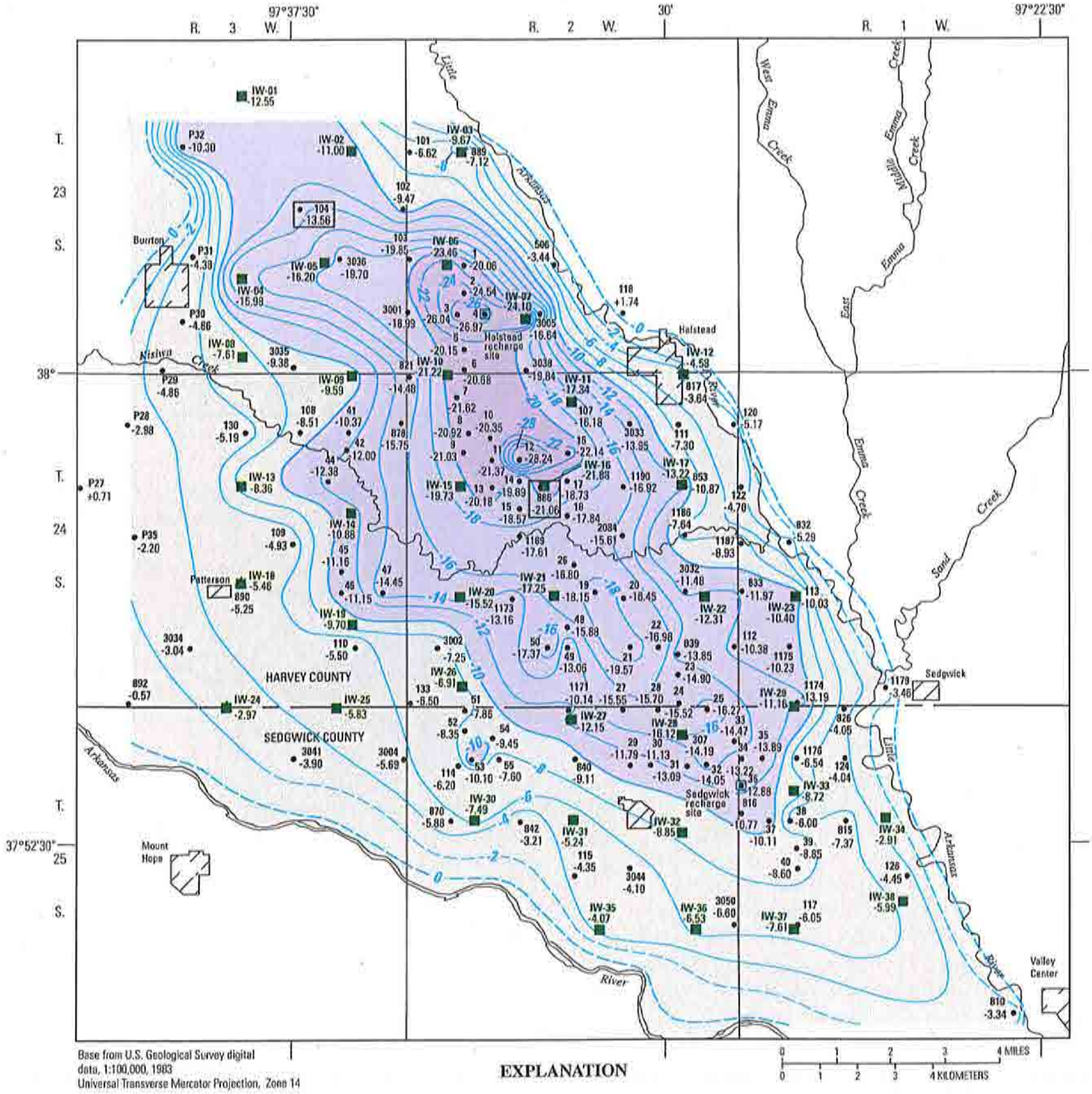
IW-01 / -5.00 *Equus* beds areal index well

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita and U.S. Geological Survey personnel.

Depletion of storage volume in study area between August 1940 and July 2002 was 162,000 acre-feet.

Figure 18. Water-level change in *Equus* Beds aquifer in study area, August 1940–July 2002.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—10— Line of equal change of water level—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -21.06 Well hydrograph shown in figure 2

1 -20.06 *Equus* beds historic observation well

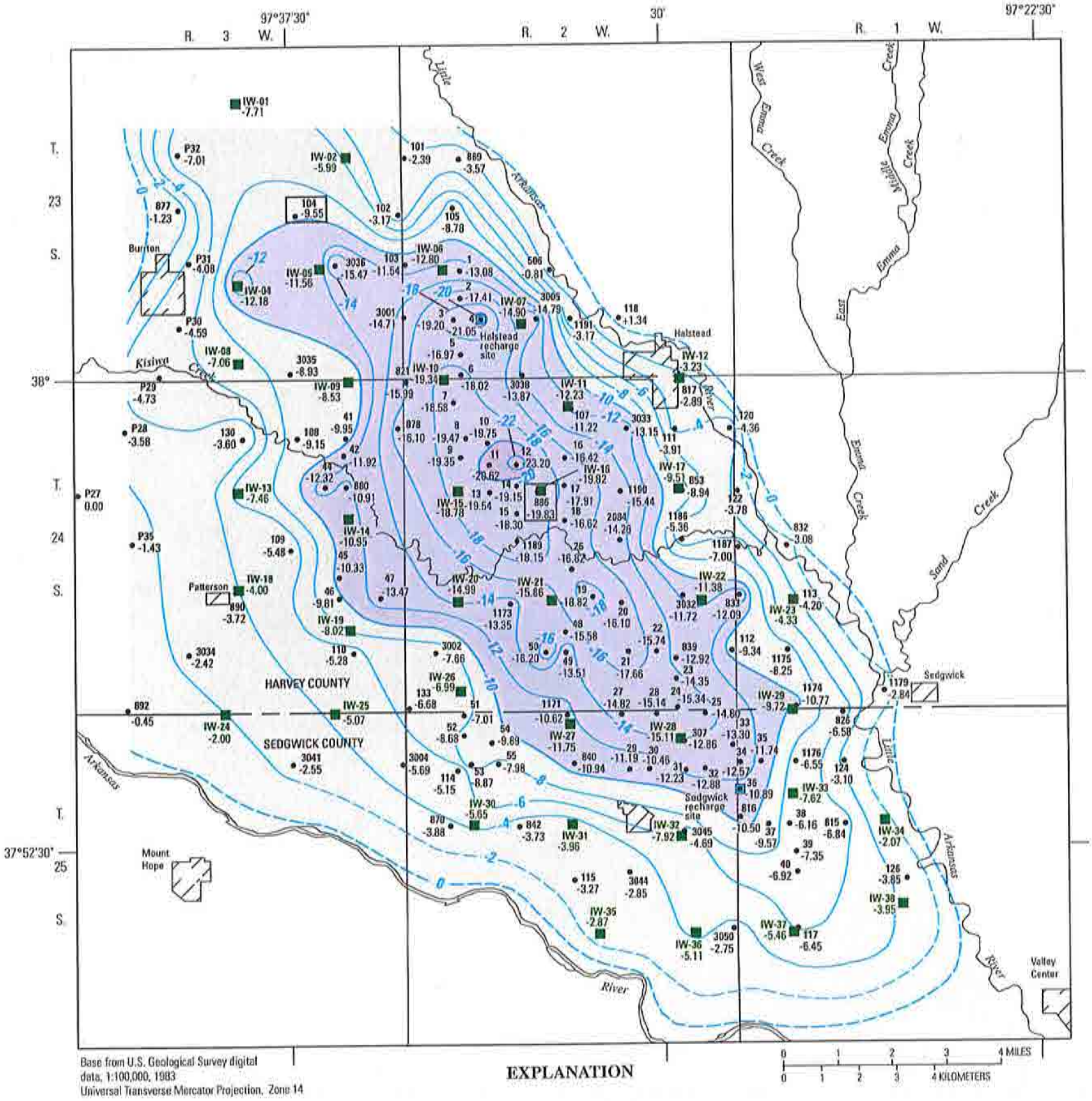
IW-01 -12.55 *Equus* beds areal index well

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita and U.S. Geological Survey personnel.

Depletion of storage volume in study area between August 1940 and October 2002 was 187,000 acre-feet.

Figure 19. Water-level change in *Equus* Beds aquifer in study area, August 1940–October 2002.



EXPLANATION

Study area

Area of water-level decline, in feet

10 – 20

20 – 30

—*-10*— **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 -19.83 **Well hydrograph shown in figure 2**

1 -13.08 *Equus* **beds historic observation well**

IW-01 -7.71 *Equus* **beds areal index well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was lower than August 1940.

Water-level measurements were made by city of Wichita and *Equus* Beds Groundwater Management District No. 2 personnel.

Depletion of storage volume in study area between August 1940 and January 2003 was 159,000 acre-feet.

Figure 20. Water-level change in *Equus* Beds aquifer in study area, August 1940–January 2003.

January 2003 in the study area was 6.88 ft in April 2000 at well 101 in the northern part of the study area (fig. 9).

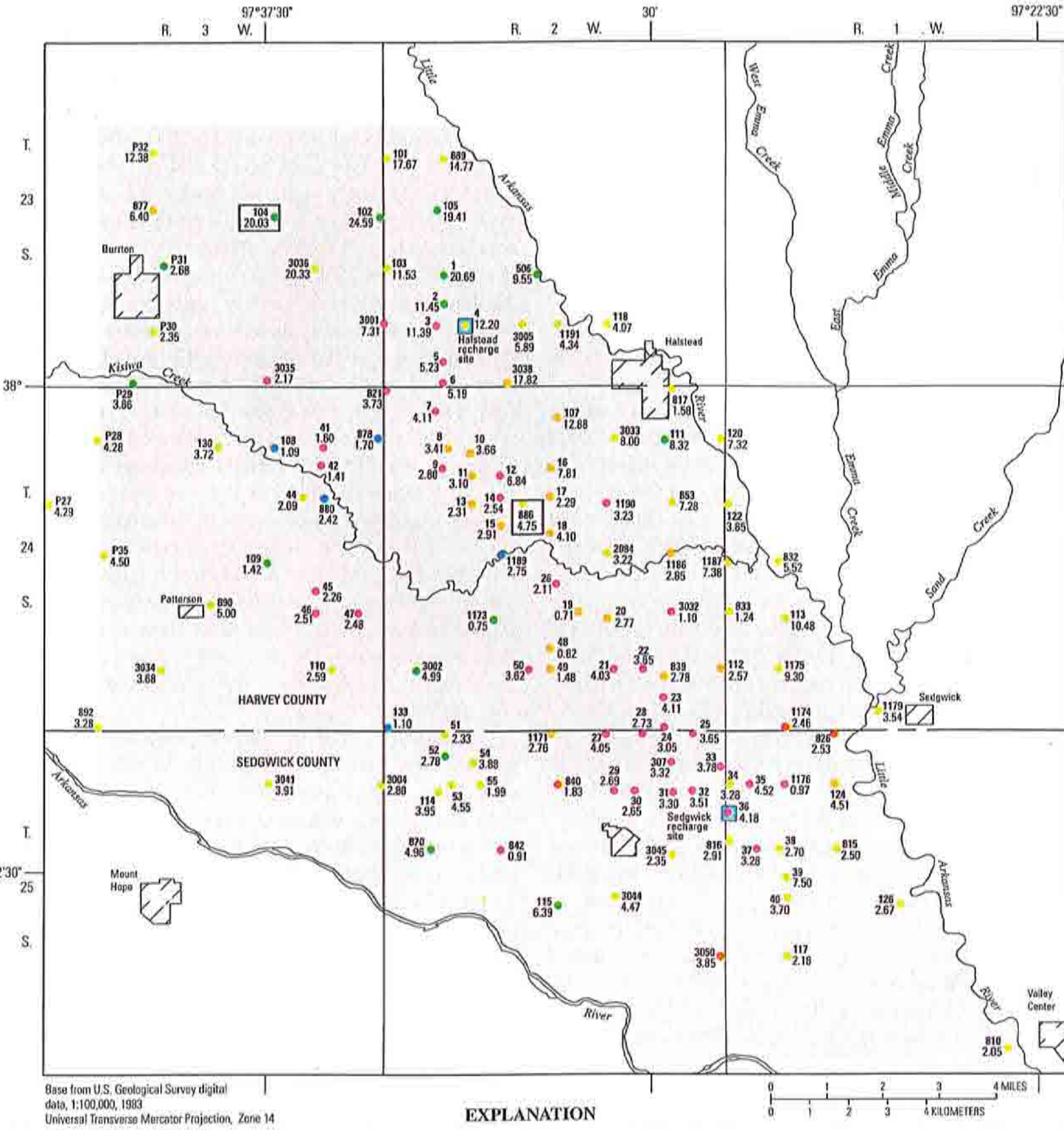
Comparisons of figures 8–20 indicate that the period with the smallest area of water-level declines since August 1940 for the period January 2000 to January 2003 was April 2000 (fig. 9). The expansion during the period January 2000 to January 2003 of the area with water-level declines (inside the zero contour) is best seen on the maps of water-level changes since August 1940 for April 2000, 2001, 2002 (figs. 9, 13, and 17) because water levels measured in the month of April represent the maximum recovery of water levels from low water levels caused by withdrawals during the previous summer and fall. This expansion of the area with water-level declines may be associated with decreased precipitation and increased irrigation pumpage compared to previous years (figs. 2*A*, 2*B*, and 2*C*). Comparison of maps of the water-level changes since August 1940 for the same months in different years during 2000 to 2003 (for example, figs. 8, 12, 16, and 20) shows that the size of areas with water-level declines of 20 to 30 ft decreased through July 2002; these areas are generally in or near the center of the study area. This decrease in water-level declines probably is due to decreased city pumpage from the *Equus* Beds aquifer in the study area that is associated with increased city use of water from Cheney Reservoir. The size of the areas with water-level declines of 20 to 30 ft during October 2002 and January 2003 (figs. 19 and 20) is larger than the same areas during October 2001 and January 2002, respectively (figs. 15 and 16). The increased water-level declines in October 2002 and January 2003 may be due to increased irrigation pumpage during the summer and fall of 2002.

Seasonal water-level changes for most wells in the study area during the period January 2000 to January 2003 continued to be larger than the cumulative water-level change during the same period (for example, see hydrographs of observation wells 104 and 886 in fig. 2*C*). Each year was divided into two seasons—a recovery or "spring" season (represented by the months of January and April) and a decline or "fall" season (represented by the months of July and October). The shallower (larger recovery) of the year's January or April water level at each well was used for each well's "spring" water level; the deeper (larger decline) of the year's July or October water level at each well was used for the well's "fall" water level. Seasonal changes for each well were determined by subtracting one season's water level from the preceding season's water level (for example, fall 2000 subtracted from spring 2000). The absolute value of the largest seasonal change for each well and the time period when the change occurred are shown in figure 21. In some wells the largest water-level change for a 12-month period occurred within the same season; thus, the seasonal changes shown in figure 21 do not always represent the largest water-level change that occurred for all wells in any 12-month period during January 2000 to January 2003. Seasonal changes for the areal index wells are not shown in figure 21 because quarterly water-level measurements did not begin at these wells before the summer of 2002.

The maximum seasonal changes in water levels in wells in the study area typically were less than 10 ft during the period January 2000 to January 2003 (fig. 21) with most changes being less than 5 ft. Seasonal changes tended to be larger towards the edges of the study area than in the center of the study area. The largest seasonal changes occurred in the northern part of the study area with most wells having seasonal changes of more than 5 ft and changes of 10 ft or more being common (fig. 21). Wells 1, 102, 104, and 3036 in the northern part of the study area had the largest seasonal changes in the study area with changes between 20 and 25 ft during 2000 and 2001 (fig. 21). The large seasonal changes in the northern part of the study area probably are due mostly to seasonal irrigation pumpage and semiconfined aquifer conditions in this part of the study area (Aucott and Myers, 1998). These conditions also may account for the large differences in seasonal changes seen in nearby wells in this part of the study area. For example, there is a difference of about 13 ft between the maximum seasonal changes at wells 102 and 103 (fig. 21). In the central part of the study area the maximum seasonal changes commonly occurred between fall 2000 and spring 2001 or between spring 2002 and fall 2002, with the former period most common in the center and the latter period most common in the northwestern and southeastern parts of the center of the study area (fig. 21). In the rest of the study area, the maximum seasonal changes most commonly occurred between spring 2000 and fall 2000 (fig. 21).

Unusually wet or dry climatic conditions or changes in ground-water pumpage strategies may modify the annual cycle of water-level rises and declines. For example, drought conditions and increases in agricultural irrigation or city pumpage may result in a cumulative decline in ground-water levels; greater-than-average precipitation and decreases in agricultural irrigation or city pumpage may result in a cumulative recovery of ground-water levels. To show some of these cumulative changes, water-level change maps were constructed for the periods October 1992 to January 2003, October 1992 to April 2000, April 2000 to January 2003, and January 2000 to January 2003, (figs. 22–25, respectively). For figures 22–25, a water level was used for the water-level-change map for a selected period only if the measured water level was used both for the map since August 1940 to the beginning date of the selected period and for the map since August 1940 to the end date of the selected period. For example, the *Equus* Beds aquifer areal index wells were not used for figures 22–25 because none had water-level measurements before 2002 and, therefore, did not have water levels for the beginning dates of any of these maps.

As pointed out by Hansen and Aucott (2001), the maximum recorded decline in the study area occurred in October 1992; therefore, a map for the period October 1992 to January 2003 was constructed to illustrate the magnitude of cumulative water-level changes since the period of maximum decline (fig. 22). The cumulative water-level changes from October 1992 to January 2003 in the study area ranged from a decline of 2.83 ft in well 826 on the eastern edge of the study area to a rise of 24.04 ft in well 12 in the central part of the study area.



EXPLANATION

Study area

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 4.75 Well hydrograph shown in figure 2

Equus beds historic observation well

117 2.18 Spring 2000 – fall 2000

124 4.51 Fall 2000 – spring 2001

115 6.39 Spring 2001 – fall 2001

133 1.10 Fall 2001 – spring 2002

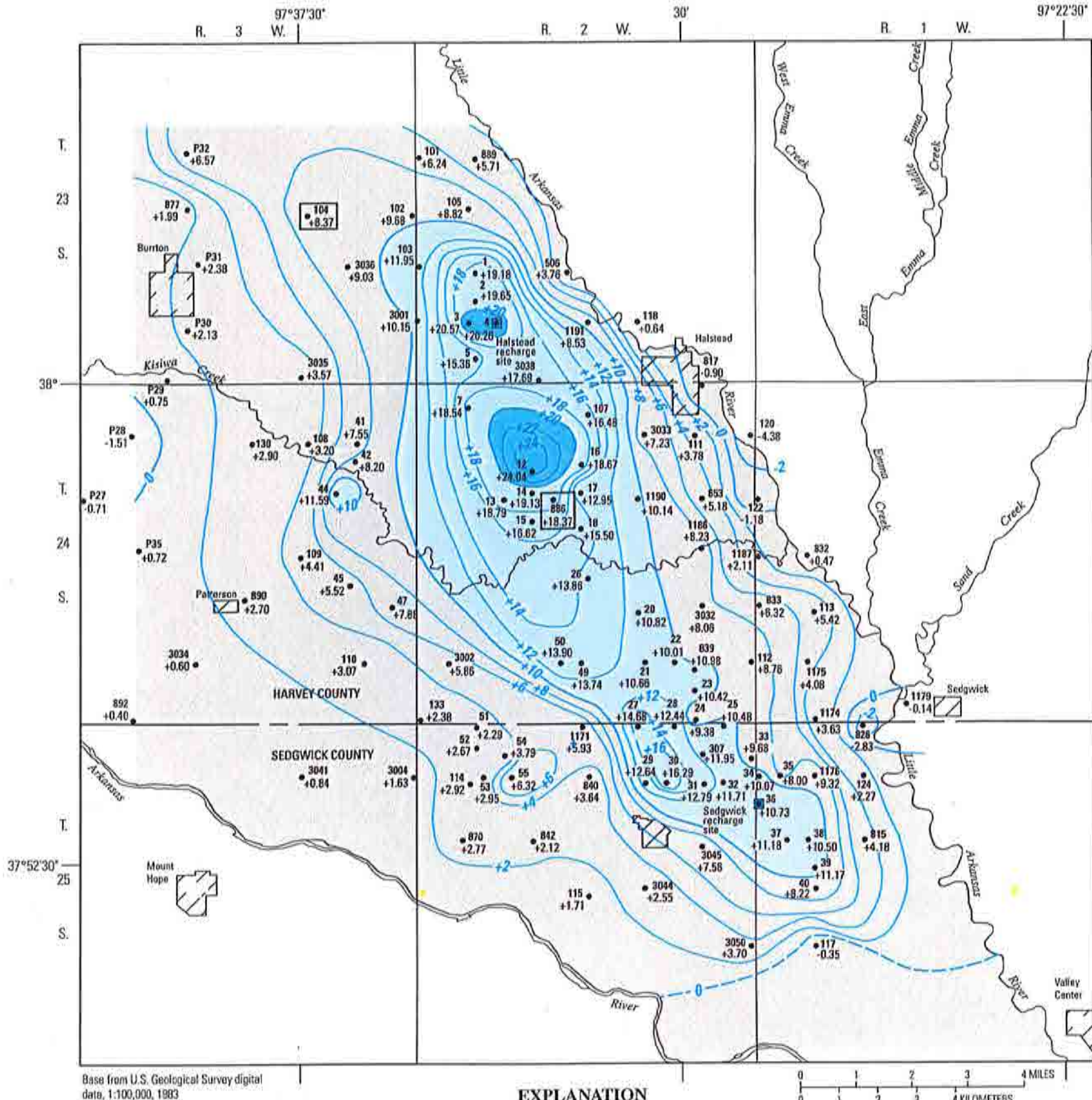
842 0.91 Spring 2002 – fall 2002

840 1.83 Fall 2002 – spring 2003

Upper number is well identification number. Lower number is absolute value of maximum seasonal water-level change during January 2000 to January 2003, in feet.

Data from water-level measurements made by city of Wichita personnel.

Figure 21. Maximum seasonal water-level changes in *Equus* Beds aquifer in study area during January 2000–January 2003.



Base from U.S. Geological Survey digital data, 1:100,000, 1983
Universal Transverse Mercator Projection, Zone 14

EXPLANATION

Study area

Area of water-level recovery, in feet

+10 to +20

+20 to +30

—10— **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 +18.37 **Well hydrograph shown in figure 2**

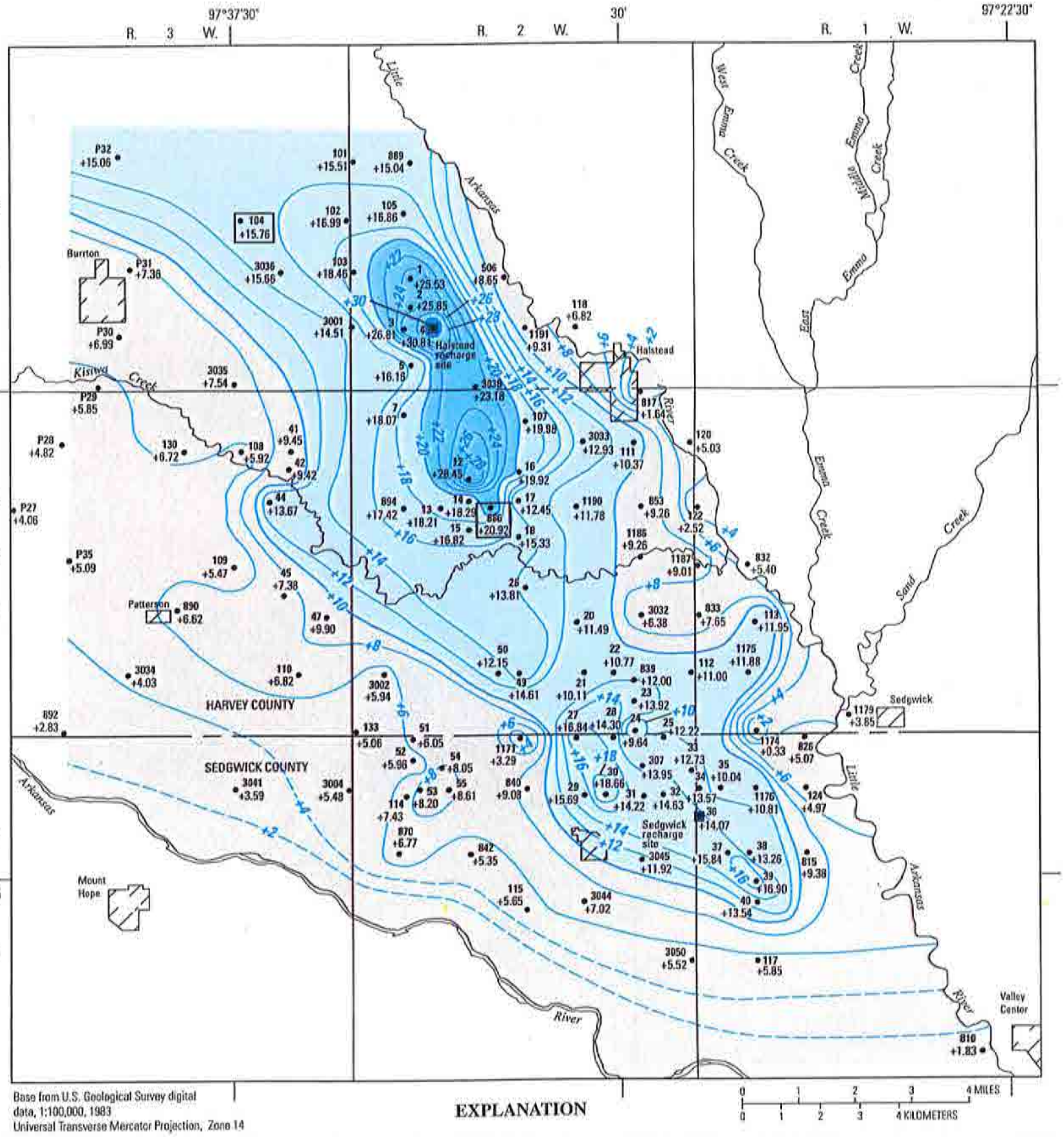
1 +19.18 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was less than in October 1992.

Water-level measurements were made by city of Wichita personnel.

Recovery of storage volume in study area between October 1992 and January 2003 was 124,000 acre-feet.

Figure 22. Water-level change in *Equus* Beds aquifer in study area, October 1992–January 2003.



EXPLANATION

Study area

Area of water-level recovery, in feet

+10 to +20

+20 to +30

Greater than +30

+10 — Line of equal change of water level—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / +20.92 Well hydrograph shown in figure 2

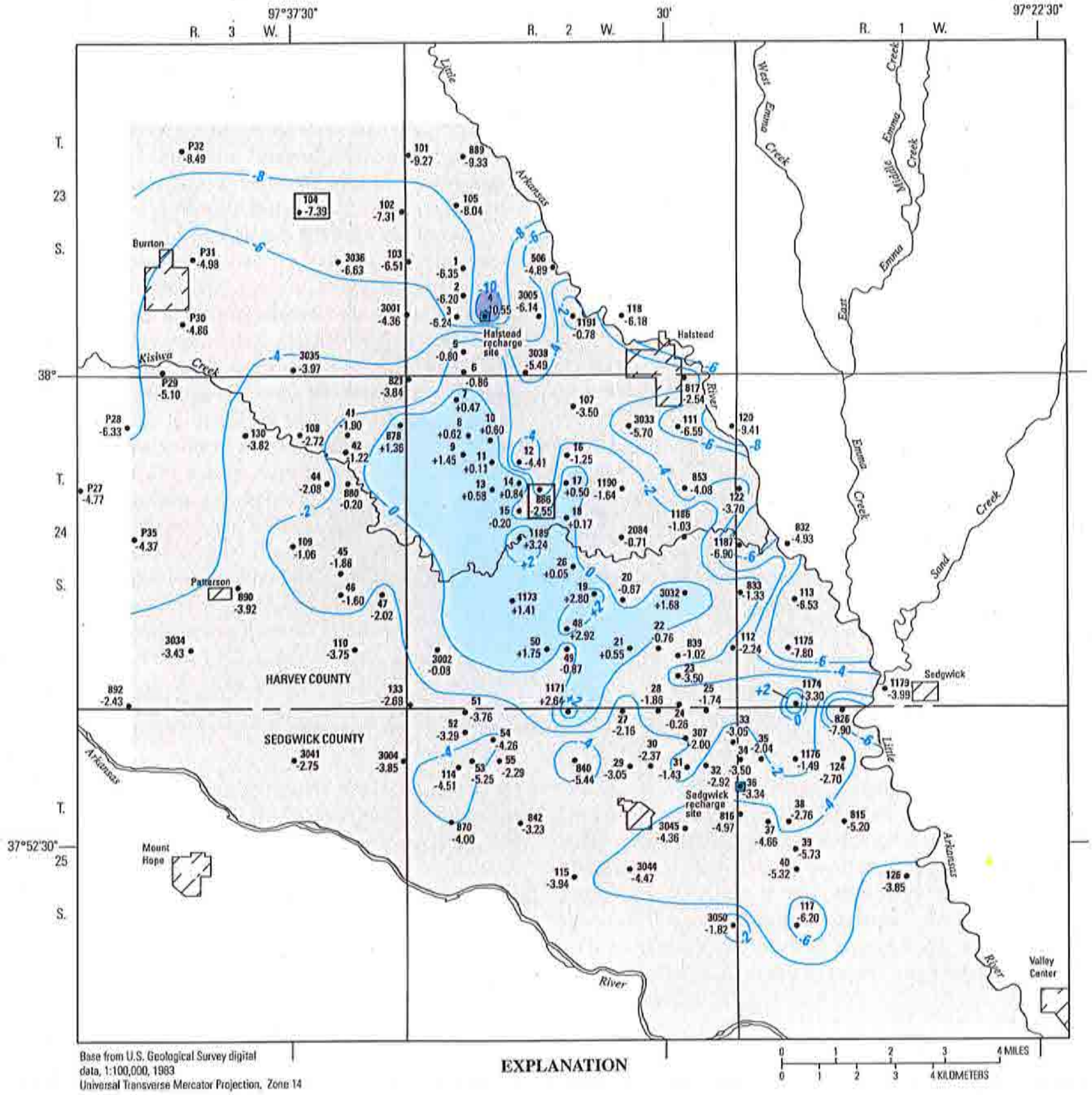
1 / +25.53 *Equus* beds historic observation well

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was less than in October 1992.

Water-level measurements were made by city of Wichita personnel.

Recovery of storage volume in study area between October 1992 and April 2000 was 182,000 acre-feet.

Figure 23. Water-level change in *Equus* Beds aquifer in study area, October 1992–April 2000.



EXPLANATION

Study area

Area of water-level recovery, in feet

0 to +10

Area of water-level decline, in feet

Greater than 10

-10 **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 / -2.55 **Well hydrograph shown in figure 2**

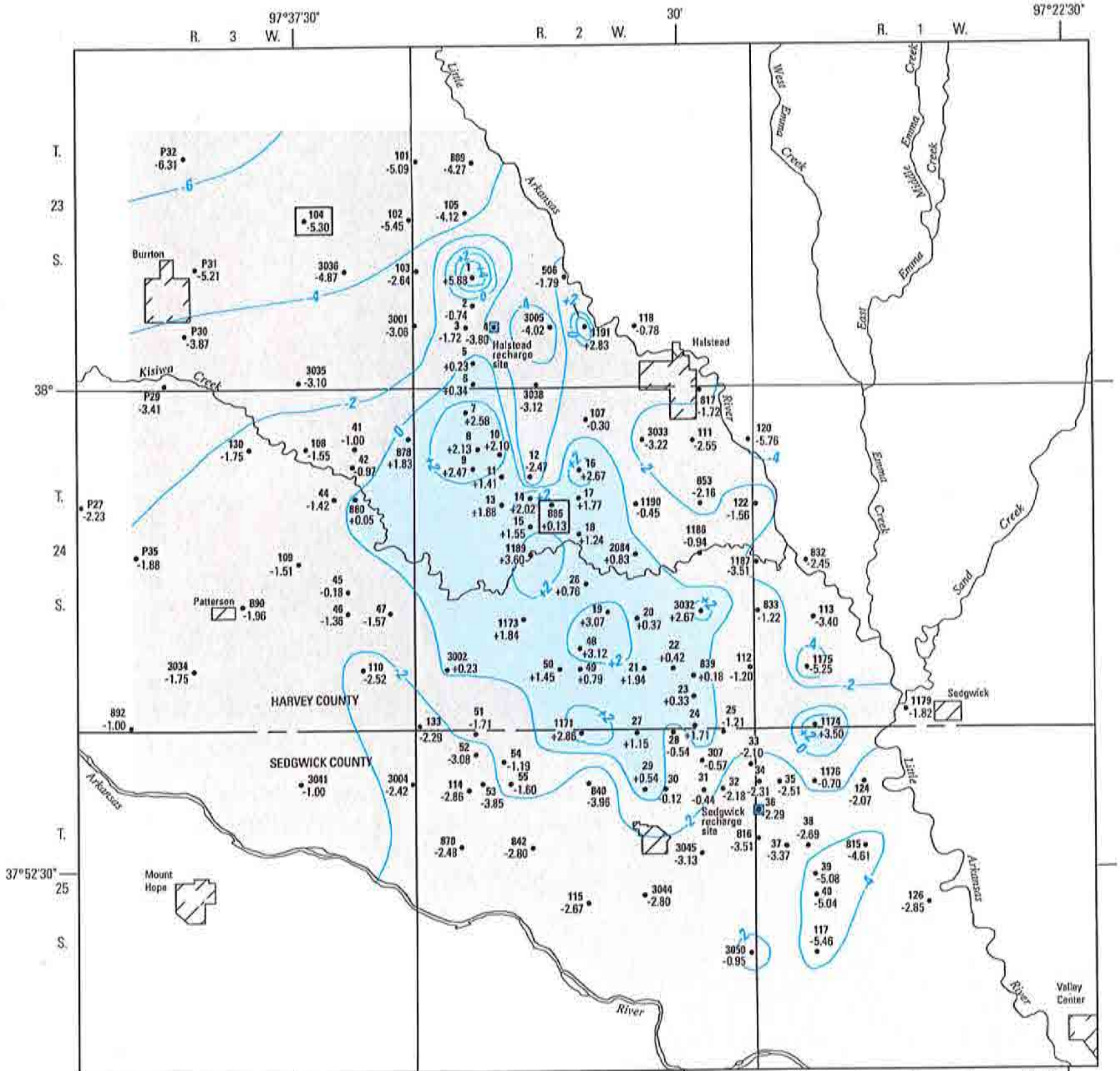
1 / -6.35 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was less than in April 2000.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between April 2000 and January 2003 was 58,000 acre-feet.

Figure 24. Water-level change in *Equus* Beds aquifer in study area, April 2000–January 2003.



Base from U.S. Geological Survey digital data, 1:100,000, 1983
Universal Transverse Mercator Projection, Zone 14

EXPLANATION

Study area

Area of water-level recovery, in feet

0 to +10

—2— **Line of equal change of water level**—Interval 2 feet. Dashed where approximately located

Artificial recharge demonstration site—Halstead site in operation May 1997–June 2002; Sedgwick site in operation April 1998–November 2000

886 +0.13 **Well hydrograph shown in figure 2**

1 +5.88 ***Equus* beds historic observation well**

Upper number is well identification number. Lower number is water-level change, in feet. Negative number indicates water level was less than in January 2000.

Water-level measurements were made by city of Wichita personnel.

Depletion of storage volume in study area between January 2000 and January 2003 was 33,000 acre-feet.

Figure 25. Water-level change in *Equus* Beds aquifer in study area, January 2000–January 2003.

Almost all wells in the study area had cumulative water-level rises for the period October 1992 to January 2003 (fig. 22). For this period, a single large area of cumulative water-level rises of 10 ft or more occurred throughout most of the central part of the study area, and two areas with rises of 20 ft or more occurred in the northern and central parts of the study area (fig. 22).

Water-level declines since August 1940 for the period January 2000 to January 2003 covered the smallest area in April 2000, as was noted previously in this report. This indicates that the general recovery of water levels from the low levels recorded in October 1992 to January 2000 discussed by Hansen and Aucott (2001) continued to April 2000. The cumulative water-level changes in the study area for the period October 1992 to April 2000 ranged from a rise of 0.33 ft in well 1174 in the eastern part of the study area to a rise of 30.81 ft in well 4 near the Halstead recharge site in the northern part of the study area (fig. 23). Water-level rises of more than 10 ft were common throughout the northern and central parts of the study area during this period with rises of more than 20 ft occurring in a large area in the northern and central parts of the study area (fig. 23). Comparison of the water-level-change map for the period October 1992 to January 2003 (fig. 22) with the map for the period October 1992 to April 2000 (fig. 23) shows three major differences as a result of water-level declines. These differences are the decrease in size from the October 1992 to April 2000 map to the October 1992 to January 2003 map of the area with water-level rises of 10 to 20 ft, especially in the northern part of the study area and towards the Little Arkansas River; the contraction and separation of a single area of water-level rises of 20 to 30 ft shown in the October 1992 to April 2000 map into two areas in the October 1992 to January 2003 map; and the disappearance in the October 1992 to January 2003 map of an area of water-level rises of 30 ft or more in the northern part of the study area that was shown in the October 1992 to April 2000 map (compare figs. 22 and 23). The water-level rises for the October 1992 to April 2000 period likely are due to the generally average to greater-than-average precipitation during this period (fig. 2*A*), the resulting decreased irrigation pumpage, and the decreased city pumpage because of increased city use of Cheney Reservoir as a source of water.

The map for the period April 2000 to January 2003 shows the cumulative water-level changes that occurred following the period of maximum recovery (fig. 24). The cumulative water-level changes for this period ranged from a decline of 10.55 ft in well 4 in the northern part of the study area to a rise of 3.30 ft in well 1174 just east of the central part of the study area (fig. 24). Cumulative water-level declines for the period April 2000 to January 2003 were common in the study area (fig. 24). Declines of 6 ft or more occurred in the northern, eastern, and southeastern parts of the study area and even included a small area of water-level declines of more than 10 ft around well 4 in the northern part of the study area (fig. 24). The water-level declines in the study area for the period April 2000 to January 2003 likely are due to increased irrigation and to precipitation that generally was less during 2000 through 2002 than during the preceding 5 years (figs. 2*A* and 2*B*). However, in the central part of the study area small water-level rises that generally were less than 2 ft were not uncommon (fig. 24), indicating that water levels inside this part of the study area continued to recover during the period April 2000 to January 2003 from the record low levels of October 1992. The continued recovery in the central part of the study area during the period April 2000 to January 2003 probably is due to decreased city pumpage that resulted from increased city use of Cheney Reservoir as a water source (fig. 2*B*).

For readers who prefer to deemphasize the effect of water-level rises and declines due to seasonal factors, the period January 2000 to January 2003 can be used instead of the period April 2000 to January 2003 to illustrate the cumulative water-level changes that occurred after the post-October 1992 to early 2000 recovery period. The water-level-change map for the period January 2000 to January 2003 (fig. 25) also shows the cumulative change that has occurred since the last report on water levels in the area (Hansen and Aucott, 2001). Maximum cumulative water-level changes from January 2000 to January 2003 ranged from a decline of 6.31 ft in well P32 in the northwestern part of the study area to a rise of 5.88 ft in well 1 in the northern part of the study area (fig. 25). The pattern of water-level changes for the period January 2000 to January 2003 (fig. 25) was similar to that for April 2000 to January 2003 (fig. 24). However, because the water-level-change map for January 2000 to January 2003 includes the recoveries that occurred between January and April 2000, the areas of water-level rises were larger, and the magnitude of the water-level declines generally were smaller than those seen in the April 2000 to January 2003 map (compare figs. 24 and 25). For example, water-level rises in the central part of the study area typically were less than 4 ft, and water-level declines of 6 ft or more were restricted to the extreme northwestern part of the study area (fig. 25). For the period January 2000 to January 2003, water-level rises in the central part of the study area probably were due to decreased city pumpage, and water-level declines in the rest of the study area probably were due to decreased precipitation and increased irrigation pumpage (figs. 2*A* and 2*B*).

Storage-Volume Changes

Changes in storage volume are defined for the purposes of this report as the change in saturated aquifer volume multiplied by the specific yield of the aquifer. A specific yield of 0.2 has been used to compute the changes in storage volume in the *Equus* Beds aquifer since Stramel (1956) first computed storage volume for the *Equus* Beds aquifer. The use of a specific yield of 0.2 was retained in this report because, as reported by Hansen and Aucott (2001), it is within the range of most estimates of specific yield and because there is no general agreement on an average value of specific yield for the *Equus* Beds aquifer in the study area.

Table 1. Storage-volume changes in *Equus* Beds aquifer near Wichita, south-central Kansas, August 1940–January 2003.

[Data on file with U.S. Geological Survey, Lawrence, Kansas]

Time period	Storage-volume change, in acre-feet		Proportion of change in study area that occurred in the central part of the study area (percent)
	Within study area	Within central part of study area	
August 1940–October 1992	[1]-283,000	[1]-159,000	56
August 1940–January 1993	[2]-255,000	[2]-154,000	60
August 1940–January 2000	[1]-126,000	[1]-70,600	56
August 1940–April 2000	-101,000	-74,500	74
August 1940–July 2000	-152,000	-76,700	50
August 1940–October 2000	-159,000	-87,000	55
August 1940–January 2001	-134,000	-78,900	59
August 1940–April 2001	-110,000	-72,500	66
August 1940–July 2001	-149,000	-74,900	50
August 1940–October 2001	-146,000	-76,700	52
August 1940–January 2002	-142,000	-77,100	54
August 1940–April 2002	-141,000	-74,900	53
August 1940–July 2002	-162,000	-78,600	49
August 1940–October 2002	-187,000	-90,100	48
August 1940–January 2003	-159,000	-83,400	52
October 1992–January 2000	+157,000	+88,400	56
October 1992–April 2000	+182,000	+84,500	46
October 1992–October 2002	+96,000	+68,000	72
October 1992–January 2003	+124,000	+75,600	61
January 2000–October 2002	-61,000	-19,500	32
January 2000–January 2003	-33,000	-12,800	39
April 2000–October 2002	-86,000	-15,600	18
April 2000–January 2003	-58,000	-8,900	15

[1]Storage-volume change previously reported by Hansen and Aucott (2001)

[2]Storage-volume change previously reported by Aucott and Myers (1998).

Changes in storage volume since August 1940 shown in table 1 were computed using the aquifer volume from areas inside lines of equal water-level change for the selected time periods. Changes in storage volume since times other than August 1940, as shown in table 1, were calculated as the difference between changes in storage volumes for August 1940 to the beginning of the selected time period and August 1940 to the end of the selected time period. For example, the change in storage volume for January 2000 to January 2003, as shown in table 1, was calculated as the difference between changes in storage volumes for August 1940 to January 2000 and August 1940 to January 2003.

The changes in storage volume since predevelopment (August 1940) and for selected periods are shown in table 1 for both the study area and the central part of the study area. The percentage of the study area's storage-volume change that occurred in the central part of the study area (table 1) was computed by dividing the storage-volume change in the central part of the study area by the storage-volume change in the whole study area and multiplying by 100.

Following the maximum loss of storage that occurred from August 1940 to October 1992 (Hansen and Aucott, 2001), storage volume recovered until April 2000 in the study area as a whole, but only until January 2000 in the central part of the study area (table 1). Storage-volume depletions from August 1940 to April 2000 were only about 101,000 acre-ft in the study area (table 1); in the central part of the study area, storage-volume depletion from August 1940 to January 2000 was about 70,600 acre-ft (table 1). These volumes represent recoveries of about 64 percent and about 56 percent of the August 1940 to

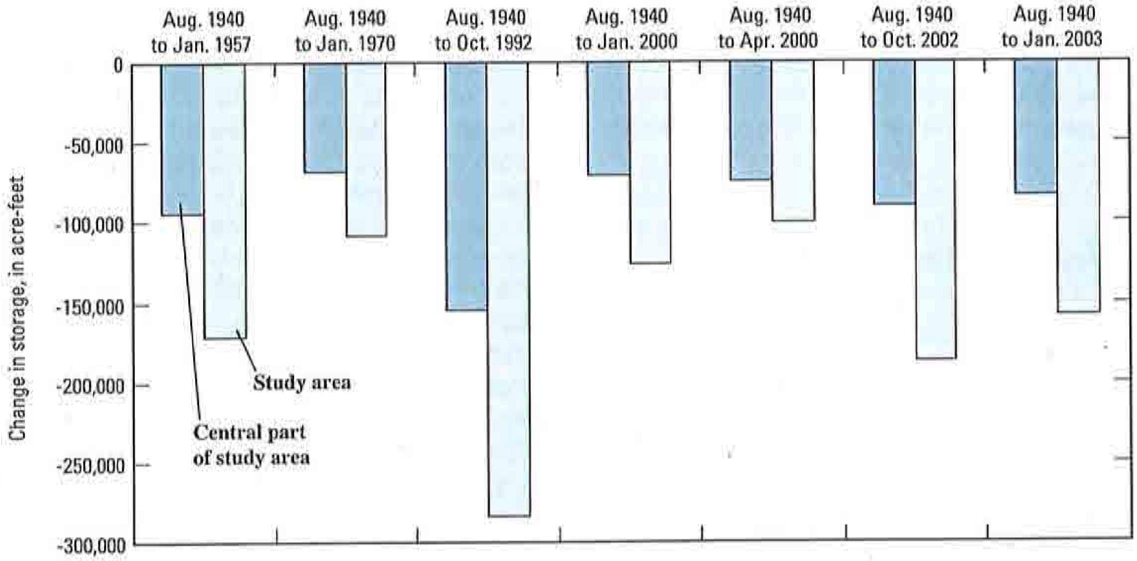


Figure 26. Storage-volume changes in *Equus* Beds aquifer in study area for significant periods during August 1940–January 2003 (source: data on file with the U.S. Geological Survey, Lawrence, Kansas).

October 1992 depletion that occurred, respectively, in the whole study area and in the central part of the study area. Although the central part of the study area makes up only 33 percent of the study area as a whole, it accounted for about 46 percent of the October 1992 to April 2000 storage-volume recovery in the study area (table 1). The storage-volume recoveries since October 1992 in both the study area and the central part of the study area were due to the general rise in ground-water levels associated with average to greater-than-average precipitation and to decreases in city and irrigation pumpage (figs. 2*A* and 2*B*). Water levels and storage-volume changes from August 1940 to April 2000 for the study area and from August 1940 to January 2000 for the central part of the study area were similar to changes observed from August 1940 to January 1970 (fig. 26), which occurred during the 1957–77 period of relatively stable water levels (figs. 2*C*) identified by Aucott and Myers (1998).

Since April 2000 in the study area and since January 2000 in the central part of the study area, aquifer storage volume has tended to decrease (table 1 and fig. 2*C*). Whether this indicates the end of the period of general water-level and storage-volume recovery that began in January 1993 (Aucott and Myers, 1998) is not clear at this time (2003). Seasonal water-level declines and storage-volume depletions during 2000–02 (represented by storage-volume changes in July and October) were not completely recovered during the months that followed (represented by storage-volume changes in January and April) (table 1 and fig. 2*C*).

The largest storage-volume depletion since August 1940 for the period January 2000 to January 2003 occurred during October 2002 both in the study area and in the central part of the study area (table 1). The storage-volume changes for the period August 1940 to October 2002 (table 1) represent about 86,000 acre-ft or a 47-percent loss of the storage recovered between October 1992 and April 2000 in the study area and about 19,500 acre-ft or a 22-percent loss of the storage recovered between October 1992 and January 2000 in the central part of the study area. Major factors in the decreases in water levels and storage volumes during the January 2000 to January 2003 period were reduced recharge associated with precipitation that generally was less than in the preceding 5 years and increased irrigation pumpage (figs. 2*A*, 2*B*, and 2*C*).

Table 1 shows the central part of the study area accounted for only 8,900 acre-ft or about 15 percent of the storage-volume depletion in the whole study area for the period April 2000 to January 2003. This disproportionately small storage-volume depletion that occurred in the central part of the study area probably was the result of decreased city pumpage. This indicates that the loss of storage in the study area probably would have been larger if the continued decrease in city pumpage, which is closely associated with water-level rises in the central part of the study area and increased city use of Cheney Reservoir as a source of water, had not occurred. Storage volume in January 2003 in the study area was similar to volumes seen in the late 1970s and mid-1990s (fig. 2*C*).

Compared to the storage changes in the entire study area, the storage changes in the central part of the study area have remained relatively constant for the period January 2000 to January 2003 (table 1). This may be due in part to the small water-level rises in the central part of the study area [which likely were due to decreased city pumpage as a result of increased city use of Cheney Reservoir as a water source (fig. 2B)] that mostly offset the larger water-level declines along the edges of the central part of the study area (fig. 25). Thus, the loss of storage for this period probably would have been larger if the continued decrease in city pumpage had not occurred. It is interesting to note that the percentage of the study area depletion that occurred in the central part of the study area was less during the "fall" season (represented by the months of July and October) than during the "spring" season (represented by the months of January and April). This likely is due to seasonal effects of increased irrigation pumpage during the growing season and to decreased city pumpage during the period January 2000 to January 2003.

Effects of Artificial Recharge

In 1995, the city of Wichita began investigating the potential for artificial recharge to meet future water-supply needs and to protect the *Equus* Beds aquifer from the intrusion of saltwater from natural and human-related sources to the west (Ziegler and others, 1999). Two artificial recharge demonstration sites are located in the central part of the study area near Halstead and Sedgwick (fig. 1). Artificial recharge operations occurred during May 1997–June 2002 at the Halstead site and during April 1998–November 2000 at the Sedgwick site.

Although artificial recharge has totaled more than 3,324 acre-ft during 1997 through May 2002 (U.S. Geological Survey, 2003b), this is equivalent to less than 3 percent of the approximately 139,000 acre-ft of water pumped by the city from the study area during 1997–2002 (Joan Kenny, U.S. Geological Survey, written commun., 2000 and 2003). The effect of the artificial recharge on water-level changes and storage volume is evident in wells near the recharge sites (Hansen and Aucott, 2001) but is not evident in study-area maps of water-level changes. For example, recharge occurred at the Halstead site throughout much of March and April 2001 (Heather Ross, U.S. Geological Survey, oral commun., September 2003), but no "mounding" of water from this recharge (indicated by contours bending around or enclosing an area of smaller declines) occurs on the August 1940–April 2001 water-level-change map near the Halstead site (fig. 13). The effects of artificial recharge probably were masked by the generally larger decreases in city pumpage in the central part of the study area, which went from about 24,600 acre-ft in 1997 to about 18,600 acre-ft in 2002 (fig. 2*B*). As artificial recharge moves from the demonstration stage to the production stage, its effect on water levels and storage volume in the study area may become more obvious.

Summary

The *Equus* Beds aquifer in southwestern Harvey County and northwestern Sedgwick County was developed to supply water to Wichita residents and for irrigation in south-central Kansas beginning on September 1, 1940. Ground-water pumpage for city and agricultural use from the aquifer caused water levels to decline in a large part of the area. Irrigation pumpage in the area increased substantially during the 1970s and 1980s and accelerated water-level declines. Most of the water-level declines can be attributed to ground-water pumping; however, climatic conditions (and thus recharge to the *Equus* Beds aquifer) also have affected ground-water levels. In 1965, the city of Wichita began using water from Cheney Reservoir in addition to water from the *Equus* Beds aquifer. Since 1995, the city has been investigating the use of artificial recharge in the study area to meet future water-supply needs and to protect the aquifer from the intrusion of saltwater from natural and human-related sources to the west.

A period of water-level rises associated with generally greater-than-average precipitation and decreased city pumpage from the study area began in 1993. An important factor in the decreased city pumpage was increased use of Cheney Reservoir by the city of Wichita as a water-supply source; as a result, city pumpage from the Equus Beds aquifer during 1993–2002 went from being greater than one-half to slightly less than one-third of Wichita's water usage.

During January 2003, the direction of ground-water flow in the *Equus* Beds aquifer in the study area generally was from west to east, similar to predevelopment of the aquifer. The maximum water-level decline since 1940 for the period January 2000 to January 2003 was 29.54 ft in July 2002 in well 3 in the northern part of the study area. The period with the smallest area of water-level declines since August 1940 for the period January 2000 to January 2003 was April 2000. The seasonal water-level changes in wells in most of the study area during the period January 2000 to January 2003 typically were less than 5 ft. However, during 2000 and 2001, seasonal water-level changes of 20 to 25 ft occurred in the northern part of the study area, probably due to seasonal agricultural irrigation pumpage and semiconfined aquifer conditions.

Almost all wells in the study area had cumulative water-level rises from the record low levels in October 1992 to January 2003. Water-level rises following October 1992 continued until April 2000 when cumulative water-level rises of 10 ft or more were common in the study area, and rises of more than 20 ft occurred in a large area in the northern and central parts of the study area. These water-level rises likely were mostly due to generally average to greater-than-average precipitation and decreased city pumpage. Cumulative water-level declines, which likely were mostly due to increased irrigation pumpage and decreased precipitation, were common in the study area from April 2000 to January 2003, and declines of 6 or more ft occurred in the northern, eastern, and southeastern parts of the study area. However, small water-level rises that generally were less than 2 ft were not uncommon in the central part of the study area and probably were the result of the continued decrease in city pumpage.

Following the maximum loss of storage that occurred from August 1940 to October 1992, storage volume recovered until April 2000 in the study area. The storage-volume depletion of 101,000 acre-ft from August 1940 to April 2000 represents a recovery of about 64 percent of the January 1940 to October 1992 depletion in the study area; about 46 percent of this recovery occurred in the central part of the study area. The recovery was due to the general rises in ground-water levels associated with average to greater-than-average precipitation and to decreases in city and irrigation pumpage. Storage-volume changes from August 1940 to January 2000 in the study area were similar to changes observed from August 1940 to January 1970, which occurred during the 1957–77 period of relatively stable water levels.

Since April 2000 aquifer storage volume in the study area has tended to decrease. The largest storage-volume depletion since August 1940 for the period January 2000 to January 2003 occurred during October 2002 when the storage volume in the study area represented about a 47-percent loss of the storage previously recovered between October 1992 and April 2000. However, the central part of the study area accounted for only about 15 percent of the loss of storage in the study area for the period April 2000 to January 2003. Thus, the loss of storage probably would have been larger if the continued decrease in city pumpage, which is closely associated with the water-level rises in the central part of the study area and increased city use of Cheney Reservoir as a source of water, had not occurred. Storage volume in January 2003 was similar to volumes during the late 1970s and mid-1990s.

Operation of the Halstead and Sedgwick artificial recharge demonstration sites began in May 1997. Artificial recharge from 1997 through May 2002 was equivalent to less than 3 percent of city pumpage from the *Equus* Beds aquifer in the study area during 1997–2002. The effects of artificial recharge on water-level changes and storage volume in the study area probably were masked by the generally larger decreases in city pumpage from the *Equus* Beds aquifer.

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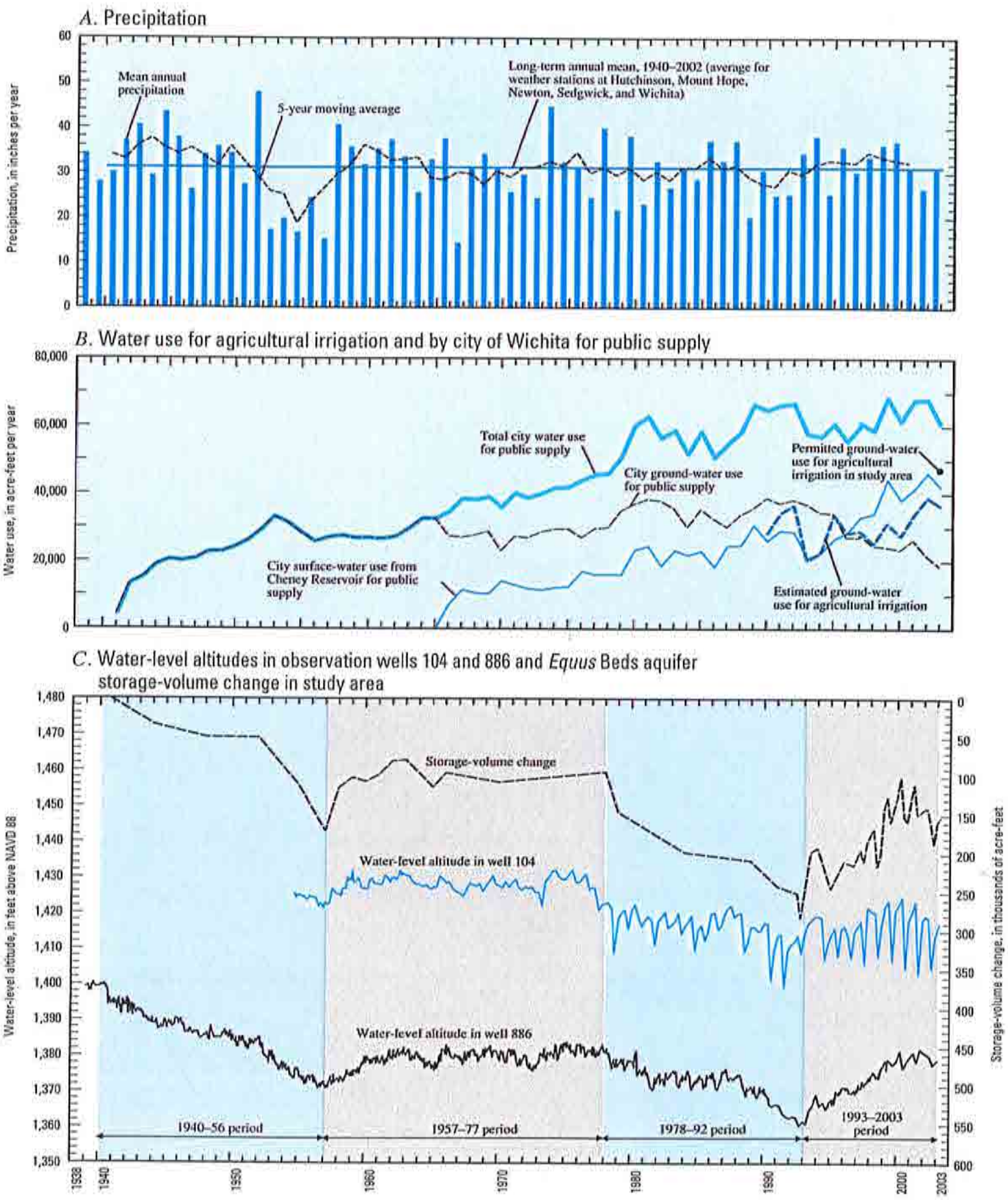

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EXHIBIT F

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Baseline Water Quality and Preliminary Effects of Artificial Recharge on Ground Water, South-Central Kansas, 1995–98

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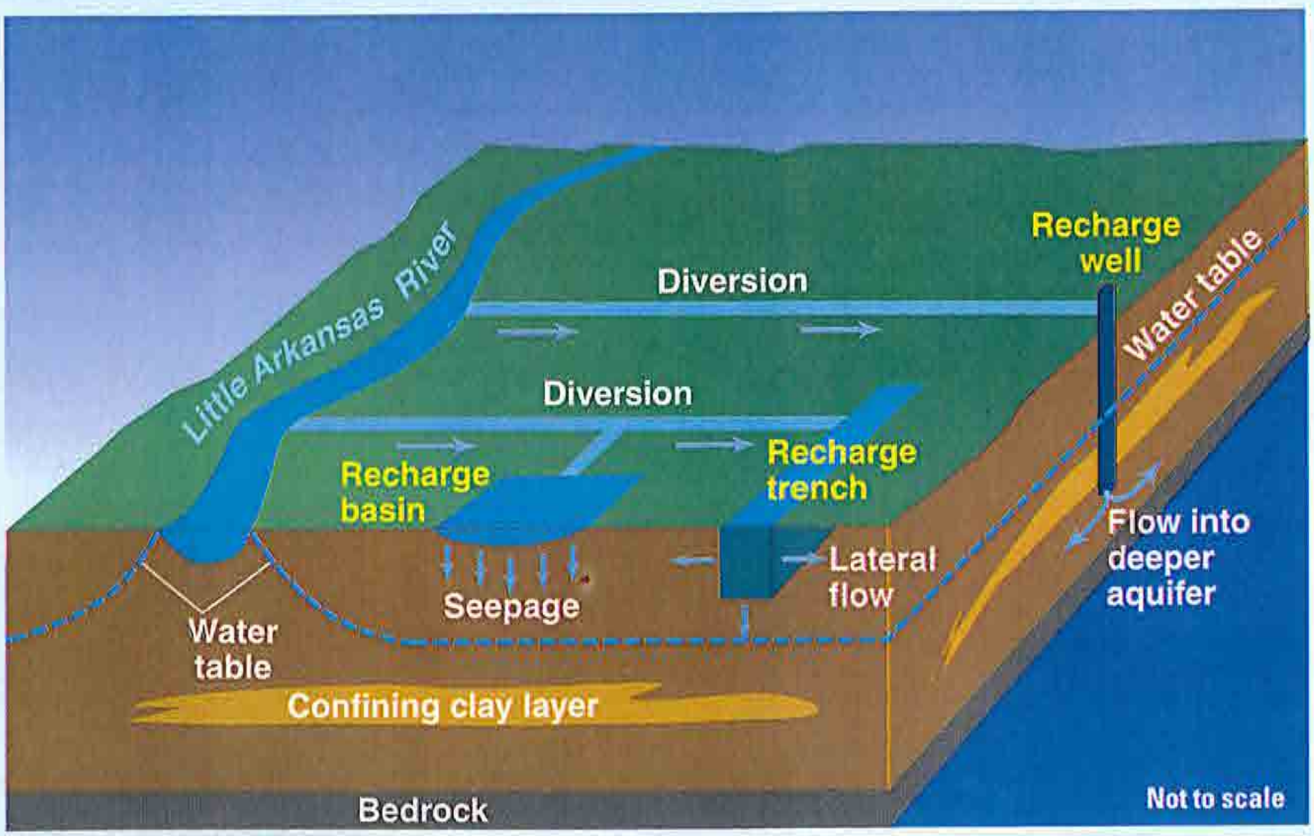

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Baseline Water Quality and Preliminary Effects of Artificial Recharge on Ground Water, South-Central Kansas, 1995–98

By ANDREW C. ZIEGLER, VICTORIA G. CHRISTENSEN, and HEATHER C. ROSS

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CONTENTS

Abstract 1
Introduction 2
  Background 2
  Purpose and Scope 2
Description of Study Area 3
  Equus Beds Ground-Water Recharge Demonstration Project 3
  Halstead Recharge System 6
  Sedgwick Recharge System 6
Methods 8
Baseline Water Quality, 1995–98 13
  Physical Properties and Selected Constituents 18
  Major Ions 18
  Nutrients 26
  Trace Elements 26
  Total Organic Carbon 28
  Bacteria 28
  Organic Compounds 33
Preliminary Effects of Artificial Recharge, 1996–98 49
  Halstead Recharge System 49
    Effects on Ground-Water Levels 49
    Effects on Ground-Water Quality 49
  Sedgwick Recharge System 56
    Effects on Ground-Water Levels 56
    Effects on Ground-Water Quality 57
Compatibility of Source Water for Artificial Recharge 61
Constituents of Concern for Future Monitoring 67
Summary and Conclusions 72
References 73

FIGURES

1–3. Maps showing:

1. Location of study area near Wichita, south-central Kansas 4
2. Location of surface-water monitoring sites and ground-water wells within the *Equus* Beds Ground-Water Recharge Demonstration Project area 5
3. Location of data-collection sites at the Halstead diversion well and recharge sites 7

4. Hydrogeologic section between deep monitoring wells DMW–H1 and DMW–H13 at Halstead recharge site.... 8
5. Map showing location of data-collection sites at Sedgwick recharge site 9
6. Hydrogeologic section between deep monitoring wells DMW–S10 and DMW–S14 at Sedgwick recharge site . 10

7–28. Graphs showing:

7. Ranges in concentrations of sodium and chloride in surface- and ground-water samples during baseline water-quality monitoring 25
8. Ranges in concentrations of nitrite plus nitrate in surface- and ground-water samples during baseline water-quality monitoring 29
9. Ranges in iron and manganese concentrations in surface- and ground-water samples during baseline water-quality monitoring 33
10. Ranges in atrazine concentrations in surface- and ground-water samples during baseline water-quality monitoring 48
11. Discharge of Little Arkansas River at Highway 50 near Halstead and water levels in the *Equus* Beds aquifer at Halstead diversion site, March 1995–July 1998 50
12. Cumulative volume of water recharged at Halstead recharge site, May 1997–July 1998 51
13. Water-level altitudes in monitoring wells completed in the *Equus* Beds aquifer at Halstead recharge site, May 1997–July 1998 51
14. Ranges in sodium concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 52
15. Ranges in chloride concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 53
16. Ranges in nitrite plus nitrate concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 54
17. Ranges in iron and manganese concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 55
18. Ranges in atrazine concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 56
19. Comparison of chloride concentrations in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and in diversion well water and ground water from Halstead recharge site, May 1997–July 1998 58
20. Comparison of atrazine concentrations in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and in diversion well water and ground water from Halstead recharge site, May 1997–July 1998 60
21. Water levels in the *Equus* Beds aquifer at Sedgwick recharge site, June 1997–July 1998 62
22. Ranges in sodium concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 63
23. Ranges in chloride concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 63
24. Comparison of chloride concentrations in surface water and treated source water from Little Arkansas River and in water from monitoring wells at Sedgwick recharge site, June 1997–July 1998 64
25. Ranges in nitrite plus nitrate concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 65
26. Ranges in iron and manganese concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 66
27. Ranges in atrazine concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions 67

FIGURES—Continued

7–28. Graphs showing:
  28. Comparison of atrazine concentrations in surface water and treated source water from Little Arkansas River, March 1995–July 1998, and in water from monitoring wells at Sedgwick recharge site, June 1997–July 1998 68

29. Stiff diagrams showing mean concentrations of major ions in surface and ground water during baseline and recharge conditions at Halstead diversion well site, Halstead recharge site, and Sedgwick recharge site 69

30. Graph showing relations between specific conductance and chloride concentrations in surface water from Little Arkansas River at Highway 50 near Halstead, ground water from Halstead diversion well, and surface water and treated source water from Little Arkansas River at Sedgwick, March 1995–July 1998 71

31. Graphs showing relation between triazine herbicide concentrations determined by enzyme-linked immunosorbent assay and atrazine concentrations determined by gas chromatography/mass spectrometry for samples collected from Halstead and Sedgwick surface-water monitoring sites, March 1995–July 1998 73

TABLES

1. Data-collection sites for *Equus* Beds Ground-Water Recharge Demonstration Project, south-central Kansas, 1995–98 11
2. Time periods defining baseline and artificial recharge conditions for data-collection-site groupings 13
3. Constituents analyzed in water samples, 1995–98 14
4. Summary of determinations of physical properties and analyses of selected constituents in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 19
5. Summary of major-ion concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 22
6. Summary of nutrient concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 27
7. Summary of trace-element concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 30
8. Summary of total organic carbon concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 34
9. Summary of total coliform bacteria densities in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 35
10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions 36

CONVERSION FACTORS, ABBREVIATIONS, AND VERTICAL DATUM

Multiply	By	To obtain
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
foot (ft)	0.3048	meter
gallon per minute (gal/min)	0.06309	liter per second
inch (in.)	2.54	centimeter
mile (mi)	1.609	kilometer
million gallons (Mgal)	3,785	cubic meter
square mile (mi^2)	2.590	square kilometer

Temperature can be converted to degrees Celsius (°C) or degrees Fahrenheit (°F) by the equations:

$$°C=5/9\ (°F - 32)$$
$$°F=9/5\ (°C) + 32.$$

Milliequivalents per liter (meq/L) can be calculated with the following equation:

meq/L=(concentration in milligrams per liter)(1/molecular weight in grams)(valence).

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

Baseline Water Quality and Preliminary Effects of Artificial Recharge on Ground Water, South-Central Kansas, 1995–98

By Andrew C. Ziegler, Victoria G. Christensen, *and* Heather C. Ross

Abstract

To investigate the feasbility of artificial recharge as a method of meeting future water-supply needs and to protect the *Equus* Beds aquifer from saltwater intrusion from natural and anthropogenic sources to the west, the *Equus* Beds Ground-Water Recharge Demonstration Project was begun in 1995. The project is a cooperative effort between the city of Wichita and the Bureau of Reclamation, U.S. Department of the Interior. During the project, high flows from the .e Arkansas River are captured and recharged into the *Equus* Beds aquifer through recharge basins, a trench, or a recharge well, located at two recharge sites near Halstead and Sedgwick, Kansas. To document baseline concentrations and compatibility of stream (recharge) and aquifer water, the U.S. Geological Survey collected water samples from February 1995 through August 1998. These samples were analyzed for dissolved solids, total and dissolved inorganic constituents, nutrients, organic and volatile organic compounds, radionuclides, and bacteria.

Results of baseline sampling indicated that the primary constituents of concern for recharge were sodium, chloride, nitrite plus nitrate, iron and manganese, total coliform bacteria, and atrazine. Chloride and atrazine were of particular concern because concentrations of these constituents in water from the Little Arkansas River frequently exceeded regulatory criteria. The Little Arkansas River is used as the source water for recharge. The ' '. Environmental Protection Agency Secondary Maximum Contaminant Level for chloride is 250 mg/L (milligrams per liter), and the Maximum Contaminant Level for atrazine is 3.0 µg/L (micrograms per liter) as an annual mean. Baseline concentrations of chloride in surface water ranged from 8.0 to 400 mg/L. Baseline concentrations of atrazine in surface water ranged from less than 0.10 to 46 µg/L.

Concentrations of chloride and atrazine have increased in water from some of the wells at both the Halstead and Sedgwick recharge sites after recharge began, although concentrations remained within the range of baseline values in the *Equus* Beds aquifer and are considerably less than U.S. Environmental Protection Agency drinking-water criteria. However, a substantial quantity of water has not been recharged at the Sedgwick site to determine the overall effects of artificial recharge on aquifer quality. Continued monitoring is necessary to determine long-term effects at both sites.

Major ion and trace element concentrations in source water and receiving water were analyzed to determine the compatibility of recharge and receiving ground water for artificial recharge. Stiff diagrams of major ions were used to show the similarity or differences between source surface water and receiving ground water. The water from both sources, for the most part, was chemically compatible to the receiving aquifer water at both recharge sites.

It may be possible to decrease the monitoring frequency at the Halstead recharge site because water-quality changes in receiving water at this

site are very gradual. However, real-time water-quality monitoring of surrogates needs to be site specific for the determination of chloride and atrazine. Real-time water-quality monitoring potentially can be used to more effectively manage the artificial recharge process, enabling project officials to respond more rapidly to changes in water quality.

INTRODUCTION

Background

The Wichita well field, initiated in the 1940's and completed in the 1950's in the *Equus* Beds aquifer, is one of the primary sources of water for the city of Wichita and the surrounding area in south-central Kansas. Historical water use for municipal supply and irrigation caused water levels in the *Equus* Beds aquifer to decline as much as 30 ft by 1993 (Aucott and others, 1998). Lower water levels not only represent a diminished water supply but also encourage saltwater intrusion from the Burrton oil field to the northwest and from the Arkansas River to the southwest into the freshwater of the *Equus* beds (Myers and others, 1996).

Cheney Reservoir was first used in 1965 to supplement Wichita's water supply. In 1994, city officials changed water-policy practices and began to use the reservoir for a larger percentage of water supply for the area. Since 1993, ground-water levels have risen by more than 10 ft in some areas of the Wichita well field, primarily because of increased use of water from Cheney Reservoir and decreased pumping in the well field area (Aucott and others, 1998). However, an expected increase in demand from both water sources could cause supply shortages in the near future (2010) (Warren and others, 1995).

The *Equus* Beds Ground-Water Recharge Demonstration Project was begun in 1995 to investigate the feasibility of artificially recharging the *Equus* Beds aquifer as one alternative to meet future water-supply needs and to protect the aquifer from saltwater intrusion from natural and anthropogenic sources. Throughout the project, high flows from the Little Arkansas River are captured and recharged into the aquifer through various techniques, including recharge basins, a trench, and a recharge well. Before artificial recharge can be determined to be a viable alternative, the water-quality effect of artificially recharging the *Equus* Beds aquifer needs to be assessed.

The *Equus* Beds Ground-Water Recharge Demonstration Project is a cooperative effort between the city of Wichita and the Bureau of Reclamation, U.S. Department of the Interior. Additional participants in the project are the U.S. Geological Survey (USGS), *Equus* Beds Groundwater Management District No. 2 (Halstead, Kansas), and the U.S. Environmental Agency (USEPA). Project work is coordinated with the Kansas Department of Health and Environment (KDHE), the Kansas Water Office, and the Kansas Department of Agriculture, Division of Water Resources. Burns and McDonnell Engineering Consultants (Kansas City, Missouri) and Mid-Kansas Engineering Consultants (Wichita, Kansas) provide engineering expertise and project management. The maintenance and operation of the recharge facilities are performed by the city of Wichita.

The *Equus* Beds Ground-Water Recharge Demonstration Project is a part of the High Plains States Ground-Water Recharge Demonstration Program, which is a cooperative effort among the Bureau of Reclamation, USGS, and USEPA to study the potential for artificial recharge and its effects in 17 Western States. The USGS also has worked cooperatively with the city of Wichita for many years in evaluating the ground-water system and interaction with streams in the area to further the understanding of the entire hydrologic system and to provide information to improve local decisionmaking.

Purpose and Scope

The purposes of this report are: (1) to describe baseline water quality of the Little Arkansas River and the *Equus* Beds aquifer for the *Equus* Beds Ground-Water Recharge Demonstration Project and (2) to describe preliminary effects of artificial recharge from April 1996 through August 1998 on ground-water levels and water quality of the aquifer at two locations—the Halstead recharge site and the Sedgwick recharge site. The compatibility of recharge source water with the receiving ground water and constituents of concern for artificial recharge as related to monitoring frequency and future recharge operations are also discussed.

Preliminary effects of artificial recharge on water levels in the *Equus* Beds aquifer were determined by comparing baseline water levels (measurements made prior to any recharge activities) to water-level measurements made after artificial recharge began. Preliminary effects of artificial recharge on water quality of the *Equus* Beds aquifer were determined by comparing baseline concentrations and artificial recharge concentrations of selected constituents in water collected from ground-water monitoring wells.

Compatibility of recharge source water with receiving ground water was determined by comparing major-ion chemistry for water from various data-collection sites during baseline and artificial recharge conditions. Also, an examination of water temperatures, turbidity, dissolved oxygen, iron, and manganese were used as measures of whether source water, when combined with receiving ground water, could cause plugging of aquifer material and thus inhibit artificial recharge activities.

Constituents of concern were identified as those water-quality constituents that frequently exceeded USEPA water-quality criteria and had the potential to affect artificial recharge operations. The benefits of continued monitoring of these constituents during future recharge operations are also outlined.

Information in this report may be used to evaluate the effects of artificial recharge to date (1999) and to adjust future monitoring frequency and (or) scope. The methodology described in this report can be applied to similar recharge studies in other parts of the United States and foreign lands with similar hydrologic conditions.

DESCRIPTION OF STUDY AREA

Equus Beds Ground-Water Recharge Demonstration Project

The study area for the *Equus* Beds Ground-Water Recharge Demonstration Project encompasses approximately 165 mi^2 and extends northwest of Wichita across parts of Harvey and Sedgwick Counties in south-central Kansas (fig. 1). The study area is bounded by the Arkansas River on the southwest and includes the Little Arkansas River on the northeast. The Wichita well field encompasses 55 mi^2 and is located within the study area. The drainage area for the Little Arkansas River Basin is about 1,200 mi^2. Land use in the basin is primarily agricultural and includes the production of livestock (pasture and rangeland) and field crops. Field crops produced include corn, sorghum, soybeans, and wheat (Kansas Department of Agriculture and U.S. Department of Agriculture, 1997). Agricultural chemicals applied to enhance crop production in the area include fertilizers (such as nitrate, ammonia, and phosphorus) and pesticides (primarily alachlor and atrazine).

The *Equus* Beds aquifer, a part of the larger High Plains aquifer, consists of alluvial deposits of sand and gravel interbedded with clay or silt. In the study area, the general direction of ground-water movement in the *Equus* Beds aquifer is to the east (Aucott and others, 1998). However, in the vicinity of the well field and the Little Arkansas River, ground-water movement has been altered by pumping wells and a low-head dam on the river (fig. 2). The Little Arkansas River is primarily a gaining stream within the study area as indicated by higher water levels in wells adjacent to the stream (Myers and others, 1996; Aucott and others, 1998). This is not the case, however, near the Halstead monitoring site (07143680, fig. 1) where a low-head dam about 1 mi downstream causes higher water levels in the stream than in the adjacent aquifer, resulting in stream-water recharge of the aquifer in this vicinity (fig. 2).

The McPherson channel is a trough of unconsolidated deposits about 200 ft thick within the *Equus* Beds aquifer that extends from Lindsborg (about 30 mi north of the study area) to Halstead (Spinazola and others, 1985). This buried alluvial valley is a major flow path for ground-water movement within the *Equus* Beds aquifer (Leonard and Kleinschmidt, 1976) and is important as it relates to the movement of chemical constituents. Flow of ground water in the vicinity of the McPherson channel is towards the center of the channel and southward. The towns of Lindsborg and McPherson are upgradient from the study area, and wastewater discharge from these towns may be sources of chemical constituents, such as chloride, in the aquifer water.

The encroachment of saltwater into the *Equus* Beds aquifer has been a concern in the area for many years. The sources of this saltwater include mineralized water from the Arkansas River (Spinazola and others, 1985; Myers and others, 1996), oil-field brines from the Burrton area west of the study area and northwest of the Wichita well field, and mineralized water in the underlying Wellington aquifer (Leonard and Kleinschmidt, 1976; Spinazola and others, 1985). Other possible sources of saltwater are municipal

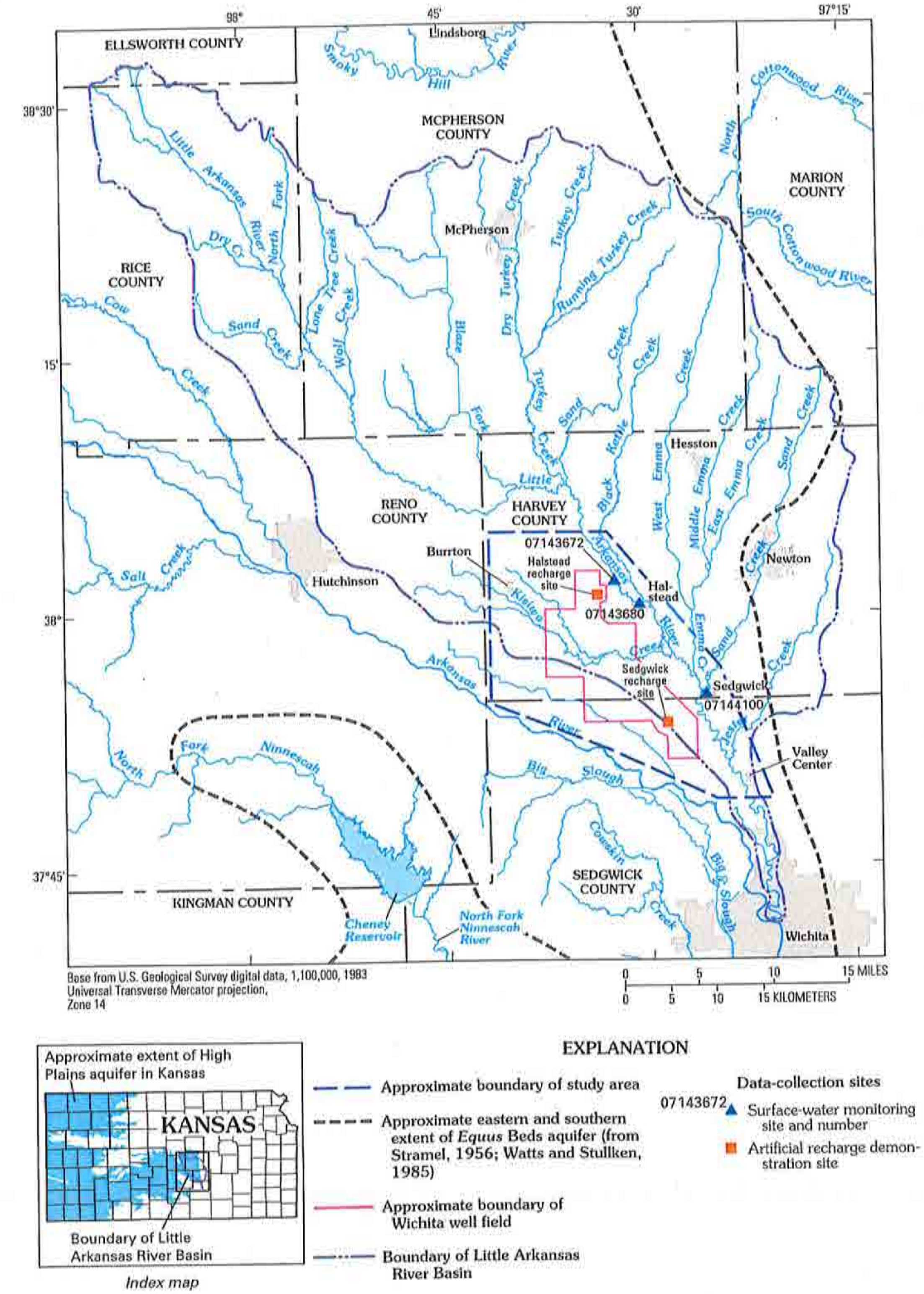


Figure 1. Location of study area near Wichita, south-central Kansas.

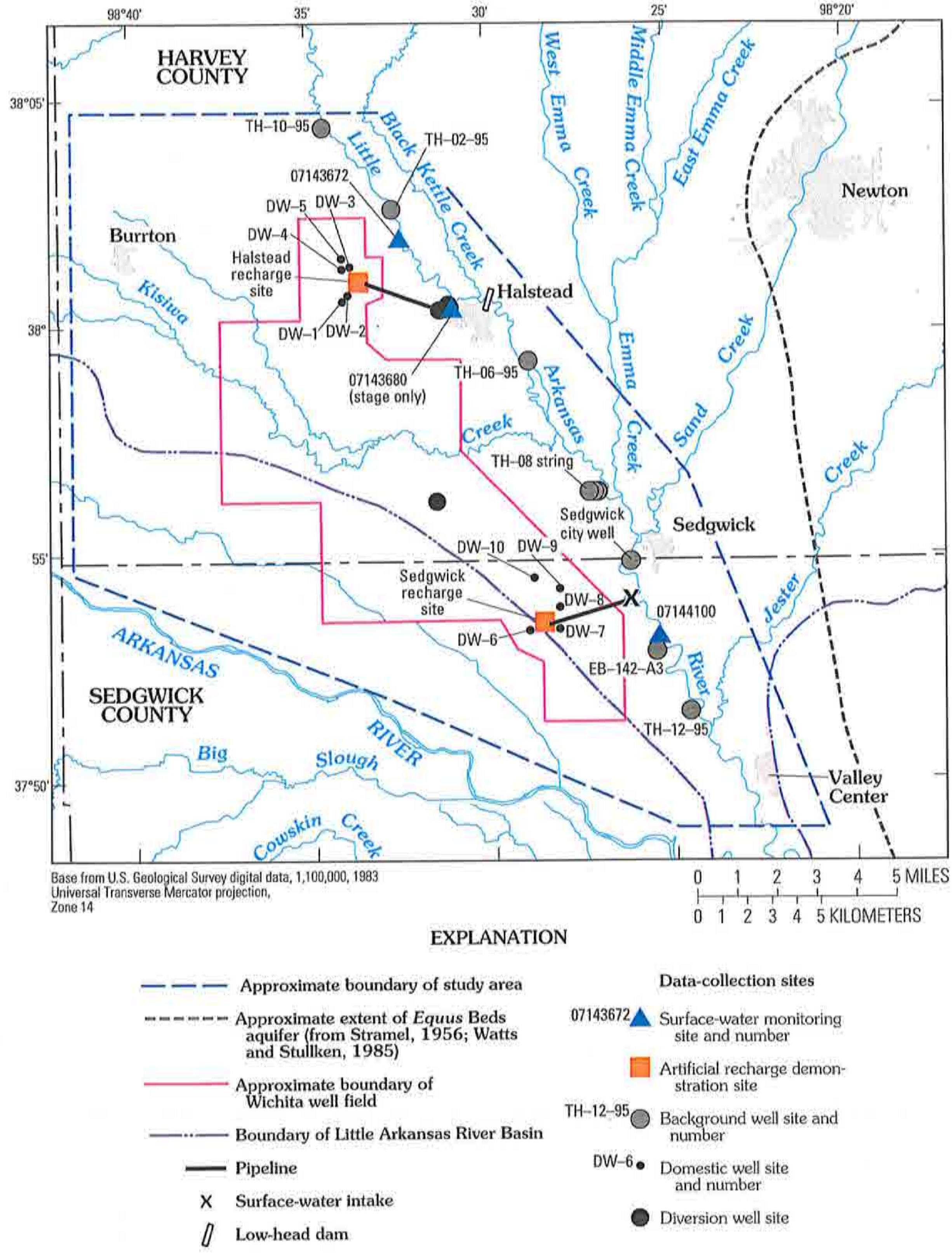


Figure 2. Location of surface-water monitoring sites and ground-water wells within the *Equus* Beds Ground-Water Recharge Demonstration Project area.

waste and industrial discharges from upgradient urban areas in McPherson and Newton (Donald Whittemore, Kansas Geological Survey, oral commun., January 1999).

Halstead Recharge System

Artificial recharge began at the Halstead recharge site on May 29, 1997. The Halstead recharge system consists of the USGS streamflow-gaging station on the Little Arkansas River at Highway 50 near Halstead (station 07143672, fig. 2), the Halstead diversion well site (fig. 2), and the Halstead recharge site (fig. 2). Water for the demonstration project may be diverted from the well completed in the alluvium adjacent to the Little Arkansas River only when flow in the river exceeds 42 ft^3/s at the gaging station from April 1 through September 30 and 20 ft^3/s from October 1 through March 31 in accordance with the Kansas Department of Agriculture, Division of Water Resources, permit conditions (Burns and McDonnell, 1998). The flow requirements at this site did not apply to aquifer tests conducted from April through July 1996. By pumping the diversion well, the ground water stored in the bank deposits of the Little Arkansas River is withdrawn, thereby decreasing the water levels surrounding the diversion well and causing surface water from the Little Arkansas River to be induced into the alluvium. The water quality and quantity at the diversion well site are monitored through samples from five shallow monitoring wells, a deep monitoring well, and the diversion well, which has a pumping capacity of about 1,000 gal/min (fig. 3).

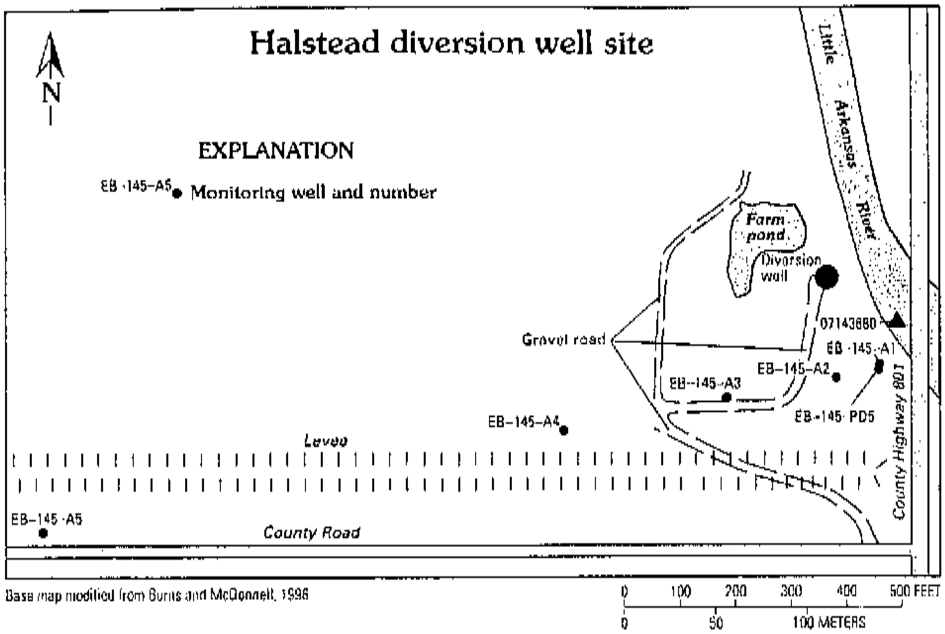
Discharge from the diversion well then is pumped about 2 mi through an underground pipeline to the Halstead recharge site (fig. 3) where it is recharged into the aquifer using one of three methods—recharge basins, a recharge trench, or a recharge well. There are two recharge basins at the Halstead site that are each capable of recharging 50 to 120 gal/min to the aquifer.

At the Halstead recharge site, a clay layer occurs approximately 30 ft below land surface (fig. 4) and impedes the vertical flow of recharge water into the *Equus* Beds aquifer. This impediment creates a "mounding" of water that rises to the level of the basin bottom and results in slowed percolation. A recharge trench was installed by the city of Wichita to promote vertical movement of recharge water into the aquifer (fig. 3). The recharge trench is 100 ft long, 3 ft wide, and approximately 15 ft deep and has been tested at recharge rates of 100 to 120 gal/min (Burns and McDonnell, written commun., 1998). In addition, a recharge well is used to inject water into the lower parts of the *Equus* Beds aquifer. The recharge well is deep (225 ft) and is capable of recharging about 900 gal/min to the aquifer. The vertical-flow problems associated with recharge water at this site do not affect the recharge well because the recharge water is injected beneath the clay layer (fig. 4).

Sedgwick Recharge System

Artificial recharge at the Sedgwick recharge site began in April 1998. Unlike the Halstead recharge system, where water is withdrawn from the alluvium, the water in the Sedgwick recharge system is diverted directly from the Little Arkansas River for recharge. In the Sedgwick recharge system, water may be withdrawn from the river near USGS streamflow-gaging station 07144100 (fig. 1) at all times when streamflow exceeds 40 ft^3/s (Burns and McDonnell, 1998). At the intake site, a polymer is added as a coagulant aid to reduce turbidity as water passes through a parallel plate separator (Burns and McDonnell, 1998). Next, powdered activated carbon (PAC) is added to remove atrazine and other organic compounds from the water. The treated source water then is pumped about 2 mi by underground pipeline to the Sedgwick recharge site.

Once the treated source water reaches the Sedgwick recharge site (fig. 5), it is pumped to a settling basin to allow the remaining suspended sediment and PAC to settle out of the water. From the settling basin, treated source water is pumped to one of three recharge basins and allowed to infiltrate into the aquifer. A hydrogeologic section across the Sedgwick recharge site between deep monitoring wells DMW–S10 and DMW–S14 is shown in figure 6. Clay layers could impede recharge to the *Equus* Beds aquifer; however, at this location, the water table is usually above the uppermost clay layer, and therefore, flow of recharge water to the water table occurs rapidly. The water levels from the shallow and deep monitoring wells shown in figure 6 are the same. This is an indication of hydraulic connection between the upper sand-and-gravel layers and the lower layers at this site. Infiltration rates as high as 950 gal/min have been observed (Burns and McDonnell, written commun., 1998). High permeability of the sand-and-gravel layer at the site also contributes to rapid infiltration.



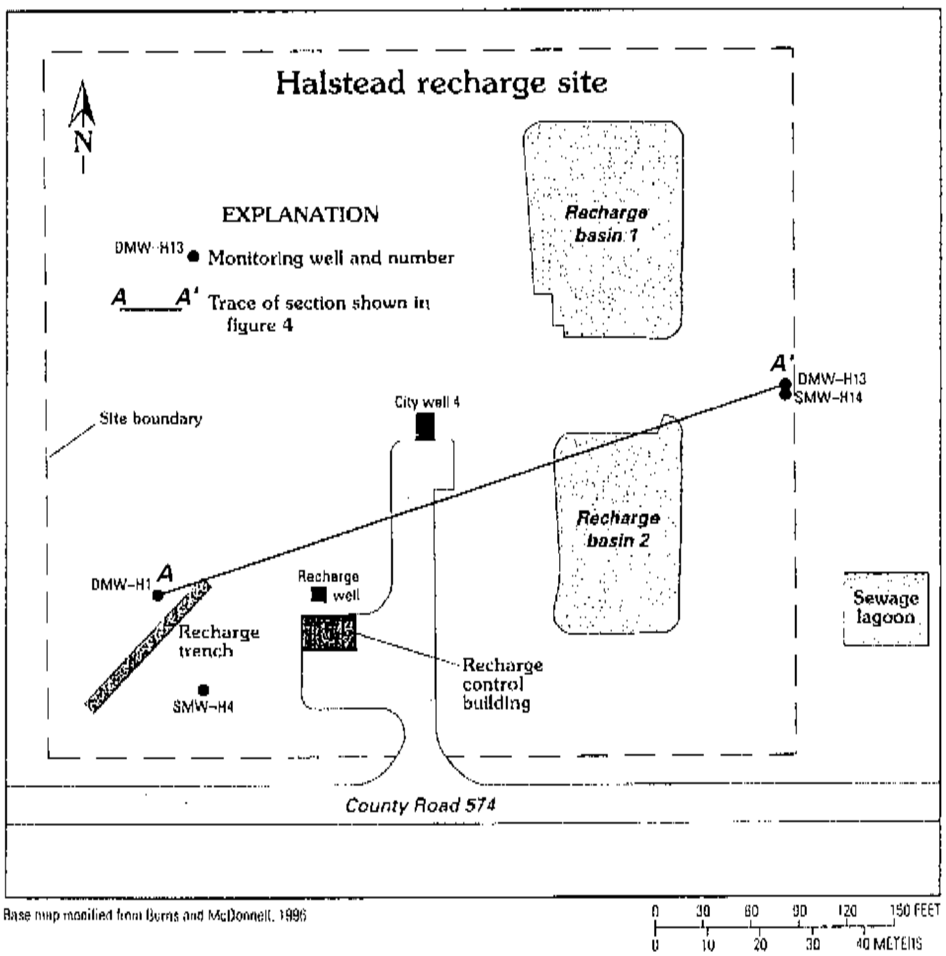


Figure 3. Location of data-collection sites at Halstead diversion well and recharge sites.

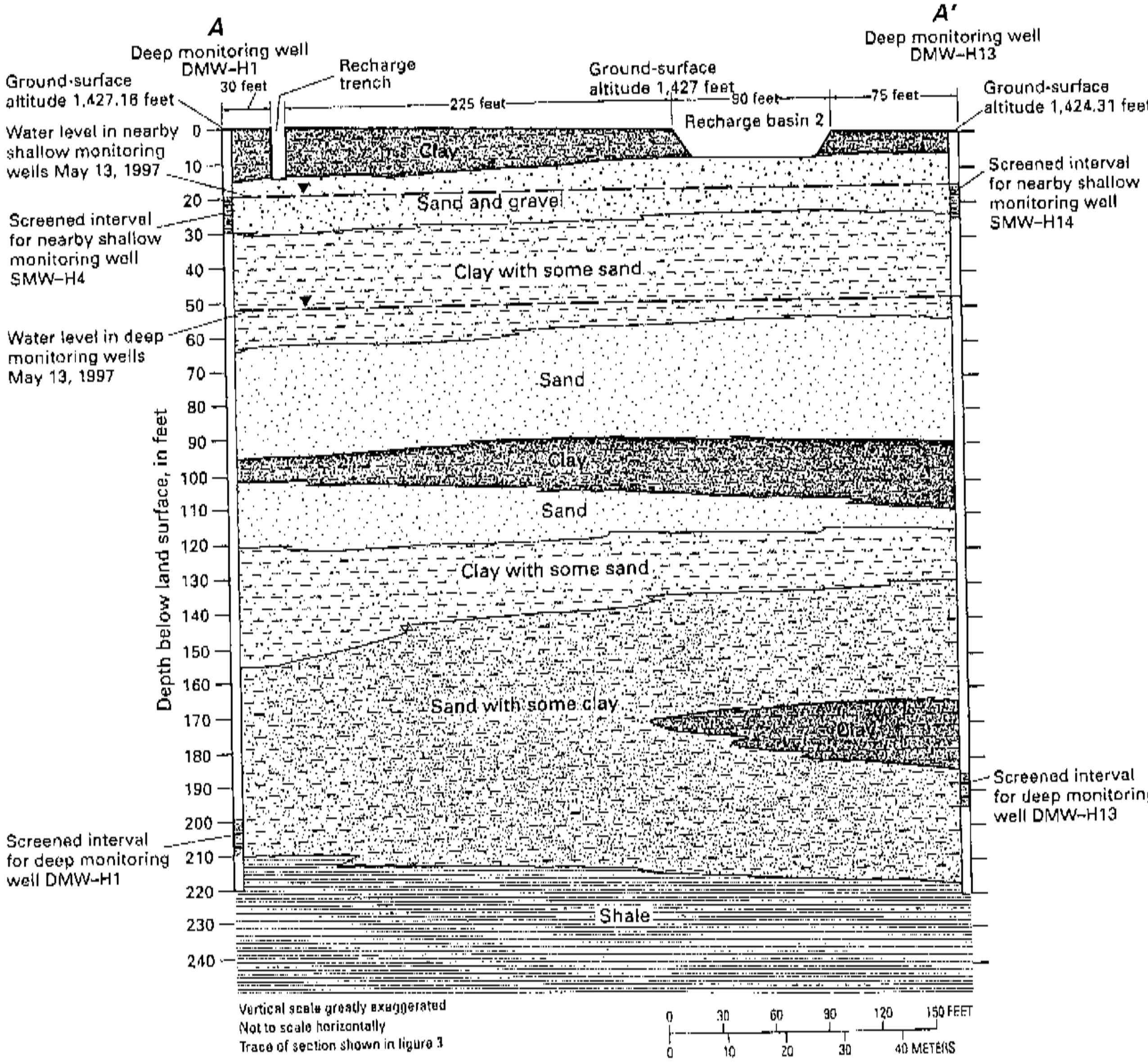


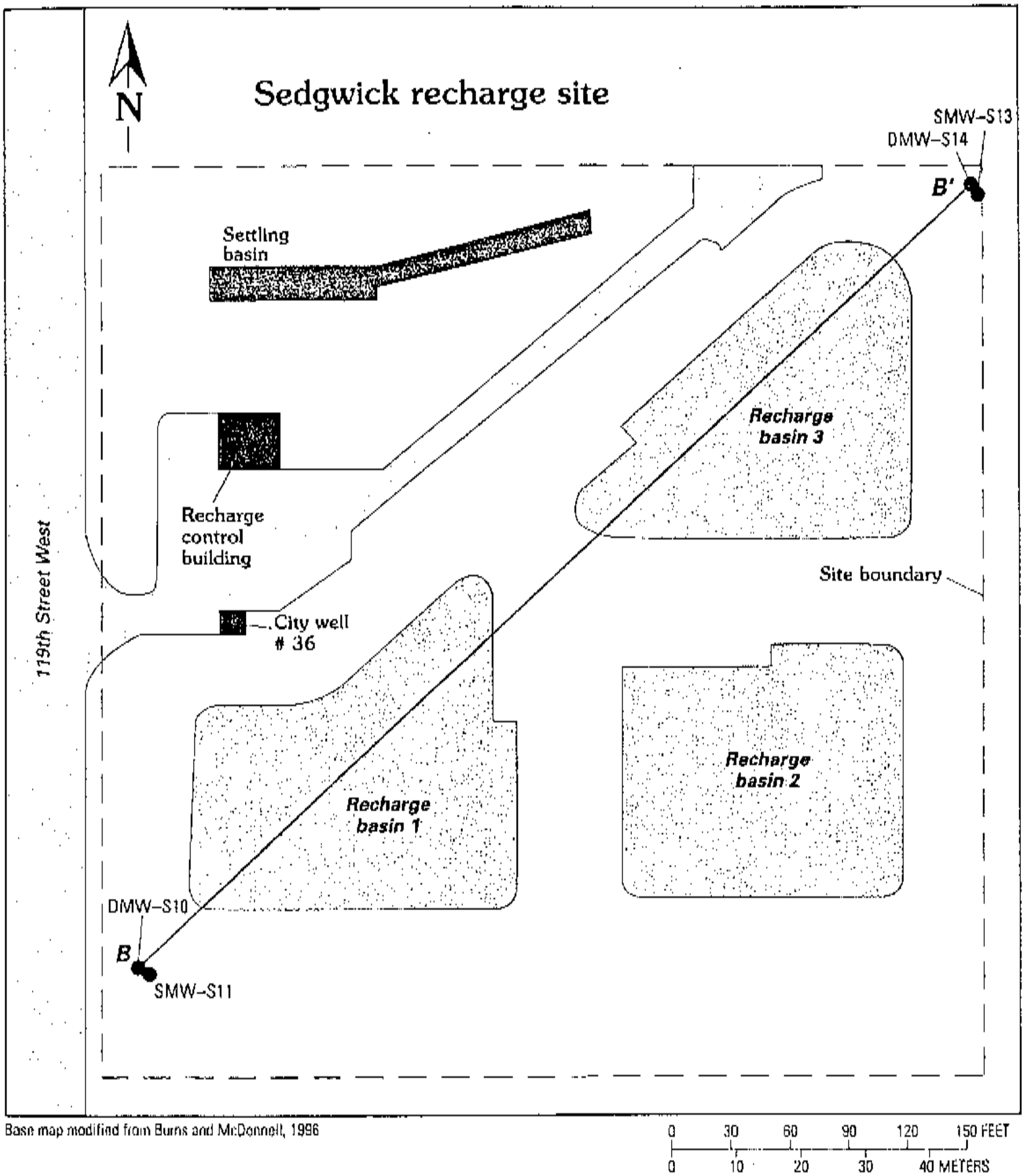
Figure 4. Hydrogeologic section between deep monitoring wells DMW–H1 and DMW–H13 at Halstead recharge site.

METHODS

The potential for water-quality degradation of an aquifer is a major concern for any artificial recharge project. For the *Equus* Beds Ground-Water Recharge Demonstration Project, surface- and ground-water quality are monitored frequently throughout the study area according to a monitoring plan established in consultation with State and Federal agencies. Surface-water quantity and quality are monitored at two USGS streamflow-gaging stations on the Little Arkansas River— Little Arkansas River at Highway 50 near Halstead (station 07143672, fig. 1) and Little Arkansas River near Sedgwick (station 07144100, fig. 1). Flow at both stations is affected by ground-water withdrawals, surface-water diversions, and return flow from irrigated areas (Putnam and others, 1997, p. 288 and 290).

Monitoring wells used in this study were installed by the city of Wichita and constructed of polyvinyl chloride pipe. Wells typically are screened in the lowermost 10 ft of the casing. For dates of completion, type of drill rig, development methods, and other information on individual monitoring wells, refer to Burns and McDonnell (1996).

Ground-water quality is monitored throughout the study area at the following data-collection sites: the Halstead diversion well site, consisting of five shallow



EXPLANATION

DMW–S10 ● Monitoring well and number

B — B' Trace of section shown in figure 6

Figure 5. Location of data-collection sites at Sedgwick recharge site.

monitoring wells (43–70 ft deep) and one deep monitoring well (120 ft deep); the Halstead recharge site, consisting of two shallow (27 and 29 ft deep) and two deep (220 ft deep) monitoring wells; the Sedgwick recharge site, consisting of two shallow (34.5 and 59 ft deep) and two deep (190 and 195 ft deep) monitoring wells; 12 background monitoring wells (40–59 ft deep) located immediately adjacent to the Little Arkansas River; and 10 domestic wells near the

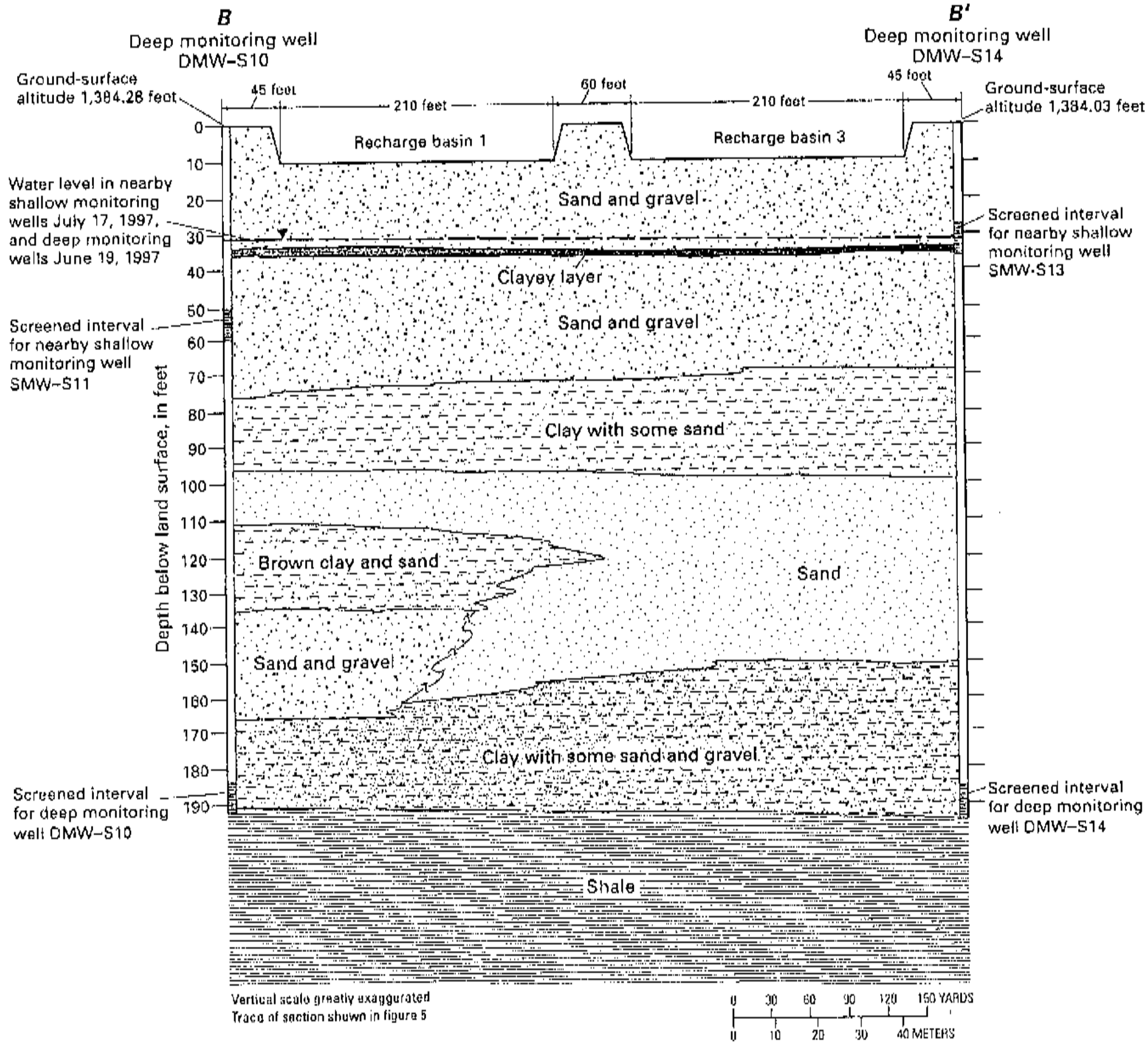


Figure 6. Hydrogeologic section between deep monitoring wells DMW–S10 and DMW–S14 at Sedgwick recharge site.

Halstead and Sedgwick recharge sites (generally less than 100 ft deep).

Background and domestic wells were used to provide baseline water-quality information on the shallow part of the *Equus* Beds aquifer in the study area. The background wells were designed to monitor water quality. However, domestic wells were designed to provide water supply to landowners and were not installed with the same specifications as the monitoring wells. This should be considered when comparing constituent concentrations for water samples from domestic wells to water samples from monitoring wells. Both shallow and deep wells were sampled during the demonstration project because the shallow and deep zones are geologically different and thus may react differently with the source water artificially recharged to the aquifer. Additional information about these wells, including altitude and screened interval, is given in table 1.

Sample collection began in February 1995 and continued at most data-collection sites through September 1997 to document baseline water quality in the Little Arkansas River and the *Equus* Beds aquifer. Samples to determine the preliminary effects of artificial recharge on the *Equus* Beds aquifer were collected from October 1997 through August 1998. Analysis of

Table 1. Data-collection sites for *Equus* Beds Ground-Water Recharge Demonstration Project, south-central Kansas, 1995–98

[--, not available; <, less than]

Site grouping	Data-collection site number (figs. 1, 2, 3, or 5)	U.S. Geological Survey site identification number	Site name	Gage datum or top of well casing altitude (feet above sea level)	Approximate well depth (feet)	Approximate depth of screened interval (feet)
Surface-water monitoring sites						
	07143672	07143672	Little Arkansas River at Highway 50 near Halstead, KS	1,370.55	--	--
	07143680	07143680	Little Arkansas River at Halstead, KS	1,371.00	--	--
	07144100	07144100	Little Arkansas River near Sedgwick, KS	1,340.00	--	--
Halstead diversion well site						
	Diversion well	380051097335501	Diversion well at TH–04–95	--	136.5	75.9–136.5
	EB–145–A1	380028097311001	Well #1 at TH–04–95	1,392.87	48	37.6–47.6
	EB–145–A2	380028097310901	Well #2 at TH–04–95	1,392.68	47	37–47
	EB–145–A3	380028097311101	Well #3 at TH–04–95	1,392.82	70	60–70
	EB–145–A4	380027097311401	Well #4 at TH–04–95	1,394.00	60	50–60
	EB–145–A5	380028097312701	Well #5 at TH–04–95	1,394.26	43	32–42
	EB–145–PD5	380028097311002	Monitoring well	1,392.40	120	112–117
Domestic wells near Halstead recharge site						
	DW–1	380047097335601	23S 02W 32BAAB01	--	90	--
	DW–2	380049097335401	23S 02W 32ABBB01	--	<100	--
	DW–3	380052097335301	23S 02W 29DCCC01	--	<100	--
	DW–4	380053097340501	23S 02W 29CDCC01	--	<100	--
	DW–5	380118097342601	23S 02W 30ADDD01	--	<100	--
Halstead recharge site						
	Source water diverted for recharge	380051097335501	Source water diverted for recharge—sampled at Halstead recharge site control building	--	--	--
	SMW–H14	380053097335101	Shallow monitoring well SMW–H14 near city well #4	1,424.25	27	17–27
	SMW–H4	380051097335601	Shallow monitoring well SMW–H4 near city well #4	1,429.22	29	19–29
	DMW–H1	380052097335701	Deep monitoring well DMW–H1 near city well #4	1,428.16	220	210–220
	DMW–H13	380053097335102	Deep monitoring well DMW–H13 near city well #4	1,424.31	220	210–220

Table 1. Data-collection sites for *Equus* Beds Ground-Water Recharge Demonstration Project, south-central Kansas, 1995–98—Continued

Site grouping	Data-collection site number (figs. 1, 2, 3, or 5)	U.S. Geological Survey site identification number	Site name	Gage datum or top of well casing altitude (feet above sea level)	Approximate well depth (feet)	Approximate depth of screened interval (feet)
Domestic wells near Sedgwick recharge site	DW–6	375258097285701	25S 02W 13AAAA01	--	<100	--
	DW–7	375310097282201	25S 01W 07DCBB01	--	<100	--
	DW–8	375344097274701	25S 01W 08BBCB01	--	<100	--
	DW–9	375356097284101	25S 01W 06CCDA01	--	<100	--
	DW–10	375357097285701	25S 02W 01DDDA01	--	<100	--
Sedgwick recharge site						
	Treated source water	375331097285301	Treated source water sampled at Sedgwick recharge site control building	--	--	--
	SMW–S11	375327097285401	Shallow monitoring well SMW–S11 near city well #36	1,383.27	59	49–59
	SMW–S13	375332097284801	Shallow monitoring well SMW–S13 near city well #36	1,383.27	34.5	24.5–34.5
	DMW–S10	375327097285402	Deep monitoring well DMW–S10 near city well #36	1,384.28	190	180–190
	DMW–S14	375332097284802	Deep monitoring well DMW–S14 near city well #36	1,384.03	195	185–195
Background wells						
	EB–142–A3	375300097253501	25S 01W 10CCCC01 Sedgwick well A3 EB–142–A3	1,363.63	57.5	47.2–57.2
	Sedgwick well	375259097252901	25S 01W 15BBBA01 Sedgwick well	1,370.38	49.3	39.1–49.1
	City well—Sedgwick	375456097260901	24S 01W 33DDA 01 city well—Sedgwick	--	--	--
	TH–02–95	380237097324401	23S 02W 16CDD 01 TH–02–95	1,400.75	54	43.6–53.6
	TH–06–95	375304097291301	24S 02W 01DCC 01 TH–06–95	1,380.18	41	30.6–40.6
	TH–10–95	380424097343801	23S 02W 06DDD 01 TH–10–95	1,407.88	59.1	48.7–58.7
	TH–12–95	375140097243301	24S 01W 23BCC 01 TH–12–95	1,359.00	50.3	39.9–49.9
	TH–08–A1	375628097270201	24S 01W 29ABAA01 TH–08–A1	1,359.58	40	30–40
	TH–08–A2	375628097270401	24S 01W 29ABAB01 TH–08–A2	1,379.26	53	43–53
	TH–08–A3	375628097270801	24S 01W 29ABAB01 TH–08–A3	1,379.96	59	48–58
	TH–08–A4	375628097271001	24S 01W 29ABBA01 TH–08–A4	1,380.08	58	46–56
	TH–08–A5	375628097271701	24S 01W 29BAAA01 TH–08–A5	1,378.53	53	42–52

Table 2. Time periods defining baseline and artificial recharge conditions for data-collection-site groupings

[--, not applicable]

Data-collection-site groupings	Time period	
	Baseline conditions	Artificial recharge conditions
Surface-water monitoring site 07143672	February 1995–September 1997	October 1997–July 1998
Surface-water monitoring site 07144100	February 1995–September 1997	October 1997–July 1998
Background wells	February 1995–September 1997	October 1997–July 1998
Domestic wells near Halstead	April 1998	--
Domestic wells near Sedgwick	August 1998	--
Halstead diversion well site	February 1995–March 1996	April 1996–July 1998
Halstead recharge site	May 13, 1997	May 29, 1997–July 1998
Sedgwick recharge site	June 1997–February 1998	April 1998–July 1998

surface and ground water was performed for dissolved solids, total and dissolved inorganic constituents, nutrients, organic compounds, volatile organic compounds (VOC's), radionuclides, and bacteria. Table 2 defines the time periods for baseline conditions and artificial recharge conditions for each of the data-collection site groupings. A complete list of constituents analyzed is given in table 3. Further information related to the data-collection sites, constituents analyzed, data-collection methods, sample frequency, preservation, holding times, and reporting limits can be found in Ziegler and Combs (1997). A preliminary determination of compatibility of recharge source water and receiving ground water was made through the examination of major-ion chemistry and comparison of particular constituents such as dissolved oxygen, iron, and manganese.

Most of the surface-water samples collected for analysis of triazine herbicides were obtained using automated samplers. Results of analyses of surface-water samples collected by automated samplers were compared to results of analyses of samples collected using depth- and width-integrating techniques (Ward and Harr, 1990). Ground-water samples were collected with a submersible pump, using methods described in Wood (1976), Koterba and others (1995), and Puls and Barcelona (1996).

Triazine herbicides were analyzed by enzyme-linked immunosorbent assay (ELISA) (Thurman and others, 1990). Selected samples were verified by gas chromatography/mass spectrometry (GC/MS). A previous study (Christensen and Ziegler, 1998a) indicated a good relation between ELISA-determined triazine concentrations and the GC/MS-determined atrazine concentrations in the artificial recharge study area. In fact, the slope of the regression line, 0.81, indicates that the results of the two analyses are similar for surface water (Christensen and Ziegler, 1998b). Therefore, triazine herbicide concentrations determined by ELISA are referred to as atrazine concentrations in this report. However, the relation between triazine herbicide concentrations determined by ELISA and atrazine concentrations determined by GC/MS is not acceptable for ground-water samples because concentrations of atrazine are more frequently equal to or less than the reporting limit for ELISA (0.10 μg/L, microgram per liter). Therefore, in figures 20, 27, and 28, atrazine concentrations determined by ELISA are reported for surface-water samples, whereas atrazine concentrations determined by GC/MS are reported for ground-water samples.

BASELINE WATER QUALITY, 1995–98

Information on the water quality of the *Equus* Beds aquifer and the Little Arkansas River before artificial recharge began was established as a basis for determining what effects, if any, artificial recharge would have on existing ground-water conditions. Baseline water-quality monitoring also was done to establish the constituents of concern for artificial recharge in the study area.

Surface- and ground-water samples were collected from February 1995 through September 1997 to document baseline concentrations of selected chemical constituents, except at the Halstead diversion well site where baseline water-quality monitoring ended in March 1996 because an aquifer test began in April 1996. At the Sedgwick recharge site, baseline water-quality monitoring was extended through February 1998 because recharge operations did not begin at this site until April 1998. In addition, baseline

Table 3. Constituents analyzed in water samples, 1995–98

Constituents analyzed in all nonautomated-collected samples		
Specific conductance	pH[1]	
Water temperature[1]	Oxygen, dissolved	
Hardness	Alkalinity as CaCO3, dissolved	
Dissolved solids	Calcium, dissolved[2]	
Magnesium, dissolved[2]	Sodium, dissolved	
Potassium, dissolved	Bicarbonate, dissolved	
Carbonate, dissolved	Sulfate, dissolved[3]	
Chloride, dissolved	Nitrite plus nitrate as nitrogen, dissolved[3]	
Total phosphorous	Iron, dissolved	
Manganese, dissolved	Triazine herbicide screen, dissolved	
Total coliform3	Fecal coliform	
Suspended solids		
Inorganic constituents analyzed in selected samples		
Turbidity	Fluoride, dissolved[3]	
Silica, dissolved	Nitrite, dissolved[3]	
Ammonia, dissolved	Orthophosphate, dissolved	
Antimony, dissolved[3]	Arsenic, dissolved[3]	
Barium, dissolved[3]	Beryllium, dissolved[3]	
Boron, dissolved	Bromide, dissolved	
Cadmium, dissolved[3]	Chromium, dissolved[3]	
Copper, dissolved[3]	Cyanide, dissolved[3]	
Lead, dissolved[3]	Mercury, dissolved[3]	
Nickel, dissolved[3]	Selenium, dissolved[3]	
Silver, dissolved	Strontium, dissolved	
Thallium, dissolved[3]	Vanadium, dissolved	
Zinc, dissolved	Total organic carbon	
Total solids		
Dissolved pesticides and metabolites analyzed in selected samples		
2,4 -DB	2,4- D[3]	2,4, 5-T
Acifluorfen	Alachlor[3]	Aldicarb
Aldicarb sulfone	Aldicarb sulfoxide	Atrazine[3]
Azinphos, methyl	Benfluralin	Bentazon
Bromacil	Bromoxynil	Butylate

Table 3. Constituents analyzed in water samples, 1995–98—Continued

Dissolved pesticides and metabolites analyzed in selected samples—Continued		
Carbaryl	Carbofuran[3]	Carbofuran-3-hydroxy
Chloramben	Chlorothalonil	Chlorpyrifos
Clopyralid	Cyanazine	Dacthl-mono-acid
DCPA	DDE	Deethylatrazine
Deisopropylatrazine	Diazinon	Dicamba
Dichlobenil	Dichlorprop (2,4-DP)	Dieldrin
Diethylanaline	Dimethoate	Dinoseb[3]
Disulfoton	Diuron	DNOC
EPTC	Esfenvalerate	Ethalfluralin
Ethoprop	Fenuron	Fluometuron
Fonofos	HCH-alpha	HCH-alpha D6
HCH-gamma	Linuron	Malathion
MCPA	MCPB	Methiocarb
Methomyl	Methyl parathion	Metolachlor
Metribuzin	Molinate	1-Naphthol
Napropamide	Neburon	Norflurazon
Oryzalin	Oxamyl[3]	Parathion
Pebulate	Pendimethalin	Permethrin-cis
Phorate	Picloram[3]	Prometon
Pronamide	Propachlor	Propanil
Propargite	Propham	Propoxur
Silvex (2,4,5-TP)[3]	Simazine[3]	Tebuthiuron
Terbacil	Terbufos	Terbuthylazine
Thiobencarb	Triallate	Triclopyr
Trifluralin		
Total recoverable volatile organic compounds analyzed in selected samples		
Acrolein	Acrylonitrile	Benzene[3]
Bromobenzene	Bromochloromethane	Bromodichloromethane[3]
Bromoform[3]	Bromomethane	n-Butylbenzene
sec-Butylbenzene	tert-Butylbenzene	Carbon tetrachloride[3]
Chlorobenzene[3]	Chloroethane	2-Chloroethylvinyl ether
Chloroform[3]	Chloromethane	1,2-Chlorotoluene
1,4-Chlorotoluene	Dibromochloromethane[3]	1,2-Dibromo-3-chloropropane[3]
Dibromomethane	1,2-Dibromoethane (EDB)[3]	1,2-Dichlorobenzene
1,3-Dichlorobenzene[3]	1,4-Dichlorobenzene[3]	Dichlorodifluoromethane
1,1-Dichloroethane	1,2-Dichloroethane[3]	1,1-Dichloroethene[3]

Table 3. Constituents analyzed in water samples, 1995--98—Continued

Total recoverable volatile organic compounds analyzed in selected samples—Continued		
cis-1,2-Dichloroethene[3]	1,2-trans-Dichloroethene[3]	1,2-Dichloropropane[3]
1,3-Dichloropropane	2,2-Dichloropropane	1,1-Dichloropropene
cis-1,3-Dichloropropene	trans-1,3-Dichloropropene	1,3-Dichloropropylene
Ethylbenzene[3]	Hexachlorobutadiene	Isopropylbenzene
p-Isopropyltoluene	Methylene chloride[3]	Methyltertbutylether
Naphthalene	n-Propylbenzene	Styrene[3]
1,1,1,2-Tetrachloroethane	1,1,2,2-Tetrachloroethane	Tetrachloroethene[3]
Toluene	1,2,3-Trichlorobenzene	1,2,4-Trichlorobenzene[3]
1,1,1-Trichloroethane[3]	1,1,2-Trichloroethane[3]	Trichloroethene[3]
Trichlorofluoromethane	1,2,3-Trichloropropane	Trichlorotrifluoroethane
1,2,4-Trimethylbenzene	1,3,5-Trimethylbenzene	Vinyl chloride[3]
Xylene (o,p,m)[3]		
Dissolved radionuclides analyzed in selected samples		
Gross beta radiation[3]	Gross alpha radiation[3]	
Total recoverable organochlorine and organophosphate pesticides analyzed in baseline samples		
Aldrin	Chlordane[3]	Chlorpyrifos
DDD	DDE	DDT
DEF	Diazinon	Dieldrin
Disulfoton	Endosulfan alpha (I)	Endrin[3]
Ethion	Fonofos	Heptachlor[3]
Heptachlor epoxide[3]	Lindane[3]	Malathion
Methoxychlor[3]	Methylparathion	Mirex
Parathion	PCBs, gross[3]	PCNs, gross
Perthane	Phorate	Toxaphene[3]
Trithion		
Acid compounds analyzed in baseline samples		
p-Chloro-m-cresol	2-Chlorophenol	2,4-Dichlorophenol
2,4-Dimethylphenol	4,6-Dinitro-o-cresol	2-Nitrophenol
4-Nitrophenol	Pentachlorophenol3	Phenol
1,2,4-Trichlorobenzene	2,4,6-Trichlorophenol	

Table 3. Constituents analyzed in water samples, 1995–98—Continued

Base/neutral compounds analyzed in baseline samples		
Acenaphthene	Acenapthylene	Anthracene
Benzidine	Benzo(a)anthracene[3]	Benzo(a)pyrene[3]
3,4-benzofluoroanthene	2,4-benzo(ghi)perylene	Benzo(k)fluoranthene[3]
Bis (2-chloroethoxy) methane	Bis (2-chloroethyl) ether	Bis (2-chloroisopropyl) ether
Bis (2-ethylhexyl) phthalate	4-Bromophenyl phenyl ether	Butylbenzyl phthalate
2-Chloronaphthalene	4-Chlorophenyl phenyl ether	Chrysene[3]
Dibenzo(a,h)anthracene[3]	1,2-Dichlorobenzene	1,3-Dichlorobenzene
1,4-Dichlorobenzene	3,3'-Dichlorobenzidine	Diethyl phthalate
Dimethyl phthalate	Di-n-butyl phthalate[3]	2,4-Dinitrotoluene
2,6-Dinitrotoluene	Di-n-octyl phthlate	1,2-Diphenylhydrazine
Fluoranthene	Fluorene	Hexachlorobenzene[3]
Hexachlorobutadiene	Hexachlorocyclopentadiene[3]	Hexachloroethane
Indeno (1,2,3-cd) pyrene	Isophorone	Napthalene
Nitrobenzene	N-nitrosodimethylamine	N-nitrosodi-n-propylamine
N-nitrosodiphenylamine	Phenanthrene	Pyreoe

[1]Analyzed immediately after sample collection.
[2]Required for calculation of hardness.
[3]On U.S. Environmental Protection Agency (1999) Maximum Contaminant Level list.

water-quality monitoring of domestic wells near Sedgwick continued until August 1998.

Summary results of baseline water-quality sampling are presented in tables 4–10. Summary tables give the range in detected concentrations, the number of samples analyzed, and the median concentrations (where applicable) for physical properties and selected constituents (table 4), filtered major ions (table 5), nutrients (table 6), selected trace elements (table 7), total organic carbon (table 8), and total coliform bacteria (table 9) in samples from surface- and ground-water data-collection sites. Table 10 displays the range in detected concentrations, number of samples analyzed, and the median concentrations (where applicable) for all organic compounds in filtered samples. Individual data values for all samples collected are on file at the USGS office in Lawrence, Kansas.

Filtered constituents were reported because Ziegler and others (1997) found that in an artificial recharge study it was more appropriate and cost effective to analyze samples for filtered rather than total-recoverable concentrations. In this recharge study, sediment is removed from surface water before it is recharged through basins or trenches. In addition, onsite turbidity measurements in ground-water samples are required to be less than 10 NTU (nephelometric turbidity units), making analysis of total-recoverable concentrations unnecessary for inorganic and most organic compounds (Ziegler and Combs, 1997). However, total concentrations are used for total organic carbon, VOC's, acid and base/neutral organic compounds, and total coliform bacteria.

Although human activities can affect the concentrations of an inorganic constituent, natural concentrations may be large. Therefore, not all detected inorganic constituents are reported in tables 4–9. Inorganic constituents are listed in those tables if the concentration in any sample was larger than 20 percent of the USEPA's Maximum Contaminant Level (MCL), the Secondary Maximum Contaminant Level (SMCL), the Drinking-Water Equivalent Level (DWEL), or the Health Advisory Level (HAL) for that constituent. In addition, some properties and constituents with no MCL, SMCL, DWEL, or HAL are included in tables 4–9 if they are of particular interest for operation or design of recharge facilities and (or) are needed to describe the water chemistry.

Table 10 reports organic compounds that were detected in any sample. Different reporting guidelines were used for these tables because many organic compounds, such as pesticides, do not occur naturally in the environment but result from human activity. VOC's, acid, or base/neutral organic compounds, were not detected in any samples from any data-collection site.

Selected constituents from tables 4–10 are discussed in the following sections. Each selected constituent is examined in terms of the concentrations in surface water and ground water. Constituents of concern for artificial recharge activities are defined, especially as related to frequent large concentrations, relative to the regulatory criteria for drinking water (MCL, SMCL, DWEL, or HAL), in surface water — the source water for recharge.

Physical Properties and Selected Constituents

Physical properties and selected constituents summarized in table 4 for baseline water-quality conditions are specific conductance, pH, water temperature, turbidity, dissolved oxygen, total hardness, alkalinity, suspended solids, and dissolved solids. It is important to consider the physical properties of a water sample because these properties are unique in a number of respects and sometimes are affected by other properties. For example, dissolved ions have a tendency to increase specific conductance (Hem, 1992).

The only physical properties or constituents that have a regulatory criterion are pH, laboratory turbidity, and dissolved solids. The range in pH for all sites during baseline monitoring was 4.4 to 8.6 (standard units). The SMCL's acceptable range is 6.5 to 8.5 (U.S. Environmental Protection Agency, 1999).

Both onsite turbidity and laboratory turbidity are reported in table 4. Onsite turbidity, which is typically smaller than laboratory turbidity, is measured because a ground-water sample is required to have an onsite turbidity not greater than 10 NTU for water-quality analyses (Ziegler and Combs, 1997). There were four instances when onsite turbidity exceeded 10 NTU. Two of these occurred at the Sedgwick recharge site in wells that were recently drilled (wells SMW–S13 and SMW–S11), one occurred in domestic well DW–10 near Sedgwick, and one occurred in the first sample collected from background well TH–08–A1. Laboratory turbidity was measured at greater than the MCL for drinking water of 0.5 to 1.0 NTU in water from some data-collection sites. The range in laboratory turbidity in baseline surface water was 0.30 to 1,200 NTU. The range in water from wells was 0.13 to 1,300 NTU. These larger laboratory turbidities in ground water probably are a result of iron precipitates being formed after sampling. The value of 1,300 NTU occurred in water from a shallow well at the Sedgwick recharge site (well SMW–S13) shortly after the well was drilled. The well may not have been completely developed, which also may account for the larger laboratory turbidity.

Dissolved solids concentrations exceeded regulatory criteria at most sites. Dissolved solids concentration is the total amount of dissolved material in the water and can be attributed to the dissolved major ions present. In the study area, the large dissolved solids concentrations detected during baseline conditions were associated with large sodium, bicarbonate, and chloride concentrations. These and other major ions are discussed in the following section.

Major Ions

Major ions result primarily from the dissolution of rocks and minerals or from discharges of municipal or industrial sources. Excessively large concentrations of major ions are objectionable in drinking water because of possible physiological effects, unpalatable mineral tastes, and greater costs because of corrosion or the need for additional treatment (U.S. Environmental Protection Agency, 1986). Regulatory criteria have been assigned for sodium, sulfate, chloride, and fluoride (table 5).

Fluoride did not exceed its MCL of 4.0 mg/L in any baseline sample. However, sodium, sulfate, and chloride exceeded their respective regulatory criteria at some data-collection sites during baseline sampling. Sodium and chloride were detected at values greater than their DWEL and SMCL of 20 and 250 mg/L, respectively, during baseline water-quality monitoring. The range of sodium in surface-water samples was 4.4 to 200 mg/L; the range in water from wells was 14 to 150 mg/L. Figure 7 shows the ranges in sodium concentrations of the baseline samples. Individual data points were used when there were less than six water samples from the site. The concentration of sodium in

Table 4. Summary of determinations of physical properties and analyses of selected constituents in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCL, Maximum Contaminant Level; SMCL, Secondary Maximum Contaminant Level; µS/cm, microsiemens per centimeter at 25 degrees Celsius; NTU, nephelometric turbidity units; mg/L milligrams per liter; *, median value is estimated by using a log-probability regression to predict the values of data less than the detection limit; <, less than; >, greater than; --, not determined. The first set of numbers under each heading (for example, 99–2,000) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 860) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Specific conductance (µS/cm)	pH (standard units)	Water temperature (degrees Celsius)	Onsite turbidity (NTU)	Laboratory turbidity (NTU)	Dissolved oxygen (mg/L)	Total hardness (mg/L)	Alkalinity, as $CaCO_3$ (mg/L)	Suspended solids, residue, total (mg/L)	Dissolved solids (mg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria											
		--	6.5–8.5 (SMCL)	--	--	0.5–1.0 (MCL)	--	--	--	--	500 (SMCL)
Surface-water monitoring sites (Little Arkansas River)											
07143672	Baseline	99–2,000 (361)	6.2–8.4 (78)	1.9–27 (71)	--	0.3–1,200 (56)	2.5–14 (66)	27–490 (70)	27–310 (70)	4.0–1,600 (56)	88–1,100 (70)
(near Halstead)	(February 1995 through September 1997)	860	7.7	17	--	59	7.5	320	230	69	590
	Recharge	190–3,600 (35)	6.9–8.3 (36)	0.22–27 (36)	17, 21 (2)	5.1–1,100 (15)	4.7–16 (34)	54–580 (33)	44–290 (33)	4.0–780 (15)	110–2,000 (33)
	(October 1997 through July 1998)	1,000	7.7	6.3	--	100	11	350	200	80	620
07144100	Baseline	40–1,600 (654)	6.1–8.6 (62)	1.6–27 (55)	--	1.4–800 (51)	4.4–19 (51)	38–410 (51)	39–290 (51)	4.0–850 (51)	92–760 (51)
(near Sedgwick)	(February 1995 through September 1997)	630	7.6	17	--	69	7.6	220	160	90	400
	Recharge	210–1,200 (212)	7.0–8.7 (31)	-0.20–32 (29)	19–120 (7)	5.3–1,000 (16)	5.6–16 (27)	56–390 (22)	18–280 (22)	5.2–940 (20)	110–730 (18)
	(October 1997 though July 1998)	700	7.9	14	--	110	9.5	240	180	140	380
Ground-water or diverted-water monitoring sites											
Background wells											
	Baseline	350–1,000 (45)	6.1–7.4 (43)	15–17 (43)	0.03–26 (39)	0.13–48 (38)	0.03–0.80 (42)	190–360 (38)	180–300 (38)	0.8–100 (38)	220–560 (38)
	(February 1995 through September 1997)	630	6.8	16	0.64	0.6	0.15	240	210	1.4*	380
	Recharge	600–780 (7)	6.6–7.2 (7)	15–16 (7)	<0.10–0.92 (7)	0.19–4.2 (7)	0.11–0.82 (7)	230–390 (7)	190–290 (7)	<4.0 (7)	380–460 (7)
	(October 1997 through July 1998)	620	7.0	15	0.27*	0.35	0.28	250	270	--	400
Domestic wells near Halstead											
	Baseline	360–810 (5)	6.3–7.0 (5)	12–16 (5)	0.02–1.1 (5)	0.15–6.1 (5)	0.07–7.5 (5)	120–260 (5)	100–220 (5)	<4.0 (5)	200–480 (5)
	(April 1998)	--	--	--	--	--	--	--	--	--	--

Table 4. Summary of determinations of physical properties and analyses of selected constituents in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Specific conductance (μS/cm)	pH (standard units)	Water temperature (degrees Celsius)	Onsite turbidity (NTU)	Laboratory turbidity (NTU)	Dissolved oxygen (mg/L)	Total hardness (mg/L)	Alkalinity, as $CaCO_3$ (mg/L)	Suspended solids, residue, total (mg/L)	Dissolved solids (mg/L)
Ground-water or diverted-water monitoring sites—Continued											
Domestic wells near Sedgwick											
	Baseline	680–1,600 (5)	4.4–7.1 (5)	15–22 (5)	0.20–25 (5)	0.13–20 (5)	0.38–3.7 (5)	260–570 (5)	10–330 (5)	<4.0–50 (5)	380–1,000 (5)
	(August 1998)	--	--	--	--	--	--	--	--	--	--
Halstead diversion well site—diversion well											
	Recharge	540–840 (337)	6.2–7.4 (337)	5–19 (334)	<0.10–28 (171)	0.25–5.2 (59)	0.002–4.7 (295)	180–320 (100)	20–280 (98)	<4.0–170 (63)	290–530 (98)
	(April 1996 through July 1998—only sampled during recharge activities)	780	7.1	15	2.6*	1.2	0.28	270	260	--	440
Halstead diversion well site—shallow monitoring wells											
	Baseline	890–1,400 (12)	6.5–6.9 (12)	14–16 (12)	0.16–3.1 (10)	0.36–20 (11)	0.05–0.25 (11)	210–390 (11)	250–360 (11)	<4.0–8.8 (11)	500–710 (11)
	(February 1995 through March 1996)	1,100	6.8	15	0.42	13	0.09	320	320	1.6*	640
	Recharge	580–1,400 (82)	6.0–7.2 (82)	10–19 (82)	<0.10–1.0 (66)	0.13–40 (80)	0.06–1.1 (81)	150–450 (80)	170–420 (80)	<4.0–<20 (78)	300–940 (80)
	(April 1996 through July 1998)	1,000	6.7	16	0.14	2.6	0.16	330	290	1.6*	640
Halstead diversion well site—deep monitoring well											
	Baseline	540–560 (3)	7.0–7.3 (3)	15–16 (3)	0.05–0.92 (3)	0.20–0.52 (3)	0.04–0.06 (3)	170–180 (3)	220–230 (3)	<4.0 (3)	300–350 (3)
	(February 1995 through March 1996)	--	--	--	--	--	--	--	--	--	--
	Recharge	450–780 (24)	6.2–7.3 (24)	15–16 (24)	0.02–0.90 (21)	0.16–1.1 (24)	0.04–0.63 (23)	140–260 (24)	190–260 (24)	<4.0 (24)	280–460 (24)
	(April 1996 through July 1998)	490	7.0	16	0.10	0.38	0.12	160	200	--	310
Halstead recharge site—shallow monitoring wells											
	Baseline	470, 1,400 (2)	5.6, 6.4 (2)	14, 15 (2)	0.75, 3.3 (2)	0.47, 1.1 (2)	5.7, 6.5 (2)	150, 310 (2)	140, 190 (2)	<4.0 (2)	320, 830 (2)
	(May 13, 1997)	--	--	--	--	--	--	--	--	--	--
	Recharge	640–810 (31)	5.3–7.2 (31)	13–19 (31)	0.09–3.2 (28)	0.12–1.3 (31)	3.3–9.2 (31)	200–300 (31)	190–270 (31)	<4.0–46 (31)	390–470 (31)
	(May 29, 1997 through July 1998)	730	6.8	15	0.58	0.50	5.9	250	240	--	430

Table 4. Summary of determinations of physical properties and analyses of selected constituents in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Specific conductance (µS/cm)	pH (standard units)	Water temperature (degrees Celsius)	Onsite turbidity (NTU)	Laboratory turbidity (NTU)	Dissolved oxygen (mg/L)	Total hardness (mg/L)	Alkalinity, as $CaCO_3$ (mg/L)	Suspended solids, residue, total (mg/L)	Dissolved solids (mg/L)
				Ground-water or diverted-water monitoring sites—Continued							
Halstead recharge site—deep monitoring wells											
	Baseline	340, 370 (2)	7.0, 6.5 (2)	16, 16 (2)	1.0, 2.8 (2)	0.60, 10 (2)	0.12, 0.29 (2)	110, 120 (2)	160, 180 (2)	<4.0, 24 (2)	190, 220 (2)
	(May 13, 1997)	--	--	--	--	--	--	--	--	--	--
	Recharge	330–830 (30)	5.9–7.2 (30)	15–16 (30)	<0.10–3.8 (27)	0.18–8.5 (30)	0.05–0.61 (30)	120–320 (30)	150–270 (30)	<4.0–36 (30)	180–500 (30)
	(May 29, 1997 through July 1998)	700	7.0	16	0.18*	0.80	0.14	240	240	--	400
Sedgwick recharge site—treated diverted source water											
	Recharge	270–1,100 (63)	6.1–8.8 (62)	11–31 (54)	1.3–1,000 (35)	3.4–28 (16)	4.2–13 (40)	83–340 (16)	25–240 (16)	<4.0–48 (62)	160–620 (16)
	(April 1998 through July 1998--only sampled during recharge activities)	660	8.1	23	6.1	8.5	8.0	210	160	11*	380
Sedgwick recharge site—shallow monitoring wells											
	Baseline	550–880 (11)	6.3–7.0 (11)	15–18 (11)	0.19–>200 (11)	0.32–1,300 (11)	3.1–7.3 (10)	240–440 (11)	130–190 (11)	<4.0–4,000 (11)	350–600 (11)
	(June 1997 through February 1998)	700	6.5	16	0.94	0.73	5.8	300	150	--	460
	Recharge	510–740 (10)	6.3–7.2 (10)	16–29 (10)	0.22–8.5 (10)	0.24–1.1 (10)	5.2–7.2 (10)	170–310 (10)	110–170 (10)	<4.0–14 (10)	300–510 (10)
	(April 1998 through July 1998)	680	6.5	16	0.52	0.63	6.5	280	140	--	430
Sedgwick recharge site—deep monitoring wells											
	Baseline	710–800 (9)	7.0–7.3 (9)	16–16 (9)	<0.10–3.7 (9)	0.15–2.5 (9)	0.05–0.69 (9)	210–260 (9)	250–270 (9)	<4.0–4.4 (9)	430–490 (9)
	(June 1997 through April 1998)	790	7.2	16	0.16*	0.43	0.20	240	270	--	460
	Recharge	620–820 (8)	6.9–7.4 (8)	16–16 (8)	0.11–1.1 (7)	0.08–1.2 (8)	0.05–0.49 (6)	160–240 (8)	200–280 (8)	<4.0 (8)	400–510 (8)
	(April 1998 through July 1998)	750	7.2	16	0.78	0.37	0.12	210	260	--	440

Table 5. Summary of major-ion concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCL, Maximum Contaminant Level DWEL, Drinking-Water Equivalent Level; SMCL Secondary Maximum Contaminant Level; mg/L, milligrams per liter; <, less than; *, median value is estimated by using a log-probability regression to predict values less than the detection limit; --, not determined. The first set of numbers under each heading (for example, 8.5–150) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 96) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Calcium (mg/L)	Magnesium (mg/L)	Sodium (mg/L)	Potassium (mg/L)	Bicarbonate (mg/L)	Sulfate (mg/L)	Chloride (mg/L)	Fluoride (mg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria									
		--	--	20 (DWEL)	--	--	250 (SMCL)	250 (SMCL)	4.0 (MCL)
Surface-water monitoring sites (Little Arkansas River)									
07143672	Baseline	8.3–150 (70)	1.6–26 (70)	4.4–200 (70)	5.4–18 (70)	33–380 (70)	<5.0–75 (70)	8–400 (70)	0.10–2.7 (19)
(near Halstead)	(February 1995 through September 1997)	96	14	83	8.4	250	41*	150	0.30
	Recharge	16–170 (33)	3.3–36 (33)	11–500 (33)	4.6–9.5 (33)	54–460 (33)	9.1–96 (33)	20–930 (34)	0.29, 0.29 (2)
	(October 1997 through July 1998)	110	18	100	7.4	260	53	190	--
07144100	Baseline	12–120 (51)	2.2–24 (51)	4.8–120 (51)	5.2–10 (51)	48–360 (51)	<5.0–210 (51)	8–260 (51)	0.12–0.82 (16)
(near Sedgwick)	(February 1995 through September 1997)	66	12	48	7.8	190	40	68	0.28
	Recharge	17–120 (22)	3.0–24 (22)	11–110 (22)	5.1–9.2 (22)	22–340 (22)	11–71 (22)	17–190 (23)	0.19–0.28 (3)
	(October 1997 through July 1998)	70	12	48	7.2	220	39	93	--
Ground-water or diverted-water monitoring sites									
Background wells									
	Baseline	45–120 (38)	4.4–18 (38)	21–91 (38)	1.4–5.7 (38)	210–360 (38)	<5.0–190 (38)	<5–65 (38)	<0.02–0.41 (26)
	(February 1995 through September 1997)	74	12	44	3.0	250	77	26*	0.33*
	Recharge	72–94 (7)	12–15 (7)	42–74 (7)	2.7–3.4 (7)	230–350 (7)	60–100 (7)	25–49 (7)	0.28, 0.32 (2)
	(October 1997 through July 1998)	77	13	46	3.2	330	82	27	--
Domestic wells near Halstead									
	Baseline	41–80 (5)	4.8–15 (5)	24–75 (5)	2.2–2.7 (5)	120–270 (5)	16–63 (5)	11–69 (5)	--
	(April 1998)	--	--	--	--	--	--	--	--

Table 5. Summary of major-ion concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Calcium (mg/L)	Magnesium (mg/L)	Sodium (mg/L)	Potassium (mg/L)	Bicarbonate (mg/L)	Sulfate (mg/L)	Chloride (mg/L)	Fluoride (mg/L)
	Ground-water or diverted-water monitoring sites—Continued								
Domestic wells near Sedgwick									
	Baseline	83–180 (5)	12–31 (5)	19–130 (5)	2.3–5.4 (5)	12–400 (5)	80–350 (5)	5–65 (5)	--
	(August 1998)	--	--	--	--	--	--	--	--
Halstead diversion well site—diversion well									
	Recharge	57–100 (98)	8.1–15 (98)	51–72 (98)	1.8–2.7 (98)	260–340 (98)	5.0–78 (98)	22–78 (264)	<0.02–0.30 (17)
	(April 1996 through July 1998—only sampled during recharge activities)	87	12	61	2.3	320	46	59	0.21
Halstead diversion well site—shallow monitoring wells									
	Baseline	68–130 (11)	9.1–17 (11)	75–125 (11)	2.8–5.5 (11)	330–440 (11)	<5.0–35 (11)	85–220 (11)	0.15–0.45 (9)
	(February 1995 through March 1996)	110	14	89	4.6	400	12*	140	0.36
	Recharge	50–150 (80)	6.9–20 (80)	50–120 (80)	2.0–8.4 (80)	200–510 (80)	3.0–180 (78)	12–280 (80)	0.02–0.52 (19)
	(April 1996 through July 1998)	110	14	83	3.9	350	14*	150	0.18
Halstead diversion well site—deep monitoring well									
	Baseline	56–62 (3)	84–88 (3)	50–53 (3)	2.0–2.1 (3)	270–280 (3)	38–40 (3)	14–15 (3)	0.30, 0.30 (2)
	(February 1995 through March 1996)	--	--	--	--	--	--	--	--
	Recharge	45–84 (24)	6.4–12 (24)	49–70 (24)	1.7–2.5 (24)	230–320 (24)	<5.0–33 (24)	16–87 (24)	0.19–0.32 (5)
	(April 1996 through July 1998)	50	7.2	50	1.9	240	31	30	--
Halstead recharge site—shallow monitoring wells									
	Baseline	47, 97 (2)	7.8, 16 (2)	40, 150 (2)	1.4, 4.0 (2)	180, 230 (2)	26, 30 (2)	20, 290 (2)	0.14, 0.32 (2)
	(May 13, 1997)	--	--	--	--	--	--	--	--
	Recharge	62–94 (31)	11–16 (31)	45–100 (31)	2.0–3.7 (31)	230–330 (31)	32–58 (31)	39–110 (31)	<0.02–0.54 (17)
	(May 29, 1997 through July 1998)	80	13	62	2.4	290	48	59	0.33*

Table 5. Summary of major-ion concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Calcium (mg/L)	Magnesium (mg/L)	Sodium (mg/L)	Potassium (mg/L)	Bicarbonate (mg/L)	Sulfate (mg/L)	Chloride (mg/L)	Fluoride (mg/L)
Ground-water or diverted-water monitoring sites—Continued									
Halstead recharge site—deep monitoring wells									
	Baseline	37, 37 (2)	4.4, 6.1 (2)	29, 31 (2)	1.9, 2.0 (2)	200, 220 (2)	9.9, 19 (2)	5.8, 8.3 (2)	0.30, 0.35 (2)
	(May 13, 1997)	--	--	--	--	--	--	--	--
	Recharge	39–100 (29)	4.6-17 (29)	27–69 (29)	2.0–3.5 (29)	180–330 (29)	9.7–73 (29)	5.8–64 (29)	<0.02–0.37 (16)
	(May 29, 1997 through July 1998)	79	10	55	2.7	290	42	50	0.24*
Sedgwick recharge site—treated diverted source water									
	Recharge	26–100 (16)	4.7–20 (16)	21–81 (16)	5.7–8.4 (16)	30–300 (16)	17–62 (16)	26–180 (62)	0.20–0.30 (3)
	(April 1998 through July 1998—only sampled during recharge activities)	65	13	50	6.8	190	38	82	--
Sedgwick recharge site—shallow monitoring wells									
	Baseline	82–140 (11)	9.2–20 (11)	14–58 (11)	2.2–3.2 (11)	160–230 (11)	4.6–270 (11)	4.5–22 (11)	0.18–0.45 (4)
	(June 1997 through February 1998)	92	15	32	2.8	180	160	11	--
	Recharge	66–100 (10)	7.9–16 (10)	20–50 (10)	2.7–4.9 (10)	140–210 (10)	34–200 (10)	10–78 (10)	0.22–0.63 (4)
	(April 1998 through July 1998)	92	12	34	2.8	180	110	22	--
Sedgwick recharge site—deep monitoring wells									
	Baseline	62–76 (9)	13–16 (9)	83–93 (9)	3.0–3.7 (9)	300–330 (9)	50–63 (9)	37–65 (9)	0.25–0.34 (5)
	(June 1997 through April 1998)	72	14	86	3.3	330	54	63	--
	Recharge	50–75 (8)	10–15 (8)	75–86 (8)	2.8–3.5 (8)	280–340 (8)	46–55 (8)	21–63 (8)	0.28–0.38 (4)
	(April 1998 through July 1998)	64	12	81	3.1	320	52	46	--

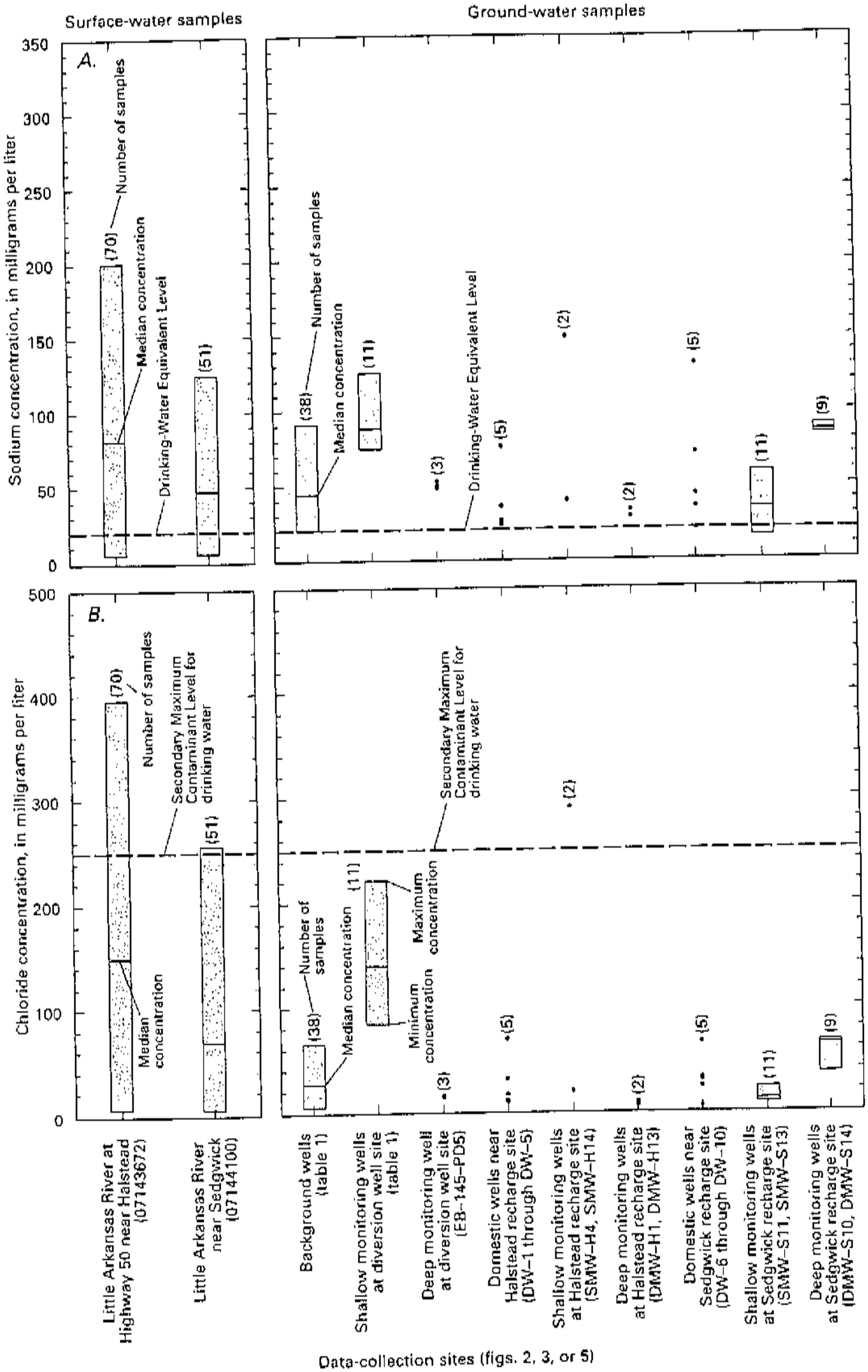


Figure 7. Ranges in concentrations of *(A)* sodium and *(B)* chloride in surface- and ground-water samples during baseline water-quality monitoring. Drinking-Water Equivalent Level and Secondary Maximum Contaminant Levels from U.S. Environmental Protection Agency (1999).

water is closely related to the concentration of chloride. Chloride concentrations in surface-water samples ranged from 8.0 to 400 mg/L; in water from ground-water wells, the range was less than 5.0 to 290 mg/L (fig. 7). In both cases, the range of concentrations is larger in the surface-water samples than in the ground-water samples. Sources for sodium and chloride in the Little Arkansas River may be related to past oil and gas activities near McPherson and Burrton or from wastewater-treatment and industrial discharges from McPherson and Newton (Donald Whittemore, Kansas Geological Survey, oral commun., 1999). Additional sources include seepage from ground water affected by the dissolution of marine sediment, concentration by irrigation, and seepage from sewage lagoons, which tend to be enriched in sodium and chloride (Kemmer, 1979).

Because baseline monitoring was limited to one sample from each of the four monitoring wells, seasonal variation of sodium and chloride concentrations could not be documented at the Halstead recharge site. However, water samples collected in April 1998 from domestic wells near Halstead were used to help define the baseline conditions in that part of the study area.

Nutrients

Nutrients, including species of nitrogen and phosphorus, are required for the growth and reproduction of plants. Agricultural activities, sewage-treatment plants, and domestic sewage lagoons are sources of nutrients in surface and ground water. Large nutrient concentrations in drinking water may have undesirable health effects in humans. For example, nitrate concentrations greater than 10 mg/L as nitrogen in drinking water can cause methemoglobinemia in infants 6 months and younger (U.S. Environmental Protection Agency, 1986); consequently, KDHE has set the MCL for nitrite plus nitrate at 10 mg/L (Kansas Department of Health and Environment, 1994). No other nutrient has regulatory criteria for drinking water.

The nitrite plus nitrate as nitrogen (referred to as nitrite plus nitrate in the remainder of this report) concentration in surface water during baseline water-quality monitoring ranged from less than 0.02 to 3.0 mg/L (table 6, fig. 8). In water from wells, the range was less than 0.01 to 15 mg/L (table 6, fig. 8). The larger concentrations occurred in water from shallow wells at the Halstead and Sedgwick recharge sites. The disposal of sewage on the land surface can cause nitrate contamination in ground water (Freeze and Cherry, 1979), and there is a sewage lagoon east of the Halstead recharge site (fig. 3). Although ground-water flow is generally to the east, mounding beneath the sewage lagoon could result in local radial flow that could affect the water in the monitoring wells at the Halstead recharge site. At the Sedgwick site, fertilizer application on nearby fields may have an effect on nitrate concentrations in shallow wells because a significant portion of the nitrogen applied to crops like corn may not be used by the plants and the nitrogen may percolate to the ground water (Hammer, 1986). None of the water samples from deep wells had nitrite plus nitrate concentrations larger than 3.0 mg/L; in fact, nitrite plus nitrate was not detectable in most samples from deep wells.

Trace Elements

Trace elements refer to solutes in natural water that nearly always occur in concentrations less than 1.0 mg/L (Hem, 1992, p. 129). Of particular concern in this study are the concentrations of iron and manganese. Iron and manganese can precipitate and cause plugging of plumbing and stain laundry. In addition, the tendency of these trace elements to form a precipitate could affect the recharge process by plugging aquifer materials or equipment. During baseline water-quality monitoring, iron was detected at concentrations larger than the USEPA SMCL of 300 µg/L in water samples from both surface-water monitoring sites (table 7). In fact, iron occurred at concentrations as large as 860 µg/L in samples from the Halstead surface-water site and is associated with suspended sediment (Ziegler and others, 1997). Iron was detected in water from all wells except the deep monitoring wells at the Sedgwick recharge site. The range of iron concentrations in water from wells was less than 5.0 to 17,000 µg/L. The largest concentration occurred in water from domestic well DW–10 near Sedgwick.

Manganese has an SMCL of 50 µg/L and is chemically similar to iron. Manganese also may precipitate and interfere with recharge. Manganese concentrations in surface-water samples ranged from less than 5.0 to 1,100 µg/L and in ground-water samples from less than 5.0 to 4,300 µg/L. Larger concentrations of both iron and manganese probably are associated with more chemically reducing conditions (small dissolved-oxygen concentrations) in ground water, especially in

Table 6. Summary of nutrient concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCL, Maximum Contaminant Level; mg/L, milligrams per liter; <, less than; *, median value includes estimated values of one-half the reporting limit for less than (<) values. --, not determined. The first set of numbers under the nitrite plus nitrate column heading (for example, <0.02–2.1) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 0.48) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Nitrite plus nitrate as nitrogen (mg/L)
	U.S. Environmental Protection Agency (1999) drinking-water criteria	
		10 (MCL)
	Surface-water monitoring sites (Little Arkansas River)	
07143672	Baseline	<0.02–2.1 (55)
(near Halstead)	(February 1995 through September 1997)	0.48*
	Recharge	<0.02–2.4 (15)
	(October 1997 through July 1998)	1.2*
07144100	Baseline	0.01–3.0 (51)
(near Sedgwick)	(February 1995 through September 1997)	0.93
	Recharge	0.02–2.9 (16)
	(October 1997 through July 1998)	1.2
	Ground-water or diverted-water monitoring sites	
Background wells		
	Baseline	<0.01–2.6 (38)
	(February 1995 through September 1997)	0.01*
	Recharge	<0.02–0.11 (7)
	(October 1997 through July 1998)	--
Domestic wells near Halstead		
	Baseline	<0.02–8.0 (5)
	(April 1998)	--
Domestic wells near Sedgwick		
	Baseline	0.70–15 (5)
	(August 1998)	--
Halstead diversion well site—diversion well		
	Recharge	<0.02–0.78 (58)
	(April 1996 through July 1998—only sampled during recharge activities)	--
Halstead diversion well site—shallow monitoring well		
	Baseline	<0.02–0.04 (11)
	(February 1995 through March 1996)	--
	Recharge	0.01–0.03 (78)
	(April 1996 through July 1998)	0.01*
Halstead diversion well site—deep monitoring wells		
	Baseline	<0.02 (3)
	(February 1995 through March 1996)	--
	Recharge	0.01–0.03 (24)
	(April 1996 through July 1998)	--
Halstead recharge site—shallow monitoring wells		
	Baseline	1.7, 7.0 (2)
	(May 13, 1997)	--
	Recharge	0.01–9.1 (31)
	(May 29, 1997 through July 1998)	1.1

Table 6. Summary of nutrient concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Nitrite plus nitrate as nitrogen (mg/L)
Ground-water or diverted-water monitoring sites—Continued		
Halstead recharge site—deep monitoring wells		
	Baseline	<0.02 (2)
	(May 13, 1997)	--
	Recharge	<0.02–0.02 (29)
	(May 29, 1997 through July 1998)	--
Sedgwick recharge site—treated diverted source water		
	Recharge	0.13–1.8 (16)
	(April 1998 through July 1998—only sampled during recharge activities)	0.82
Sedgwick recharge site—shallow monitoring wells		
	Baseline	1.8–13 (11)
	(June 1997 through February 1998)	8.9
	Recharge	1.2–15 (10)
	(April 1998 through July 1998)	8.3
Sedgwick recharge site—deep monitoring wells		
	Baseline	0.05–2.7 (9)
	(June 1997 through April 1998)	0.09
	Recharge	0.16–4.6 (8)
	(April 1998 through July 1998)	2.0

water from the background wells and the Halstead diversion well site (fig. 9).

Other trace elements, such as arsenic and selenium, which may have health concerns, were detected occasionally. The largest arsenic concentration was 24 µg/L detected in water from shallow monitoring well EB–145–A3 at the Halstead diversion well site; this concentration is 48 percent of the arsenic MCL. The largest selenium concentration was 11 µg/L in water from shallow monitoring well SMW–S13 at the Sedgwick recharge site; this concentration is 22 percent of the selenium MCL. These concentrations of arsenic and selenium are probably naturally occurring but may concentrate in irrigation drainage in some areas.

Total Organic Carbon

Total organic carbon (TOC) is an approximate determination of the total concentration of organic material in an unfiltered water sample (Drever, 1982). The largest TOC concentration during baseline sampling was 27 mg/L in a sample from the Halstead surface-water monitoring site (table 8). Organic carbon concentrations in ground water are generally smaller than those in surface water (Hem, 1992). The largest TOC concentration in a ground-water sample was 8.2 mg/L from well SMW–S13 at the Sedgwick recharge site. Large TOC concentrations can form trihalo-methanes (THMs) when combined with chlorine during water-treatment processes. THMs are suspected cancer-causing agents (Pine and others, 1996). TOC also may be a concern for the recharge demonstration project because large concentrations in surface water may interfere with treatment of the water for the removal of atrazine by competing for adsorption sites on the powdered activated carbon (PAC). There is no MCL for TOC.

Bacteria

The presence of total coliform bacteria in water is not directly harmful to humans, but in large numbers it may indicate the presence of other species that are pathogenic (Hem, 1992). The largest density of total coliform bacteria detected during baseline water-quality monitoring was 9,000,000 col/100 mL (colonies per 100 milliliters of water) in a sample from the

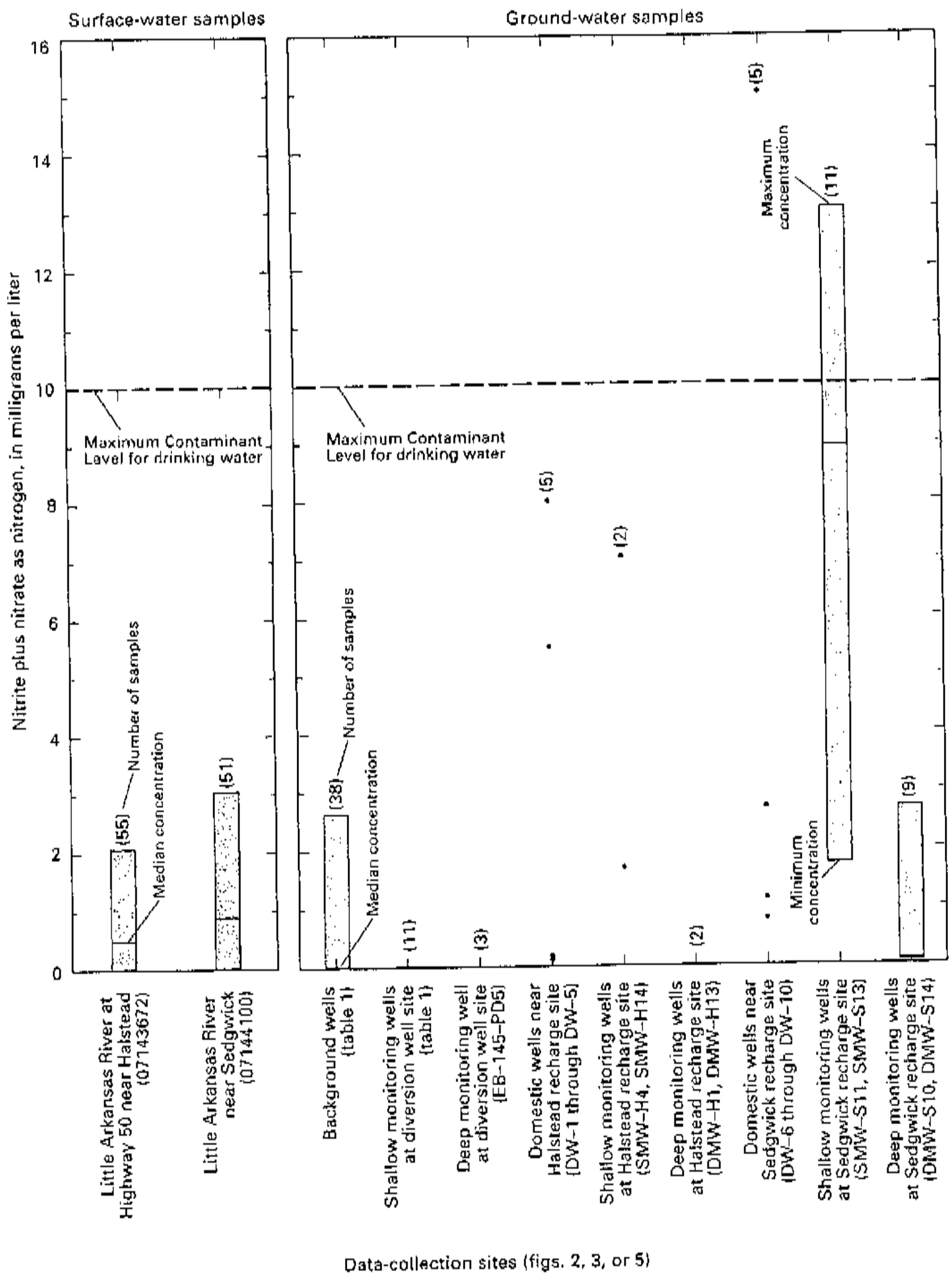


Figure 8. Ranges in concentrations of nitrite plus nitrate in surface- and ground-water samples during baseline water-quality monitoring. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

Sedgwick surface-water monitoring site (table 9). The largest densities were associated with large discharges. Large densities in surface water may be the result of municipal wastewater discharge or runoff from livestock-producing areas. The MCL goal for total coliform bacteria is 0 col/100 mL in finished drinking water. Baseline coliform densities in water from most wells in the study area ranged from less than 1 to 64 col/100 mL. Water from the background wells, however, had densities as large as 490 col/100 mL, possibly due to their immediate proximity to the Little Arkansas River (fig. 1). Results from baseline

Table 7. Summary of trace-element concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCL, Maximum Contaminant Level; SMCL, Secondary Maximum Contaminant level; µg/L, micrograms per liter; <, less than; *, median value is estimated by using a log-probability regression to predict values of the data less than the detection limit; --, not determined. The first set of numbers under each heading (for example, <8.0–460) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 8.4) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Aluminum (µg/L)	Antimony (µg/L)	Arsenic (µg/L)	Barium (µg/L)	Beryllium (µg/L)	Boron (µg/L)	Iron (µg/L)	Lead (µg/L)	Mercury (µg/L)	Manganese (µg/L)	Selenium (µg/L)	Silver (µg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria													
		50 (SMCL)	6.0 (MCL)	50 (MCL)	2,000 (MCL)	4.0 (MCL)	600 (HAL)	300 (SMCL)	15 (MCL)	2.0 (MCL)	50 (SMCL)	50 (MCL)	100 (SMCL)
Surface-water monitoring sites (Little Arkansas River)													
07143672 (near Halstead)	Baseline (February 1995 through September 1997)	<8.0–960 (19) 8.4*	<2.5–<5.0 (18) --	2.0–13 (18) 6.0	50–320 (19) 210	<0.001–2.6 (19) --	28–130 (19) 70	<5.0–860 (70) 13*	<1.0 (18) --	<0.02–0.10 (19) 0.02*	<5.0–1,100 (84) 120*	<2.0–2.3 (18) --	<7.0–23 (19) --
	Recharge (October 1997 through July 1998)	<10 (2) --	<2.5 (2) --	5.2, 6.1 (2) --	210, 213 (2)	<1.0 (2) --	72, 81 (2) --	<5.0–30 (33) 5.23	<1.0 (2) --	<0.10, 0.14 (2) --	<5.0–340 (33) 100*	2.1, 2.3 (2) --	11, 12 (2) --
07144100 (near Sedgwick)	Baseline (February 1995 through September 1997)	<8.0–840 (17) 20*	<2.5 (16) --	3.3–13 (16) 6.4	62–270 (17) 170	<1.0–1.8 (17) --	30–150 (17) 86	<5.0–620 (51) 19*	<1.0–1.1 (16) --	<0.02–0.45 (16) 0.02*	<5.0–740 (51) 58*	<2.0–2.6 (16) --	<7.0–28 (17) --
	Recharge (October 1997 through July 1998)	<10 (3) --	<2.5 (3) --	3.0–7.4 (3) --	66–220 (3) --	<1.0–1.0 (3) --	39–75 (3) --	<5.0–30 (22) 6.8*	<1.0 (3) --	<0.10 (3) --	<5.0–160 (22) 14*	<2.0 (3) --	11–18 (3) --
Ground-water or diverted-water monitoring sites													
<u>Background wells</u>													
	Baseline (February 1995 through September 1997)	<8.0–11 (26) --	<2.5 (26) --	<1.0–15 (26) 2.4*	51–380 (26) 130	<1.0 (26) --	24–57 (26) 47	<5.0–1,800 (38) 550	<1.0–9.4 (26) --	<0.02–<0.20 (26) --	87–4,300 (38) 170	<1.0–2.0 (26) --	<1.0–<10 (26) --
	Recharge (October 1997 through July 1998)	<10 (2) --	<2.5 (2) --	1.0, 6.8 (2) --	120, 170 (2) --	<1.0 (2) --	50 (1) --	<10–660 (7) 610*	<1.0 (2) --	<0.02–<0.10 (2) --	84–620 (7) 97	<2.0 (2) --	<10 (2) --
<u>Domestic wells near Halstead</u>													
	Baseline (April 1998)	--	--	--	--	--	--	<5.0–1,600 (5) --	--	--	<5.0–450 (5) --	--	

,e 7. Summary of trace-element concentrations in water samples ,ected during baseline, 1995–98, and artificial recharge, 1996–98 conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Aluminum (µg/L)	Antimony (µg/L)	Arsenic (µg/L)	Barium (µg/L)	Beryllium (µg/L)	Boron (µg/L)	Iron (µg/L)	Lead (µg/L)	Mercury (µg/L)	Manganese (µg/L)	Selenium (µg/L)	Silver (µg/L)
Ground-water or diverted-water monitoring sites—Continued													
Domestic wells near Sedgwick													
	Baseline (August 1998)	--	--	--	--	--	--	<5.0–17,000 (5) --	--	--	<5.0–180 (5) --	--	--
Halstead diversion well site—diversion well													
	Recharge (April 1996 through July 1998—only sampled during recharge activities)	<10 (17) --	<2.5 (17) --	18–24 (17) 21	83–150 (17) 130	<1.0–2.0 (17) --	50–56 (17) 53	170–1,200 (98) 380	<1.0 (17) --	<0.02–<0.10 (17) --	450–810 (98) 680	<2.0 (17) --	<10–14 (17) --
Halstead diversion well site—shallow monitoring wells													
	Baseline (February 1995 through March 1996)	<10 (9) --	<2.5 (9) --	<1.0–24 (9) 1.4*	300–690 (9) 465	<1.0 (9) --	49–120 (9) 71	260–4,200 (11) 2,630	<1.0–3.4 (9) --	<0.02–0.03 (9) --	980–2,400 (11) 1,600	<2.0 (9) --	<7.0–<10 (9) --
	Recharge (April 1996 through July 1998)	<10–14 (19) --	<2.5 (19) --	<1.0–31 (19) 1.1*	190–840 (19) 280	<1.0 (19) --	39–93 (19) 59	<10–4,900 (80) 1,400	<1.0 (19) --	<0.02–<0.10 (19) --	<5.0–3,400 (80) 1,100	<2.0 (19) --	<10 (19) --
Halstead diversion well site—deep monitoring well													
	Baseline (February 1995 through March 1996)	<10 (2) --	<2.5 (2) --	21, 22 (2) --	74, 75 (2) --	<1.0 (2) --	53, 56 (2) --	<10–13 (3) --	<1.0 (3) --	<0.02 (2) --	417–481 (3) --	<2.0 (2) --	<10 (2) --
	Recharge (April 1996 through July 1998)	<10 (5) --	<2.5 (5) --	21–28 (5) --	68–110 (5) --	<1.0 (5) --	47–51 (5) --	59–310 (24) 130	<1.0 (5) --	<0.02–0.10 (5) --	380–670 (24) 418	<2.0 (5) --	<10 (5) --
Halstead recharge site—shallow monitoring wells													
	Baseline (May 13, 1997)	<10 (2) --	<2.5 (2) --	1.0, 1.7 (2) --	160, 490 (2) --	<1.0 (2) --	54, 98 (2) --	<5.0, 11 (2) --	<1.0 (2) --	0.08, 0.12 (2) --	20, 68 (2) --	2.0, 2.8 (2) --	<10 (2) --
	Recharge (May 29, 1997 through July 1998)	<10 (17) --	<2.5 (17) --	<1.0–4.1 (17) 2.5*	220–310 (17) 290	<1.0–1.5 (17) --	41–67 (17) 53	<5.0–6.6 (31) --	<1.0 (17) --	<0.02–0.05 (17) --	<5.0–15 (31) 1.1*	<2.0–3.6 (17) --	<10–21 (17) --

Table 7. Summary of trace-element concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Aluminum (µg/L)	Antimony (µg/L)	Arsenic (µg/L)	Barium (µg/L)	Beryllium (µg/L)	Boron (µg/L)	Iron (µg/L)	Lead (µg/L)	Mercury (µg/L)	Manganese (µg/L)	Selenium (µg/L)	Silver (µg/L)
Ground-water or diverted-water monitoring sites—Continued													
Halstead recharge site—deep monitoring wells													
	Baseline (May 13, 1997)	<10 (2) --	<2.5 (2) --	8.3, 11 (2) --	82, 150 (2) --	<1.0 (2) --	37, 43 (2) --	7.9, 300 (2) --	<1.0 (2) --	0.12, 0.13 (2) --	210, 260 (2) --	<2.0 (2) --	<10 (2) --
	Recharge (May 29, 1997 through July 1998)	<10 (16) --	<2.5 (16) --	8.1–22 (16) 11	76–390 (16) 150	<1.0–1.6 (16) --	35–56 (16) 51	17–1,300 (29) 180	<1.0 (16) --	<0.02–<0.10 (16) --	230–750 (29) 560	<2.0 (16) --	<10–23 (16) --
Sedgwick recharge site—treated diverted source water													
	Recharge (April 1998 through July 1998—only sampled during recharge activities)	<10–23 (3) --	<2.5 (3) --	3.8–5.9 (3) --	82–210 (3) --	<1.0 (3) --	45–71 (3) --	<5.0 (16) --	<1.0 (3) --	<0.10 (3) --	<5.0–20 (16) 4.4*	<2.0 (3) --	<10–18 (3) --
Sedgwick recharge site—shallow monitoring wells													
	Baseline (June 1997 through February 1998)	<10 (4) --	<2.5 (4) --	<1.0–2.7 (4) --	71–110 (4) --	<1.0 (4) --	35–48 (4) --	<5.0–1,400 (11) --	<1.0 (4) --	<0.02–0.07 (4) --	<5.0–180 (11) 26*	3.1–11 (4) --	<10–10 (4) --
	Recharge (April 1998 through July 1998)	<10 (4) --	<2.5 (4) --	<1.0–1.4 (4) --	74–130 (4) --	<1.0 (4) --	23–62 (4) --	<5.0–68 (10) --	<1.0 (4) --	<0.02–<0.10 (4) --	<5.0–16 (10) 6.4*	<2.0–7.9 (4) --	<10–10.4 (2) --
Sedgwick recharge site—deep monitoring wells													
	Baseline (June 1997 through April 1998)	<10 (5) --	<2.5 (5) --	3.2–4.2 (5) --	38–66 (5) --	<1.0 (5) --	49–55 (5) --	<10 (9) --	<1.0 (5) --	<0.02–<0.10 (5) --	130–250 (9) 230	<2.0 (5) --	<10 (5) --
	Recharge (April 1998 through July 1998)	<10–11.1 (4) --	<2.5 (4) --	2.3–4.5 (4) --	33–60 (4) --	<1.0–4.2 (4) --	33–54 (4) --	<10 (8) --	<1.0 (4) --	<0.02 (4) --	140–250 (8) 200	<2.0 (4) --	<10–17 (4) --

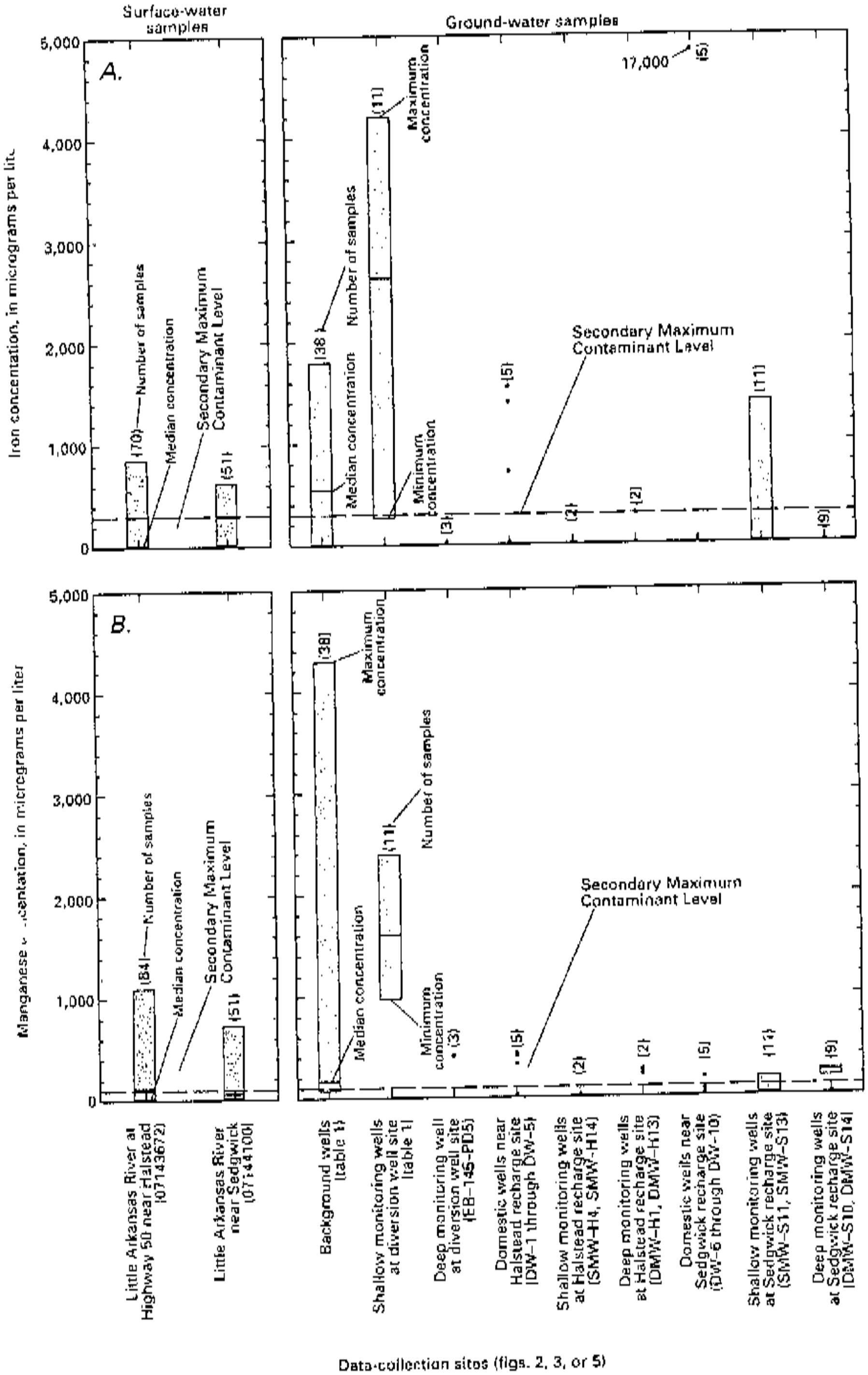


Figure 9. Ranges in *(A)* iron and *(B)* manganese concentrations in surface- and ground-water samples during baseline water-quality monitoring. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

monitoring indicated that total coliform bacteria would affect ground-water recharge only if treatment of the source water did not remove bacteria.

Organic Compounds

Of the 173 organic compounds analyzed (including dissolved pesticides, total recoverable organochlorine and organophosphate pesticides, total recoverable volatile organic compounds, acid compounds, and base/neutral compounds), 32 different pesticide parent compounds or metabolites were detected and are listed in table 10. Of these 32 organic compounds, 28 were detected in surface-water samples, and 16 were detected in ground-water samples collected during baseline water-quality monitoring. Of the organic compounds in surface-water samples, concentrations of alachlor, atrazine, cyanazine, metolachlor, and propazine exceeded 20 percent of their respective MCLs (or HALs) in at least one sample (table 10).

Alachlor is an herbicide commonly used on corn, grain sorghum, and soybeans. Alachlor has an MCL of 2.0 µg/L. Concentrations in excess of the MCL occurred in samples from both surface-water monitoring sites during baseline water-quality monitoring, with the largest concentration of 7.4 µg/L occurring at the Halstead surface-water monitoring site (table 10). Alachlor was detected in some ground-water samples as well but did not exceed the MCL.

Atrazine, a herbicide used on corn and grain sorghum, has an MCL of 3.0 µg/L as an annual mean (Kansas Department of Health and Environment, 1994). The largest concentrations of atrazine detected during baseline water-quality monitoring at the Halstead and Sedgwick surface-water monitoring sites were 46 and 34 µg/L (by ELISA analysis), respectively (fig. 10). However, neither

Table 8. Summary of total organic carbon concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[mg/L, milligrams per liter; --, not determined. The first set of numbers under each heading (for example, 3.8–27) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 7.2) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Total organic carbon (mg/L)	Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Total organic carbon (mg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria		--	**Ground-water or diverted-water monitoring sites—Continued**		
Surface-water monitoring sites (Little Arkansas River)			**Halstead diversion well site—deep monitoring well**		
07143672	Baseline	3.8–27 (19)		Baseline	0.86, 1.0 (2)
(near Halstead)	(February 1995 through September 1997)	7.2		(February 1995 through March 1996)	--
	Recharge	6.6, 6.7 (2)		Recharge	0.82–1.2 (5)
	(October 1997 through July 1998)	--		(April 1996 through July 1998)	--
07144100	Baseline	0.59–16 (16)			
(near Sedgwick)	(February 1995 through September 1997)	6.4			
	Recharge	5.7–7.3 (3)	**Halstead recharge site—shallow monitoring wells**		
	(October 1997 through July 1998)	--		Baseline	0.90, 1.0 (2)
Ground-water or diverted-water monitoring sites				May 13, 1997	--
Background wells				Recharge	0.85–1.2 (17)
	Baseline	0.45–1.9 (26)		(May 29, 1997 through July 1998)	0.54
	(February 1995 through September 1997)	0.75	**Halstead recharge site—deep monitoring wells**		
	Recharge	0.68, 0.75 (2)		Baseline	0.59, 0.73 (2)
	(October 1997 through July 1998)	--		(May 13, 1997)	--
Domestic wells near Halstead				Recharge	0.42–1.4 (16)
	Baseline	--		May 29, 1997 through July 1998)	0.54
	(April 1998)	--	**Sedgwick recharge site—treated diverted source water**		
Domestic wells near Sedgwick				Recharge	4.3–5.9 (3)
	Baseline	--		(April 1998 through July 1998—only sampled during recharge activities)	--
	(August 1998)	--	**Sedgwick recharge site—shallow monitoring wells**		
Halstead diversion well site—diversion well				Baseline	0.52–8.2 (4)
	Recharge	0.97–1.6 (17)		(June 1997 through February 1998)	--
	(April 1996 through July 1998—only sampled during recharge activities)	1.2		Recharge	0.58–2.2 (4)
Halstead diversion well site—shallow monitoring wells				(April 1998 through July 1998)	--
	Baseline	2.4–5.2 (9)	**Sedgwick recharge site—deep monitoring wells**		
	(February 1995 through March 1996)	3.8		Baseline	0.41–0.50 (5)
	Recharge	0.8–4.3		(June 1997 through April 1998)	--
	(April 1996 through July 1998)	3.3		Recharge	0.36–0.46 (4)
				(April 1998 through July 1998)	--

monitoring site on the Little Arkansas River had an annual mean atrazine concentration greater than the MCL from February 1995 through September 1997. A previous study (Christensen and Ziegler, 1998a) reported that 90 percent of the annual runoff load of atrazine in the Little Arkansas River generally occurred during a short period of time from May through July, indicating that atrazine is of primary concern during spring and early summer. Atrazine was detected in water from nearly all wells, although at smaller concentrations than in surface water, and concentrations were all less than the MCL.

Cyanazine has an HAL of 1.0 μg/L, and in the study area it is used most often to control weeds in the production of corn. Cyanazine was detected in water from both surface-water monitoring sites during baseline water-quality sampling, with a range in concentrations from less than 0.004 to 0.53 μg/L in water

Table 9. Summary of total coliform bacteria densities in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCLG, Maximum Contaminant Level Goal; col/100 mL, colonies per 100 milliliters of water; <, less than; *, median value is estimated using a log-probability regression to predict values of the data less than the detection limit; --, not determined. The first set of numbers under each heading (for example, –2,000,000) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 360) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Total coliform bacteria (col/100mL)
U.S. Environmental Protection Agency (1999) drinking-water criteria		
		0 (MCLG)
Surface-water monitoring sites (Little Arkansas River)		
07143672	Baseline	<1–2,000,000 (55)
(near Halstead)	(February 1995 through September 1997)	360*
	Recharge	110–53,000 (15)
	(October 1997 through July 1998)	2,200
07144100	Baseline	<4-9,000,000 (50)
(near Sedgwick)	(February 1995 through September 1997)	280*
	Recharge	30–27,000 (16)
	(October 1997 through July 1998)	2,700
Ground-water or diverted-water monitoring sites		
Background wells		
	Baseline	<1–490 (38)
	(February 1995 through September 1997)	0.10*
	Recharge	<1–10 (7)
	(October 1997 through July 1998)	--
Domestic wells near Halstead		
	Baseline	<1–2 (5)
	(April 1998)	--
Domestic wells near Sedgwick		
	Baseline	<1–64 (5)
	(August 1998)	--
Halstead diversion well site—diversion well		
	Recharge	<1–<100 (59)
	(April 1996 through July 1998—only sampled during recharge activities)	0.24*
Halstead diversion well site—shallow monitoring wells		
	Baseline	<2–14 (11)
	(February 1995 through March 1996)	--
	Recharge	<1–7 (80)
	(April 1996 through July 1998)	0.03*
Ground-water or diverted-water monitoring sites—Continued		
Halstead diversion well site—deep monitoring well		
	Baseline	<2–<10 (3)
	(February 1995 through March 1996)	--
	Recharge	<1–9 (24)
	(April 1996 through July 1998)	--
Halstead recharge site—shallow monitoring wells		
	Baseline	<1, 1 (2)
	(May 13, 1997)	--
	Recharge	<1–4 (31)
	(May 29, 1997 through July 1998)	0.20*
Halstead recharge site—deep monitoring wells		
	Baseline	<1, 2 (2)
	(May 13, 1997)	--
	Recharge	<1–80 (30)
	(May 29, 1997 through July 1998)	--
Sedgwick recharge site—treated diverted source water		
	Recharge	<1–400 (16)
	(April 1998 through July 1998—only sampled during recharge activities)	82*
Sedgwick recharge site—shallow monitoring wells		
	Baseline	<1–<10 (11)
	(June 1997 through February 1998)	--
	Recharge	<1–1 (4)
	(April 1998 through July 1998)	--
Sedgwick recharge site—deep monitoring wells		
	Baseline	<1–3 (9)
	(June 1997 through April 1998)	--
	Recharge	<1 (8)
	(April 1998 through July 1998)	--

from the Halstead surface-water site and from less than 0.004 to 2.1 μg/L in water from the Sedgwick surface-water site. Cyanazine concentrations were less than 0.10 μg/L in samples from all ground-water monitoring sites.

Metolachlor has an HAL of 70 μg/L and is used to control weeds in the production of corn, grain sorghum, and soybeans. The largest concentration of metolachlor detected during baseline water-quality monitoring was 45 μg/L in water from the Halstead

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions

[MCL, Maximum Contaminant Level; HAL, Health Advisory Level; GC/MS, gas chromatography/mass spectrometry; ELISA, enzyme-linked immunosorbent assay; µg/L, micrograms per liter; <, less than; *, median value is estimated using a log-probability regression to predict values of data less than the detection limit; --, not determined; e, estimated. The first set of numbers under each heading (for example, <0.05–1.1) indicates the range in concentrations. The number in parentheses () indicates the number of samples collected, and the number below (for example, 0.14) is the median value]

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	2,4-D (µg/L)	2,6-Diethyl-analine (µg/L)	Acetochlor (µg/L)	Alachlor (µg/L)	Ametryne (µg/L)	Atrazine (determined by GC/MS) (µg/L)	Atrazine (determined by ELISA) (µg/L)	Benfluralin (µg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria									
		70 (MCL)	--	--	2.0 (MCL)	60 (HAL)	3.0 (MCL)	--	--
Surface-water monitoring sites (Little Arkansas River)									
07143672 (near Halstead)	Baseline (February 1995 through September 1997)	<0.05–1.1 (5) --	<0.006 (18) --	<0.009–2.2 (67) --	<0.009–7.4 (67) 0.14*	<0.05–0.09 (49) 0.05*	0.03–46 (67) 3.5	<0.10–46 (577) 1.08*	<0.002–<0.013 (18) --
	Recharge (October 1997 through July 1998)	--	<0.003 (2) --	<0.002–0.15 (12) --	<0.005–1.5 (12) 0.054*	<0.05 (10) --	0.11–17 (12) 2.8	<0.10–12 (34) 0.20*	<0.002 (2) --
07144100 (near Sedgwick)	Baseline (February 1995 through September 1997)	<0.035–0.34 (7) --	<0.003–<0.006 (16) --	<0.002–0.46 (155) 0.009*	<0.009–6.1 (155) 0.28*	<0.05 (139) --	0.04–27 (155) 3.9	<0.10–34 (877) 1.9	<0.002–<0.013 (16) --
	Recharge (October 1997 through July 1998)	0.15 (1) --	<0.003 (3) --	<0.002–0.34 (36) 0.005*	<0.05–3.9 (36) 0.35*	<0.05–0.92 (33) --	0.13–22 (36) 5.5	<0.10–27 (214) 3.8	<0.002 (3) --
Ground-water or diverted-water monitoring sites									
Background wells									
	Baseline (February 1995 through September 1997)	<0.035–<0.05 (14) --	<0.003–<0.006 (26) --	<0.002–<0.05 (27) --	<0.002–<0.05 (27) --	<0.05 (1) --	<0.001–1.6 (27) 0.003*	<0.1–2.0 (47) --	<0.002–<0.013 (26) --
	Recharge (October 1997 through July 1998)	--	<0.003 (2) --	<0.002–<0.05 (3) --	<0.002–<0.05 (3) --	<0.05 (1) --	<0.001–<0.05 (3) --	<0.1 (7) --	<0.002 (2) --
Domestic wells near Halstead									
	Baseline (April 1998)	--	--	<0.05 (2) --	<0.05 (2) --	<0.05 (2) --	<0.05–0.10 (2) --	<0.1 (5) --	--
Domestic wells near Sedgwick									
	Baseline (August 1998)	--	--	<0.05 (5) --	<0.05 (5) --	<0.05 (5) --	<0.05–0.22 (5) --	<0.05–1.1 (5) --	--

Table 10. Summary of organic compound concentrations in wa[illegible] samples collected during baseline, 1995–98, and artificial rechar[illegible] 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	2,4-D (µg/L)	2,6-Diethyl-analine (µg/L)	Acetochlor (µg/L)	Alachlor (µg/L)	Ametryne (µg/L)	Atrazine (determined by GC/MS) (µg/L)	Atrazine (determined by ELISA) (µg/L)	Benfluralin (µg/L)
Ground-water or diverted-water monitoring sites—Continued									
Halstead diversion well site—diversion well									
	Recharge (April 1996 through July 1998—only sampled during recharge activities)	<0.035–<0.5 (6) --	e 0.0017–e 0.0034 (17) 0.002*	<0.002–<0.05 (77) --	< 0.002–<0.05 (77) --	<0.05 (60) --	<0.006–0.09 (77) 0.070*	<0.1–0.21 (231) 0.064*	<0.002 (17) --
Halstead diversion well site—shallow monitoring wells									
	Baseline (February 1995 through March 1996)	<0.035–<0.05 (6) --	<0.005–0.009 (9) 0.007*	<0.002–<0.05 (14) --	<0.002–e 0.24 (14) --	<0.05 (6) --	e 0.008–0.48 (14) 0.043*	<0.1–1.03 (13) 0.024*	<0.002–<0.013 (9) --
	Recharge (April 1996 through July 1998)	--	e 0.012–0.015 (18) 0.006*	<0.002–<0.05 (59) --	<0.002–<0.05 (59) --	<0.05 (41) --	<0.001–2.2 (59) 0.017*	<0.1–2.6 (82) 0.12*	<0.002 (18) --
Halstead diversion well site—deep monitoring well									
	Baseline (February 1995 through March 1996)	<0.035 (2) --	<0.003 (2) --	<0.002 (2) --	<0.002 (2) --	--	<0.001, 0.004 (2) --	<0.1 (3) --	<0.002 (2) --
	Recharge (April 1996 through July 1998)	--	e 0.0017–0.0053 (5) --	<0.002–<0.05 (14) --	<0.002–<0.05 (14) --	<0.05 (9) --	0.042–0.083 (14) 0.059*	<0.1–0.14 (23) --	<0.002 (5) --
Halstead recharge site—shallow monitoring wells									
	Baseline (May 13, 1997)	<0.035 (2) --	<0.003 (2) --	<0.002 (2) --	<0.002 (2) --	--	0.026, 0.14 (2) --	<0.1, 0.2 (2) --	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.035–<0.15 (7) --	<0.003 (17) --	<0.002–<0.05 (34) --	<0.002–<0.05 (34) --	<0.05 (17) --	<0.027–0.09 (34) 0.04*	<0.1–0.18 (30) --	<0.002 (17) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	2,4-D (µg/L)	2,6-Diethyl-analine (µg/L)	Acetochlor (µg/L)	Alachlor (µg/L)	Ametryne (µg/L)	Atrazine (determined by GC/MS) (µg/L)	Atrazine (determined by ELISA) (µg/L)	Benfluralin (µg/L)
				Ground-water or diverted-water monitoring sites—Continued					
Halstead recharge site—deep monitoring wells									
	Baseline (May 13, 1997)	<0.035 (2) --	<0.003 (2) --	<0.002 (2) --	<0.002 (2) --	--	<0.001, 0.0099 (2) --	<0.1 (2) --	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.035–<0.15 (7) --	e 0.0029–e 0.0032 (17) --	<0.002–<0.05 (36) --	<0.002–0.05 (36) --	<0.05 (19) --	<0.001–0.084 (36) 0.01*	<0.1–<0.19 (30) --	<0.002 (17)
Sedgwick recharge site—treated diverted source water									
	Recharge (April 1998 through July 1998--only sampled during recharge activities)	<0.05, <0.15 (2) --	<0.003 (3) --	<0.002–<0.05 (24) --	<0.017–0.72 (24) 0.029*	<0.05 (21) --	<0.05–6.7 (24) 0.25*	<0.1–6.8 (62) 0.21*	<0.002 (3) --
Sedgwick recharge site—shallow monitoring wells									
	Baseline (June 1997 through February 1998)	<0.035 (2) --	<0.003 (4) --	<0.002–<0.05 (12) --	<0.002–<0.05 (12) --	<0.05 (8) --	<0.05–0.10 (12) 0.014*	<0.1–0.21 (11) --	<0.002 (4) --
	Recharge (April 1998 through July 1998)	<0.035, <0.15 (2) --	<0.003 (4) --	<0.002–<0.05 (9) --	<0.002–<0.05 (9) --	<0.05 (6) --	<0.016–0.36 (9) 0.057*	<0.1–0.81 (10) --	<0.002 (4) --
Sedgwick recharge site—deep monitoring wells									
	Baseline (June 1997 through April 1998)	<0.035 (3) --	<0.003 (5) --	<0.002–<0.05 (9) --	<0.002–<0.05 (9) --	<0.05 (4) --	<0.001–<0.05 (9) --	<0.1 (9) --	<0.002 (5) --
	Recharge (April 1998 through July 1998)	<0.015 (2) --	<0.003 (4) --	<0.002–<0.05 (6) --	<0.002–<0.05 (6) --	<0.05 (3) --	<0.004–<0.05 (6) --	<0.1 (8) --	<0.002 (4) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Butylate (µg/L)	Carbaryl (µg/L)	Carbofuran (µg/L)	Cyanazine (µg/L)	DCPA (µg/L)	Deethyl-atrazine (µg/L)	Deisopropyl-atrazine (µg/L)	Diazinon (µg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria									
		350 (HAL)	700 (HAL)	40 (MCL)	1.0 (HAL)	--	--	--	0.600 (HAL)
Surface-water monitoring sites (Little Arkansas River)									
07143672 (near Halstead)	Baseline (February 1995 through September 1997	<0.002–<0.008 (18) --	<0.003–0.083 (18) --	<0.003–0.93 (18) 0.017*	<0.004–0.53 (67) 0.026*	<0.002–<0.004 (18) --	0.007–1.22 (67) 0.23	<0.05–0.71 (49) 0.16	<0.008–0.023 (18) 0.003*
	Recharge (October 1997 through July 1998)	<0.002 (2) --	e 0.01, e 0.027 (2) --	<0.002, 0.48 (2) --	<0.004–<0.05 (12) --	0.005, 0.01 (2) --	<0.05–1.2 (12) 0.14*	<0.05–0.80 (10) 0.06*	0.06, 0.08 (2) --
07144100 (near Sedgwick)	Baseline (February 1995 through September 1997)	<0.002–<0.008 (16) --	<0.003–0.089 (16) --	<0.013–0.45 (16) 0.05*	<0.004–2.1 (153) 0.013*	<0.002–0.005 (16) 0.002*	0.008–1.29 (155) 0.37	0.05–0.67 (139) 0.21	<0.002–0.019 (16) 0.006*
	Recharge (October 1997 through July 1998)	<0.002 (3) --	<0.003–e 0.014 (3) --	0.026–0.16 (3) --	<0.05–0.23 (36) --	<0.002–0.002 (3) --	<0.05–1.3 (36) 0.56*	<0.05–0.72 (33) 0.24*	<0.002–0.025 (3) --
Ground-water or diverted-water monitoring sites									
Background wells									
	Baseline (February 1995 through September 1997)	<0.002–<0.008 (26) --	<0.003–0.046 (26) --	<0.003–<0.013 (26) --	<0.004–<0.05 (27) --	<0.002–<0.004 (26) --	<0.002–0.11 (27) --	0.08 (1) --	<0.002–<0.008 (26) --
	Recharge (October 1997 through July 1998)	<0.002 (2) --	<0.002 (2) --	<0.003 (2) --	<0.004–<0.05 (3) --	<0.002 (2) --	<0.002–<0.05 (2) --	<0.05 (1) --	<0.002 (2) --
Domestic wells near Halstead									
	Baseline (April 1998)	--	--	--	<0.05 (2) --	--	<0.05–0.05 (2) --	<0.05 (2) --	--

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Butylate (µg/L)	Carbaryl (µg/L)	Carbofuran (µg/L)	Cyanazine (µg/L)	DCPA (µg/L)	Deethyl-atrazine (µg/L)	Deisopropyl-atrazine (µg/L)	Diazinon (µg/L)
Ground-water or diverted-water monitoring sites—Continued									
Domestic wells near Sedgwick									
	Baseline (August 1998)	--	--	--	<0.05 (5) --	--	<0.05–0.66 (5) --	<0.05–0.11 (5) --	--
Halstead diversion well site—diversion well									
	Recharge (April 1996 through July 1998—only sampled during recharge activities)	<0.002 (17) --	<0.003 (17) --	<0.003 (17) --	<0.004–<0.05 (77) --	<0.002 (17) --	e 0.001–<0.05 (4) --	<0.05 (60) --	<0.002 (17) --
Halstead diversion well site—shallow monitoring wells									
	Baseline (February 1995 through March 1996)	<0.002–<0.008 (9) --	<0.003–<0.046 (9) --	<0.003–e 0.10 (9) --	<0.004–<0.05 (14) --	<0.002–<0.004 (9) --	<0.002–<0.05 (4) --	<0.05 (9) --	<0.002–<0.008 (6) --
	Recharge (April 1996 through July 1998)	e 0.0017–0.004 (18) 0.002*	<0.003 (18) --	<0.003–e 0.0084 (18) --	<0.004–<0.05 (59) --	<0.002 (18) --	<0.002–0.16 (59) --	<0.05 (41) --	<0.002 (18) --
Halstead diversion well site—deep monitoring well									
	Baseline (February 1995 through March 1996)	<0.002 (2) --	<0.003 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002 (1) --	<0.05 (1) --	<0.002 (2) --
	Recharge (April 1996 through July 1998)	<0.002 (5) --	<0.002 (5) --	<0.003–e 0.004 (5) --	<0.004-0.07 (14) --	<0.002 (5) --	e 0.002–<0.05 (12) --	<0.05 (7) --	<0.002 (5) --
Halstead recharge site—shallow monitoring wells									
	Baseline (May 13, 1997)	<0.002 (2) --	<0.003 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	e 0.02, e 0.12 (2) --	--	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.002 (17) --	<0.003 (17) --	<0.003 (17) --	<0.004–<0.05 (34) --	<0.002 (17) --	e 0.003–e 0.08 (24) --	<0.05 (17) --	<0.002 (17) --

Table 10. Summary of organic compound concentrations in wa… samples collected during baseline, 1995–98, and artificial rechar… 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Butylate (µg/L)	Carbaryl (µg/L)	Carbofuran (µg/L)	Cyanazine (µg/L)	DCPA (µg/L)	Deethyl-atrazine (µg/L)	Deisopropyl-atrazine (µg/L)	Diazinon (µg/L)
					Ground-water or diverted-water monitoring sites—Continued				
Halstead recharge site—deep monitoring wells									
	Baseline (May 13, 1997)	<0.002 (2) --	<0.003 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002, e 0.0072 (2) --	--	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.002 (17) --	<0.003 (17) --	<0.003 (17) --	<0.004–<0.05 (36) --	<0.002 (17) --	e 0.0012–<0.05 (36) --	<0.05 (19) --	<0.002 (17) --
Sedgwick recharge site—treated diverted source water									
	Recharge (April 1998 through July 1998—only sampled during recharge activities)	<0.002 (3) --	<0.003 (3) --	<0.003–e 0.023 (3) --	<0.004–<0.05 (24) --	<0.002 (3) --	e 0.0096–0.39 (24) --	<0.05–0.19 (21) --	<0.002 (3) --
Sedgwick recharge site—shallow monitoring wells									
	Baseline (June 1997 through February 1998)	<0.002 (4) --	<0.003 (4) --	<0.003 (4) --	<0.004–<0.05 (12) --	<0.002 (4) --	e 0.011–0.21 (12) --	<0.05–0.05 (8) --	<0.002 (4) --
	Recharge (April 1998 through July 1998)	<0.002 (4) --	<0.003 (4) --	<0.003– e 0.0088 (4) --	<0.004–<0.05 (9) --	<0.002 (4) --	e 0.013–0.07 (9) --	<0.05 (6) --	<0.002 (4) --
Sedgwick recharge site—deep monitoring wells									
	Baseline (June 1997 through April 1998)	<0.002 (5) --	<0.003 (5) --	<0.003 (5) --	<0.004–<0.05 (9) --	<0.002 (5) --	<0.002–<0.05 (9) --	<0.002–0.05 (4) --	<0.002 (5) --
	Recharge (April 1998 through July 1998)	<0.002 (4) --	<0.003 (4) --	<0.003 (4) --	<0.004–<0.05 (6) --	<0.002 (4) --	<0.05 (6) --	<0.05 (3) --	<0.002 (4) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Diuron (µg/L)	EPTC (µg/L)	Ethoprop (µg/L)	Ethal-fluralin (µg/L)	Linuron (µg/L)	Metol-achlor (µg/L)	Metribuzin (µg/L)	Naprop-amide (µg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria									
		10 (HAL)	--	--	--	--	70 (HAL)	100 (HAL)	--
Surface-water monitoring sites (Little Arkansas River)									
07143672 (near Halstead)	Baseline (February 1995 through September 1997	<0.05–0.26 (5) --	<0.002–0.061 (18) --	<0.002–<0.012 (18) --	<0.004–<0.013 (18) --	<0.002–<0.039 (18) --	0.01–45 (67) 1.0	<0.004–0.28 (67) 0.01*	<0.01 (18) --
	Recharge (October 1997 through July 1998)	--	<0.002 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.05–16 (12) 2.5*	<0.05–0.16 (12) 0.04*	<0.003 (2) --
07144100 (near Sedgwick)	Baseline (February 1995 through September 1997)	<0.02–0.28 (7) --	<0.002–0.06 (16) --	<0.003–0.005 (16) --	<0.004–0.007 (18) --	<0.002–0.008 (16) --	0.009–22 (155) 1.2	<0.004–0.44 (155) 0.014*	<0.003–<0.01 (16) --
	Recharge (October 1997 through July 1998)	0.24 (1) --	<0.002–0.005 (3) --	<0.003 (3) --	<0.004 (3) --	<0.002 (3) --	0.08–16 (36) 2.8	<0.05–0.23 (36) 0.02*	<0.003 (3) --
Ground-water or diverted-water monitoring sites									
Background wells									
	Baseline (February 1995 through September 1997)	<0.02–<0.05 (14) --	<0.002–<0.005 (26) --	<0.003–<0.012 (26) --	<0.004–<0.013 (26) --	<0.002–<0.039 (26) --	<0.002–0.77 (27) --	<0.004–<0.05 (27) --	<0.003–<0.01 (26) --
	Recharge (October 1997 through July 1998)	--	<0.002 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002–<0.05 (3) --	<0.004–<0.05 (3) --	<0.003 (2) --
Domestic wells near Halstead									
	Baseline (April 1998)	--	--	--	--	--	<0.05 (2) --	<0.05 (2) --	--

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Diuron (µg/L)	EPTC (µg/L)	Ethoprop (µg/L)	Ethalfluralin (µg/L)	Linuron (µg/L)	Metolachlor (µg/L)	Metribuzin (µg/L)	Napropamide (µg/L)
Ground-water or diverted-water monitoring sites—Continued									
Domestic wells near Sedgwick									
	Baseline (August 1998)	--	--	--	--	--	<0.05 (5) --	<0.05 (5) --	--
Halstead diversion well site—diversion well									
	Recharge (April 1996 through July 1998—only sampled during recharge activities)	<0.02 (6) --	<0.002 (17) --	<0.003 (17) --	<0.004 (17) --	<0.002 (17) --	<0.002–<0.05 (77) 0.007*	<0.004–<0.05 (77) --	<0.003 (17) --
Halstead diversion well site—shallow monitoring wells									
	Baseline (February 1995 through March 1996)	<0.02–<0.05 (8) --	<0.002–<0.005 (9) --	<0.003–<0.012 (9) --	<0.004–<0.013 (9) --	<0.002–<0.039 (9) --	<0.008–e 0.17 (14) 0.011*	<0.004–<0.05 (14) --	<0.003–<0.01 (9) --
	Recharge (April 1996 through July 1998)	--	e 0.0018–e 0.002 (18) --	<0.003 (18) --	<0.004 (18) --	<0.002 (28) --	<0.002–1.0 (59) 0.019*	<0.004–<0.05 (59) --	<0.003 (18) --
Halstead diversion well site—deep monitoring well									
	Baseline (February 1995 through March 1996)	<0.02 (2) --	<0.002 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002 (2) --	<0.004 (2) --	<0.003 (2) --
	Recharge (April 1996 through July 1998)	--	<0.002 (5) --	<0.003 (5) --	<0.004 (5) --	<0.002 (5) --	0.007–<0.05 (14) 0.011*	<0.004–<0.05 (14) --	<0.003 (5) --
Halstead recharge site—shallow monitoring wells									
	Baseline (May 13, 1997)	<0.02 (2) --	<0.002 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002 (2) --	<0.004 (2) --	<0.003 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.02 (7) --	<0.002 (17) --	<0.003 (17) --	<0.004 (17) --	<0.002 (17) --	<0.002–<0.05 (34) 0.016*	<0.004–<0.05 (34) --	<0.003 (17) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Diuron (µg/L)	EPTC (µg/L)	Ethoprop (µg/L)	Ethal-fluralin (µg/L)	Linuron (µg/L)	Metol-achlor (µg/L)	Metribuzin (µg/L)	Naprop-amide (µg/L)
				Ground-water or diverted-water monitoring sites—Continued					
Halstead recharge site—deep monitoring wells									
	Baseline (May 13, 1997)	<0.02 (2) --	<0.002 (2) --	<0.003 (2) --	<0.004 (2) --	<0.002 (2) --	<0.002 (2) --	<0.004 (2) --	<0.003 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.02 (7) --	<0.002 (17) --	<0.003 (17) --	<0.004 (17) --	<0.002 (17) --	<0.002–<0.05 (36) 0.016*	<0.004–<0.05 (36) --	<0.0044–e 0.003 (17) --
Sedgwick recharge site—treated diverted source water									
	Recharge (April 1998 through July 1998—only sampled during recharge activities)	<0.02 (2) --	<0.002 (3) --	<0.003 (3) --	<0.004 (3) --	<0.002 (3) --	<0.05–3.46 (24) 0.14*	<0.004–0.06 (24) --	<0.003 (3) --
Sedgwick recharge site—shallow monitoring wells									
	Baseline (June 1997 through February 1998)	<0.02 (2) --	<0.002–0.014 (4) --	<0.003 (4) --	<0.004 (4) --	<0.002 (4) --	e 0.0029–7.0 (12) 2.5*	<0.004–<0.05 (12) --	<0.003 (4) --
	Recharge (April 1998 through July 1998)	<0.02 (2) --	<0.002–0.0065 (4) --	<0.003 (4) --	<0.004 (4) --	<0.002 (4) --	0.0079–3.0 (9) 1.2*	<0.004–<0.05 (9) --	<0.003 (4) --
Sedgwick recharge site—deep monitoring wells									
	Baseline (June 1997 through April 1998)	<0.02 (3) --	<0.002 (5) --	<0.003 (5) --	<0.004 (5) --	<0.002 (5) --	e 0.0016–<0.05 (9) --	<0.004–<0.05 (9) --	<0.003 (5) --
	Recharge (April 1998 through July 1998)	<0.02 (2) --	<0.002 (4) --	<0.003 (4) --	<0.004 (4) --	<0.002 (4) --	<0.002–<0.05 (6) --	<0.004–<0.05 (6) --	<0.003 (4) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Pendimethalin (µg/L)	p,p'DDE (µg/L)	Prometon (µg/L)	Propachlor (µg/L)	Propazine (µg/L)	Simazine (µg/L)	Terbacil (µg/L)	Tebuthiuron (µg/L)	Trifluralin (µg/L)
U.S. Environmental Protection Agency (1999) drinking-water criteria										
		--	--	100 (HAL)	90 (HAL)	10 (HAL)	4.0 (MCL)	90 (HAL)	500 (HAL)	5.0 (HAL)
Surface-water monitoring sites (Little Arkansas River)										
07143672 (near Halstead)	Baseline (February 1995 through September 1997	<0.004–<0.018 (18) --	<0.01 (18) --	<0.008–0.06 (67) 0.018*	<0.007–0.2 (67) 0.006*	<0.01–2.5 (49) 0.04*	<0.005–0.11 (67) 0.008*	<0.007–e 0.009 (18) --	<0.01–0.039 (18) 0.01*	<0.012–0.026 (18) 0.003*
	Recharge (October 1997 through July 1998)	<0.004–0.013 (2) --	<0.006 (2) --	<0.05–0.20 (12) 0.05*	<0.007–<0.05 (12) --	<0.05–0.16 (10) --	<0.005–0.05 (12) --	<0.007 (2) --	<0.01 (2) --	0.003, 0.006 (2) --
07144100 (near Sedgwick)	Baseline (February 1995 through September 1997)	<0.003–<0.018 (16) --	<0.006–<0.01 (16) --	<0.003–0.8 (155) 0.024*	<0.007–0.94 (149) 0.004*	<0.01–5.2 (139) 0.04*	<0.005–0.23 (155) 0.011*	<0.007–<0.03 (16) --	<0.01–0.032 (16) 0.009*	<0.002–0.038 (16) 0.002*
	Recharge (October 1997 through July 1998)	<0.004 (3) --	<0.006 (3) --	<0.05–0.38 (36) 0.03*	<0.007–<0.05 (36) --	<0.05–0.23 (33) 0.07*	<0.05–0.11 (36) 0.03*	<0.007 (3) --	<0.01–0.08 (3) --	0.002–0.007 (3) --
Ground-water or diverted-water monitoring sites										
Background wells										
	Baseline (February 1995 through September 1997)	<0.004–<0.018 (26) --	<0.006–<0.01 (26) --	e 0.002–<0.05 (27) --	<0.007–<0.05 (27) --	<0.05 (1) --	<0.005–<0.05 (27) --	<0.007–<0.03 (26) --	<0.01–<0.015 (26) --	<0.002–<0.012 (26) --
	Recharge (October 1997 through July 1998)	<0.004 (2) --	<0.006 (2) --	<0.018–<0.05 (3) --	<0.007–<0.05 (3) --	<0.05 (1) --	<0.005–<0.05 (3) --	<0.007 (2) --	<0.01 (2) --	<0.002 (2) --
Domestic wells near Halstead										
	Baseline (April 1998)	--	--	<0.05 (2) --	<0.05 (2) --	<0.05 (2) --	<0.05 (2) --	--	--	--

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Pendime-thalin (µg/L)	p,p'DDE (µg/L)	Prometon (µg/L)	Propachlor (µg/L)	Propazine (µg/L)	Simazine (µg/L)	Terbacil (µg/L)	Tebu-thiuron (µg/L)	Trifluralin (µg/L)
Ground-water or diverted-water monitoring sites—Continued										
Domestic wells near Sedgwick										
	Baseline (August 1998)	--	--	<0.05 (5) --	<0.05 (5) --	<0.05 (5) --	<0.05 (5) --	--	--	--
Halstead diversion well site—diversion well										
	Recharge (April 1996 through July 1998—only sampled during recharge activities)	<0.004 (17) --	<0.006 (17) --	e 0.00089–<0.05 (77) --	<0.007–<0.05 (77) --	<0.05 (60) --	<0.005–<0.05 (77) --	<0.007 (17) --	e 0.003–<0.01 (17) 0.005*	<0.002 (17) --
Halstead diversion well site—shallow monitoring wells										
	Baseline (February 1995 through March 1996)	<0.004–<0.018 (9) --	<0.006–<0.01 (9) --	e 0.0045–<0.05 (14) --	<0.007–<0.05 (14) --	<0.05 (6) --	<0.005–<0.05 (14) --	<0.007–<0.03 (9) --	<0.01–e 0.028 (9) 0.013*	<0.002–<0.012 (9) --
	Recharge (April 1996 through July 1998)	<0.004 (18) --	e 0.0016–<0.006 (18) --	e 0.0053–<0.05 (59) --	<0.007–<0.05 (59) --	<0.05 (41) --	<0.005–<0.05 (59) --	<0.007 (18) --	e 0.0055–0.016 (18) 0.008*	<0.002 (18) --
Halstead diversion well site—deep monitoring well										
	Baseline (February 1995 through March 1996)	<0.004 (2) --	<0.006 (2) --	<0.018 (2) --	<0.007 (2) --	--	<0.005 (2) --	<0.007 (2) --	<0.01 (2) --	<0.002 (2) --
	Recharge (April 1996 through July 1998)	<0.004 (5) --	<0.006 (5) --	e 0.0024–<0.05 (14) --	<0.007–<0.05 (14) --	<0.05 (9) --	<0.005–<0.05 (14) --	<0.007 (5) --	e 0.0087–<0.01 (5) --	<0.002 (5) --
Halstead recharge site—shallow monitoring wells										
	Baseline (May 13, 1997)	<0.004 (2) --	<0.006 (2) --	<0.018 (2) --	<0.007 (2) --	--	<0.005 (2) --	<0.007 (2) --	<0.01 (2) --	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.004 (17) --	<0.006 (17) --	e 0.0024–<0.05 (34) --	<0.007–<0.05 (34) --	<0.05 (17) --	e 0.0013–<0.05 (17) --	<0.007 (17) --	e 0.0028–<0.01 (17) --	<0.002 (17) --

Table 10. Summary of organic compound concentrations in water samples collected during baseline, 1995–98, and artificial recharge, 1996–98, conditions—Continued

Data-collection site (figs. 1, 2, 3, or 5)	Water-quality conditions (dates of collection)	Pendime-thalin (µg/L)	p,p'DDE (µg/L)	Prometon (µg/L)	Propachlor (µg/L)	Propazine (µg/L)	Simazine (µg/L)	Terbacil (µg/L)	Tebu-thiuron (µg/L)	Trifluralin (µg/L)
				Ground-water or diverted-water monitoring sites—Continued						
Halstead recharge site—deep monitoring wells										
	Baseline (May 13, 1997)	<0.003, <0.004 (2) --	<0.006 (2) --	<0.018 (2) --	<0.007 (2) --	--	<0.005 (2) --	<0.007 (2) --	<0.01 (2) --	<0.002 (2) --
	Recharge (May 29, 1997 through July 1998)	<0.004 (17) --	<0.006 (17) --	e 0.0032–<0.05 (36) --	<0.007–<0.05 (36) --	<0.007–<0.05 (17) --	<0.005–<0.05 (36) --	<0.007 (17) --	e 0.0045–<0.01 (17) --	<0.002 (17) --
Sedgwick recharge site—treated diverted source water										
	Recharge (April 1998 through July 1998—only sampled during recharge activities)	<0.004 (3) --	<0.006 (3) --	e 0.0099–0.06 (24) --	<0.007–0.06 (24) --	<0.05–0.07 (21) --	<0.005–<0.05 (24) --	<0.007 (3) --	e 0.0045–<0.01 (3) --	<0.002 (3) --
Sedgwick recharge site—shallow monitoring wells										
	Baseline (June 1997 through February 1998)	<0.004 (4) --	<0.006 (4) --	<0.018–<0.05 (12) --	<0.007–<0.05 (12) --	<0.05–0.07 (8) --	e 0.0025–<0.05 (12) --	<0.007 (4) --	e 0.004–<0.01 (4) --	<0.002 (4) --
	Recharge (April 1998 through July 1998)	<0.004 (4) --	<0.006 (4) --	e 0.0077–<0.05 (9) --	<0.007–<0.05 (9) --	<0.05–0.05 (6) --	<0.005–<0.05 (9) --	<0.007 (4) --	<0.01 (4) --	<0.002 (4) --
Sedgwick recharge site—deep monitoring wells										
	Baseline (June 1997 through April 1998)	<0.004 (5) --	e0.00061–<0.006 (5) --	<0.018–<0.05 (9) --	<0.007–<0.05 (9) --	<0.05 (4) --	<0.005–<0.05 (9) --	<0.007 (5) --	<0.01 (5) --	<0.002 (5) --
	Recharge (April 1998 through July 1998)	<0.004 (4) --	<0.006 (4) --	<0.018–<0.05 (6) --	<0.007–<0.05 (6) --	<0.05 (3) --	<0.005–<0.05 (6) --	<0.007 (4) --	<0.01 (4) --	<0.002 (4) --

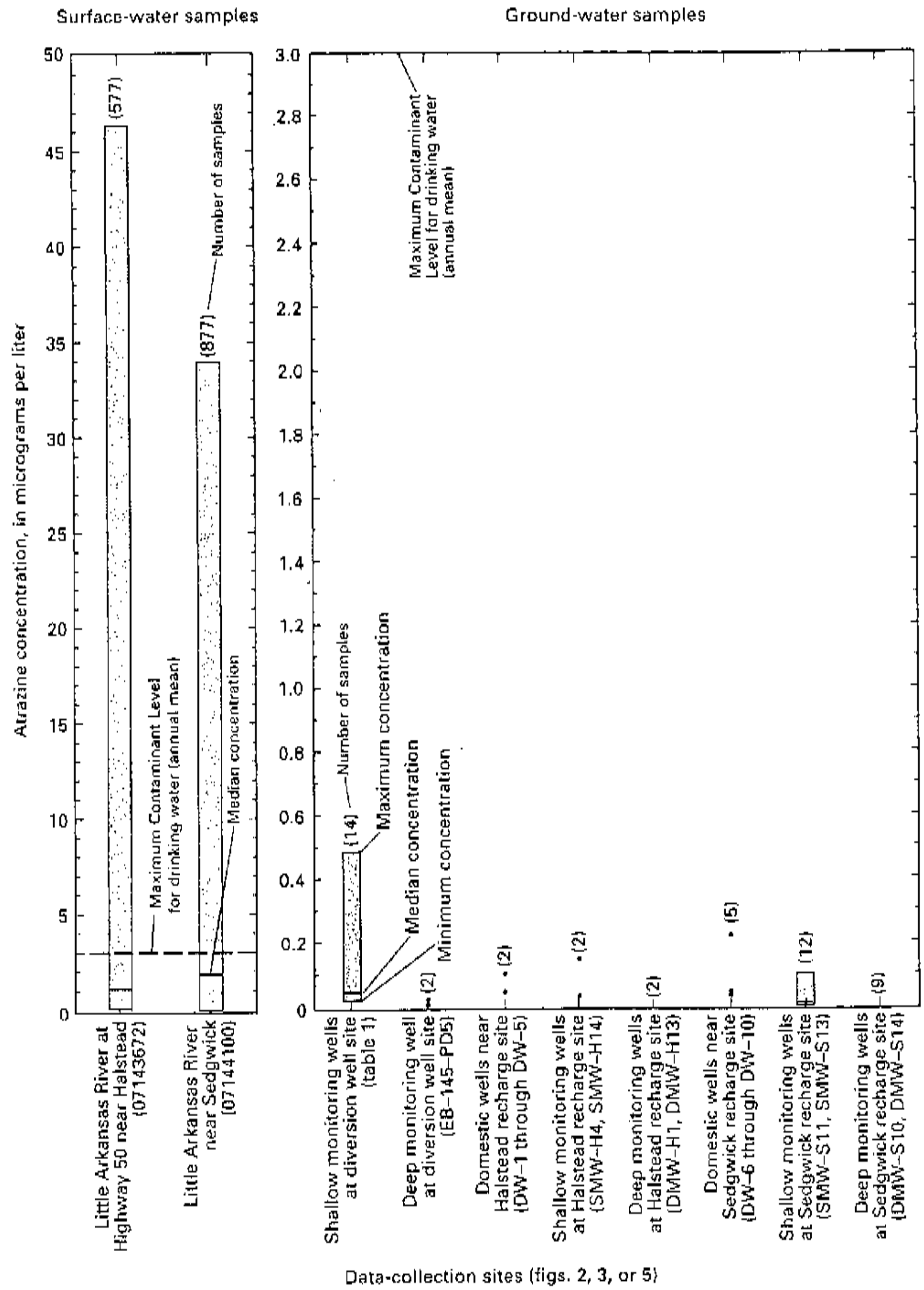


Figure 10. Ranges in atrazine concentrations in surface- and ground-water samples during baseline water-quality monitoring. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

surface-water monitoring site (table 10). Metolachlor was detected in smaller concentrations in water from the Sedgwick surface-water monitoring site and in ground water from the Sedgwick recharge site, from the Halstead diversion well site, and from background monitoring wells adjacent to the Little Arkansas River.

Propazine is an herbicide that has not been sold in the United States since 1990, but it is often found as an impurity in the atrazine that is applied to fields of corn and grain sorghum and as a result is detected in small amounts in surface water (Thurman and others, 1998). Propazine has an HAL of 10 μg/L. Baseline concentrations in surface-water samples ranged from less than 0.01 to 5.2 μg/L. Propazine was detected in samples from two shallow monitoring wells (SMW–S11 and SMW–S13) at the Sedgwick recharge site, but it was not detected in ground-water samples from any other site during baseline monitoring.

Although alachlor, cyanazine, metolachlor, and propazine were all detected in concentrations exceeding 20 percent of their respective MCL or HAL during baseline water-quality monitoring, atrazine was the only pesticide detected with great frequency at concentrations greater than its MCL, especially during spring and summer runoff when herbicide application was followed by periods of intense rainfall (Christensen and Ziegler, 1998a). Atrazine also was detected in samples from nearly all wells.

PRELIMINARY EFFECTS OF ARTIFICIAL RECHARGE, 1996–98

Halstead Recharge System

The instantaneous discharge of the Little Arkansas River at Highway 50 near Halstead (fig. 11) frequently exceeded the minimum flow requirements of the demonstration project permit (42 ft^3/s) during late spring and early summer. The minimum flow requirements for October 1 through March 31 (20 ft^3/s) also were frequently exceeded. In fact, from October 1, 1997, through March 31, 1998, streamflow in the Little Arkansas River at Highway 50 near Halstead remained above 20 ft^3/s for the entire period. The flow requirements at this site did not apply to aquifer tests conducted from April through July 1996.

Effects on Ground-Water Levels

Water levels in the Little Arkansas River at Halstead (station 07143680, fig. 3), near the diversion well site, were nearly always higher than water levels in the adjacent monitoring wells at the site (fig. 11), indicating that the stream generally was recharging the aquifer at this location. Water levels in the stream were higher at this location as a result of backwater from a low-head dam located about 1 mi downstream from the diversion site (fig. 2).

From May 29, 1997, through July 31, 1998, at the Halstead recharge site, a total of about 307 Mgal were recharged into the *Equus* Beds aquifer (fig. 12). Most of this water was recharged using the recharge well at the site. The recharge well began operation in August 1997, and as of July 31, 1998, it had recharged about 272 Mgal (Burns and McDonnell, 1998). The amount of water recharged with the recharge trench and recharge basins is small in comparison. The effect of these recharge activities on water levels can be evaluated by examining water-level data from the monitoring wells at the Halstead recharge site (fig. 13). Water levels in shallow monitoring wells showed little or no change from May 1997 through July 1998. However, water levels in deep wells showed increasing water-level altitudes during extended periods of artificial recharge. Water levels receded, however, when artificial recharge stopped.

Effects on Ground-Water Quality

The water pumped from the Halstead diversion well was sampled approximately every 5 days during recharge activities at a control building on the Halstead recharge site. As indicated in the previous description of the "Halstead Recharge System," the source water at the Halstead diversion well site originates in the river alluvium; therefore, the quality of the source water diverted for recharge was not the same as the quality of the surface water from the Little Arkansas River. Generally, the constituents in the diverted water, such as dissolved solids, bacteria, and organic compounds, occurred in smaller concentrations than in the surface water. This happens, in part, because the aquifer materials, especially clay and organic matter, act as a natural filter capable of removing some chemical constituents as the water passes through. In addition, the ground water near the stream mixes with the surface water that is induced into the alluvial aquifer from the Little Arkansas River and dilutes the concentrations in the ground water.

During artificial recharge conditions at the Halstead site (May 1997 through July 1998), sodium concentrations in water from the Little Arkansas River at Highway 50 near Halstead ranged from 11 to 500 mg/L, with a median concentration of 100 mg/L; from April 1996 through July 1998 concentrations in ground water from the Halstead diversion well site ranged from 49 to 120 mg/L (fig. 14). Sodium concentrations in water diverted for recharge from the diversion well ranged from 51 to 72 mg/L. From May 1997 through July 1998, sodium concentrations in water from shallow monitoring wells SMW–H4 and SMW–H14 at the Halstead recharge site ranged from 45 to 100 mg/L compared to the baseline concentrations of 40 and 150 mg/L (table 5). The largest effect on sodium concentrations in water from the shallow monitoring wells may be related to the small domestic

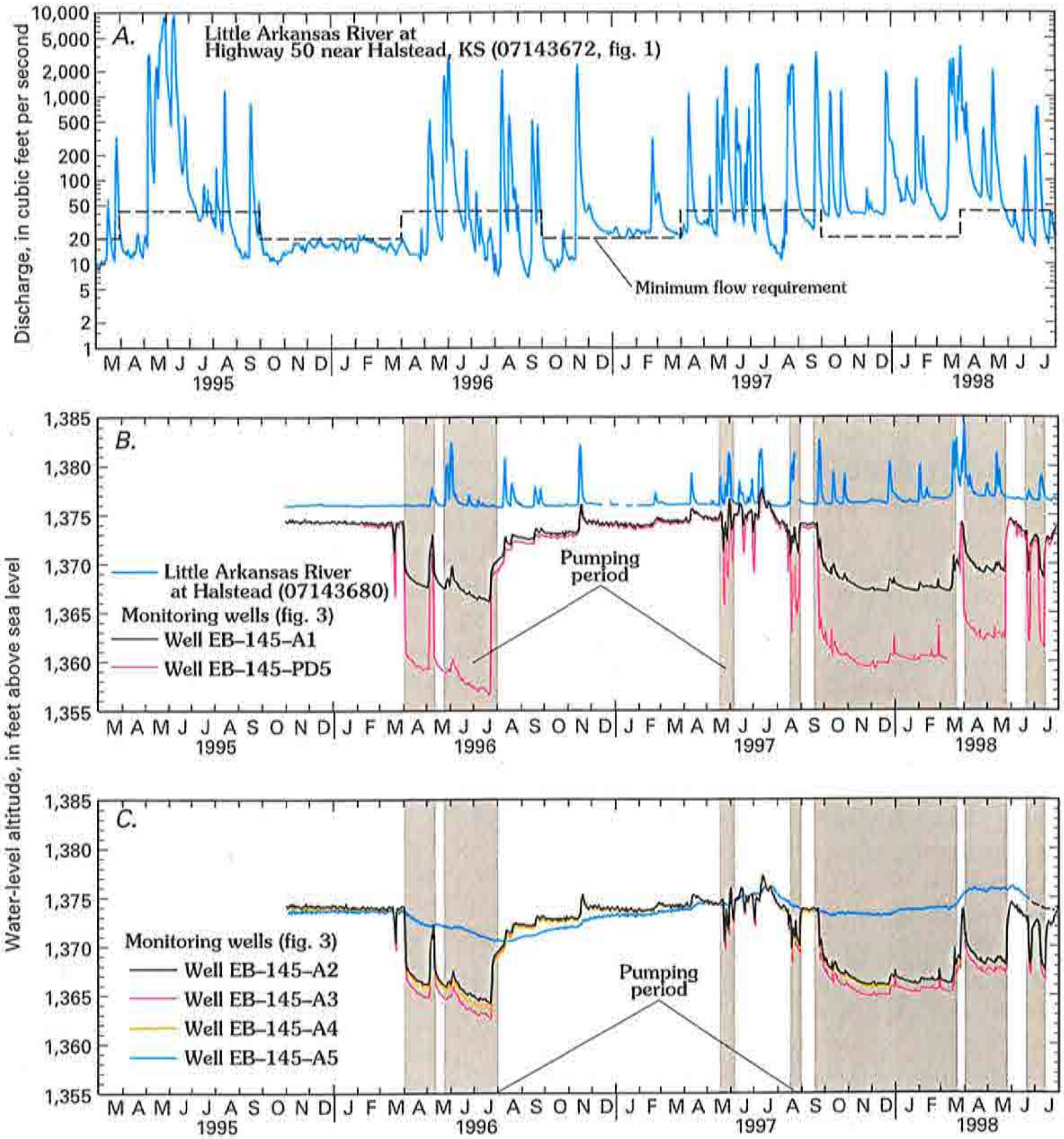


Figure 11. *(A)* Discharge of Little Arkansas River at Highway 50 near Halstead and *(B, C)* water levels in the *Equus* Beds aquifer at Halstead diversion well site, March 1995–July 1998.

sewage-treatment lagoon east of the recharge site. An additional source of sodium could be subsurface geologic formations. In water from deep monitoring wells DMW–H1 and DMW–H13, sodium concentrations ranged from 27 to 69 mg/L, compared to the baseline concentrations of 29 and 31 mg/L (fig. 14). Sodium concentrations in water from the deep monitoring wells have increased; however, not enough baseline samples were collected to describe changes in seasonal variability of sodium (table 5).

From October 1997 through July 1998, chloride concentrations in surface water from the Little Arkansas River at Highway 50 near Halstead ranged from 20 to 930 mg/L (table 5); from April 1996 through July 1998 in ground water from the Halstead diversion well site, the range in chloride concentrations was from 12 to 280 mg/L (fig. 15). Chloride concentrations in water from the diversion well ranged from 22 to 78 mg/L (table 5). From May 29, 1997, through July 1998, chloride concentrations in water from shallow

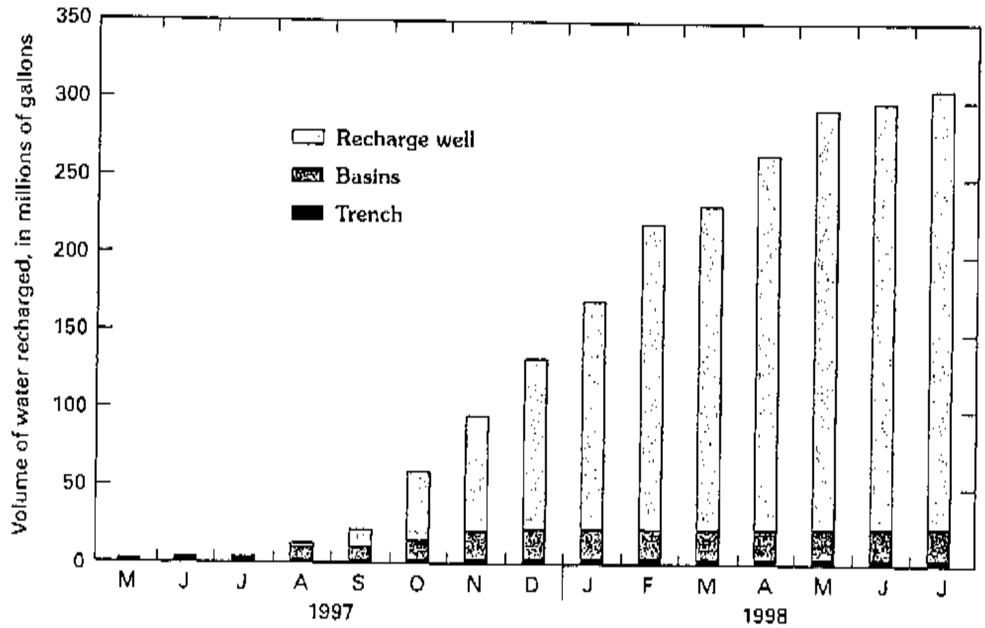


Figure 12. Cumulative volume of water recharged at Halstead recharge site, May 1997–July 1998 (source of data: Burns and McDonnell, 1998).

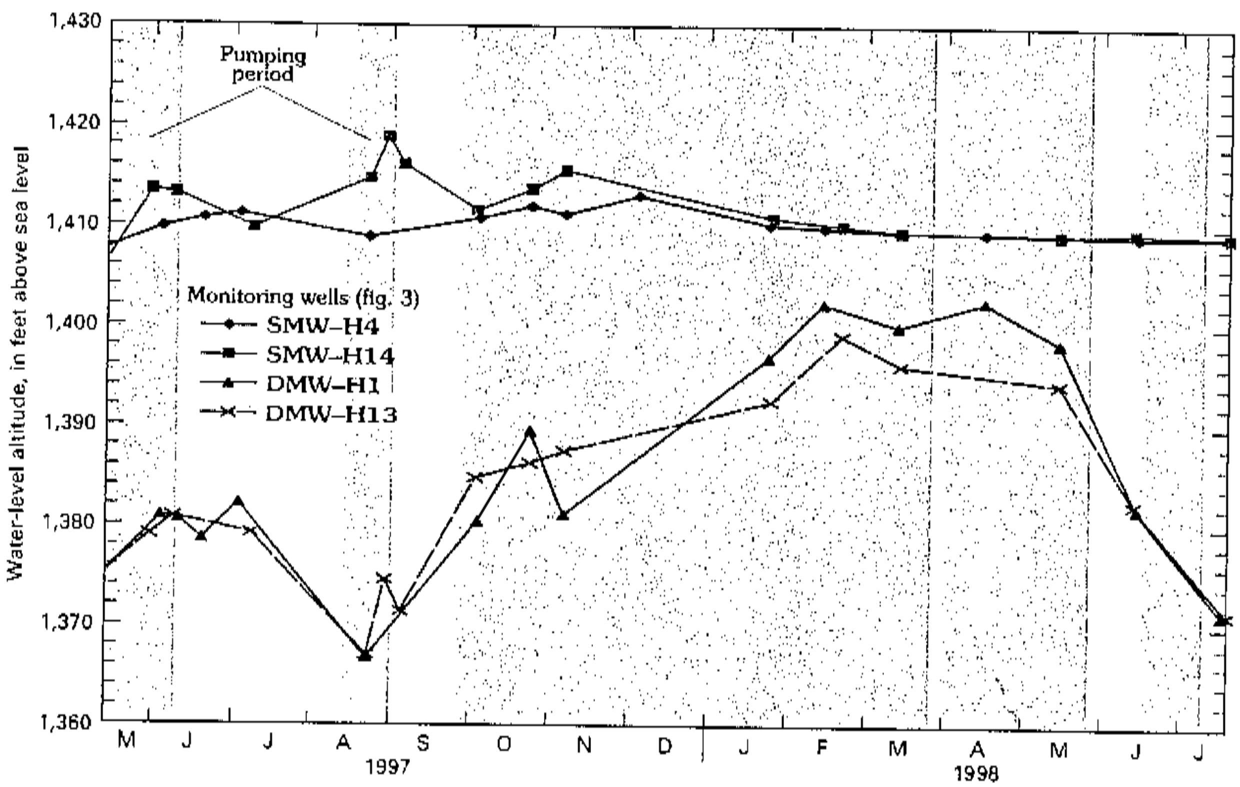


Figure 13. Water-level altitudes in monitoring wells completed in the *Equus* Beds aquifer at Halstead recharge site, May 1997–July 1998.

monitoring wells SMW–H4 and SMW–H14 at the Halstead recharge site ranged from 39 to 110 mg/L, compared to baseline concentrations of 20 and 290 mg/L. In water from the two shallow monitoring wells, the chloride concentration of one sample exceeded the SMCL of 250 mg/L prior to recharge; this did not occur in any of the 31 samples collected after recharge. The sewage lagoon east of the site probably contributes to the larger chloride concentrations in shallow wells during both baseline and artificial recharge conditions. In water from deep monitoring wells DMW–H1 and DMW–H13, chloride concentrations ranged from 5.8 to 64 mg/L, compared to baseline concentrations of 5.8 and 8.3 mg/L (fig. 15).

From October 1997 through July 1998, nitrite plus nitrate concentrations in water from the Little Arkansas River at Highway 50 near Halstead ranged from less than 0.02 to 2.4 mg/L; from April 1996 through July 1998 in ground water from the Halstead diversion well site, the range was 0.01 to 0.03 mg/L (table 6, fig. 16). Nitrite plus nitrate concentrations in diverted water ranged from less than 0.02 to 0.78 mg/L. From May 29, 1997, through July 1998, in water from shallow monitoring wells SMW–H4 and SMW–H14 at the Halstead recharge site, concentrations ranged from 0.01 to 9.1 mg/L, compared to baseline concentrations of 1.7 and 7.0 mg/L (table 6), indicating similar nitrite plus nitrate concentrations before and after recharge in the shallow wells. However, these concentrations are near the MCL of 10 mg/L for nitrite plus nitrate. These large nitrite plus nitrate concentrations in water from the shallow monitoring wells may be due to the sewage lagoon east of the Halstead recharge site and fertilizers applied on nearby fields. In 29 water samples from

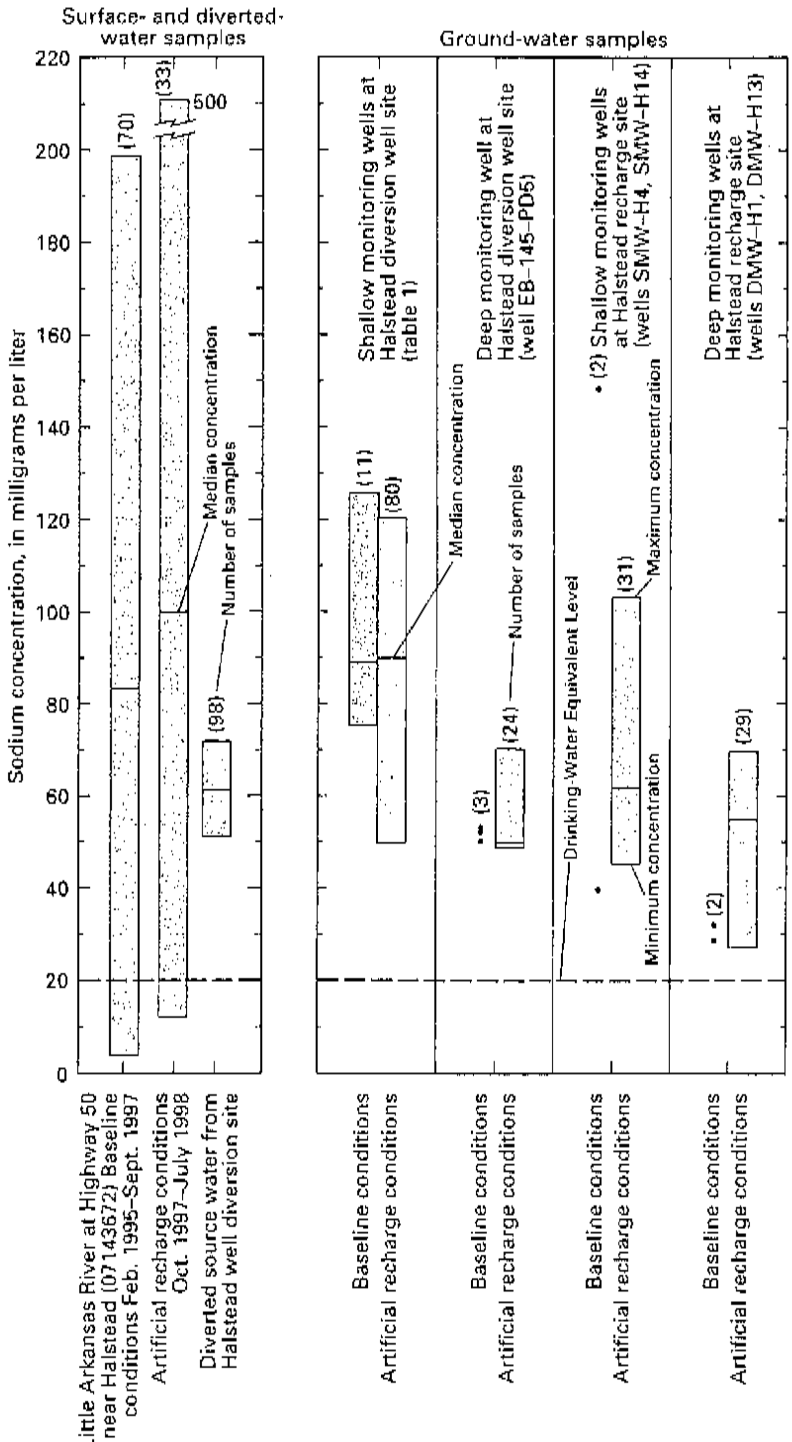


Figure 14. Ranges in sodium concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Drinking-Water Equivalent Level from U.S. Environmental Protection Agency (1999).

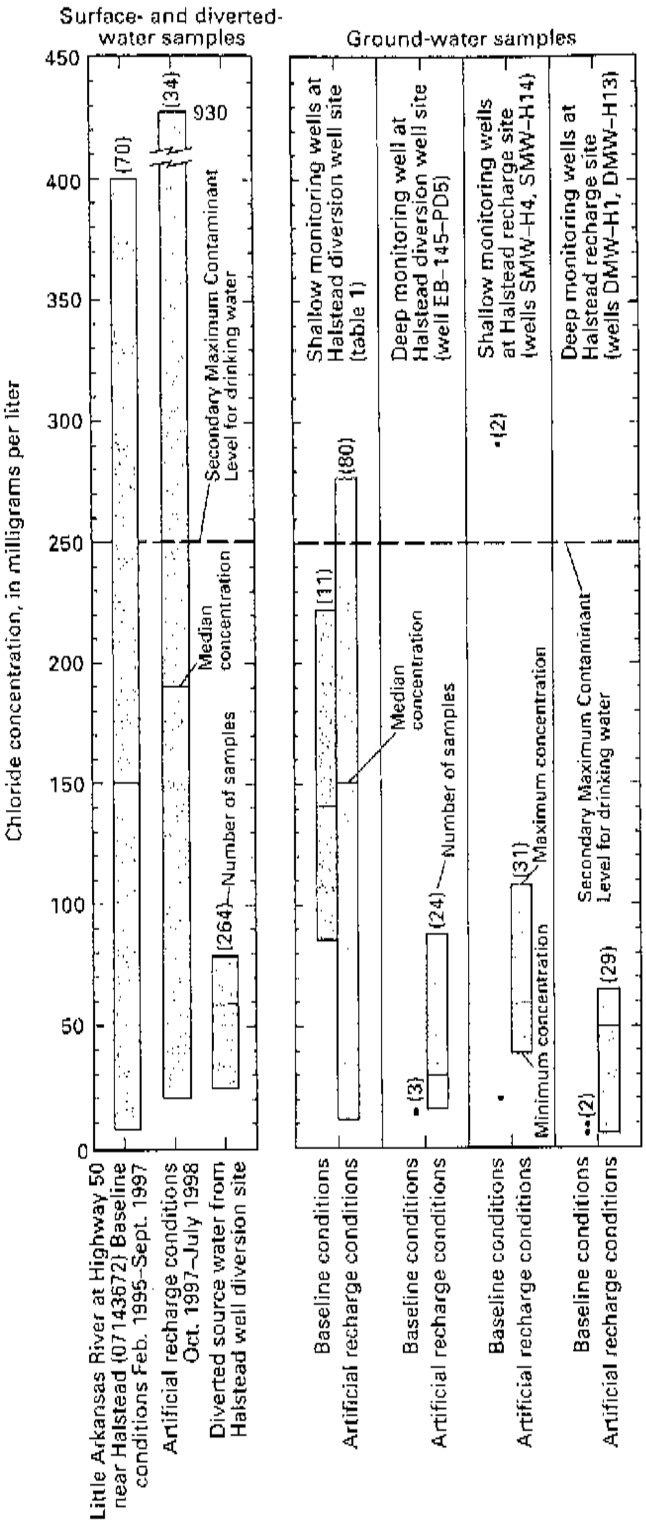


Figure 15. Ranges in chloride concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

deep monitoring wells at the Halstead recharge site, there was only one detection of nitrite plus nitrate greater than 0.02 mg/L. Nitrite plus nitrate was not detected in water from the deep monitoring wells during baseline water-quality sampling. As with the shallow wells, nitrite plus nitrate concentrations in water from the deep monitoring wells at the Halstead recharge site were similar before and after recharge.

From October 1997 through July 1998, iron concentrations in water from the Little Arkansas River at Highway 50 near Halstead ranged from less than 5.0 to 30 μg/L (table 7). From April 1996 through July 1998 in water from monitoring wells at the Halstead diversion well site, iron concentrations ranged from less than 10 to 4,900 μg/L (fig. 17). Water from the well at the diversion well site (well EB–145–PD5, fig. 3) showed an increase in iron concentrations since recharge activities began, probably because of induced surface water from the Little Arkansas River. Iron concentrations in water from the diversion well ranged from 170 to 1,200 μg/L. Iron concentrations in water from shallow monitoring wells SMW–H4 and SMW–H14 at the Halstead recharge site were relatively small from May 29, 1997, through July 1998, with a range from less than 5.0 to 6.6 μg/L, compared to similar baseline concentrations of less than 5.0 and 11 μg/L. In water from deep monitoring wells DMW–H1 and DMW–H13, the range was 17 to 1,300 μg/L, compared to baseline concentrations of 7.9 and 300 μg/L. Although there was an increase in iron concentrations in water from the deep monitoring wells, samples from domestic wells in the Halstead area during April 1998 also indicate some large concentrations of iron in water from the aquifer (table 7).

From October 1997 through July 1998, manganese concentrations in water from the Little Arkansas River at Highway 50 near Halstead ranged from less than 5.0 to 340 μg/L. In ground water from the Halstead diversion well site for April 1996 through July 1998, concentrations of manganese ranged from less than 5.0 to 3,400 μg/L (fig. 17). Manganese concentrations in diverted water ranged from 450 to 810 μg/L. For the Halstead recharge site from May 29, 1997, through July 1998, manganese concentrations ranged from less than 5.0 to 15 μg/L in water from shallow monitoring wells SMW–H4 and SMW–H14, compared to baseline concentrations of 20 and 68 μg/L (table 7). The range in manganese concentrations in water from deep monitoring wells DMW–H1 and DMW–H13 was from 230 to 750 µg/L, compared to baseline concentrations of 210 and 260 µg/L.

Concentrations of both iron and manganese are affected by environmental conditions (Hem, 1992). Because the conditions are more chemically reducing (less oxygen available) in ground water, concentrations of iron and manganese tend to be higher in ground water from the diversion well site than in surface water from the Little Arkansas River near Halstead. Iron and manganese can plug the aquifer material when they precipitate as oxides. At the Halstead recharge site, precipitation of iron caused some problems during infiltration tests by plugging the upper filter fabric in the recharge trench (Burns and McDonnell, 1998). The original filter fabric was replaced to alleviate this problem, and the new filter fabric was cleaned during subsequent recharge operations. However, no iron plugging problems have been observed in the recharge well. Manganese has an oxidation process similar to iron (Hem, 1992) and also may precipitate and interfere with recharge.

Total organic carbon concentrations in water from the Little Arkansas River at Highway 50 near Halstead were 6.6 and 6.7 mg/L (table 8) for the two samples collected during artificial recharge conditions (October 1997 through July 1998). The range in total organic carbon concentrations during baseline conditions was 3.8 to 27 mg/L. This greater range probably is due to the greater number of samples collected during baseline conditions. No substantial changes in total organic carbon were observed in water samples from the Halstead diversion well site or the Halstead recharge site.

The possible filtering effect of clay layers overlying the *Equus* Beds aquifer is particularly evident in the concentrations of total coliform bacteria and atrazine in water from

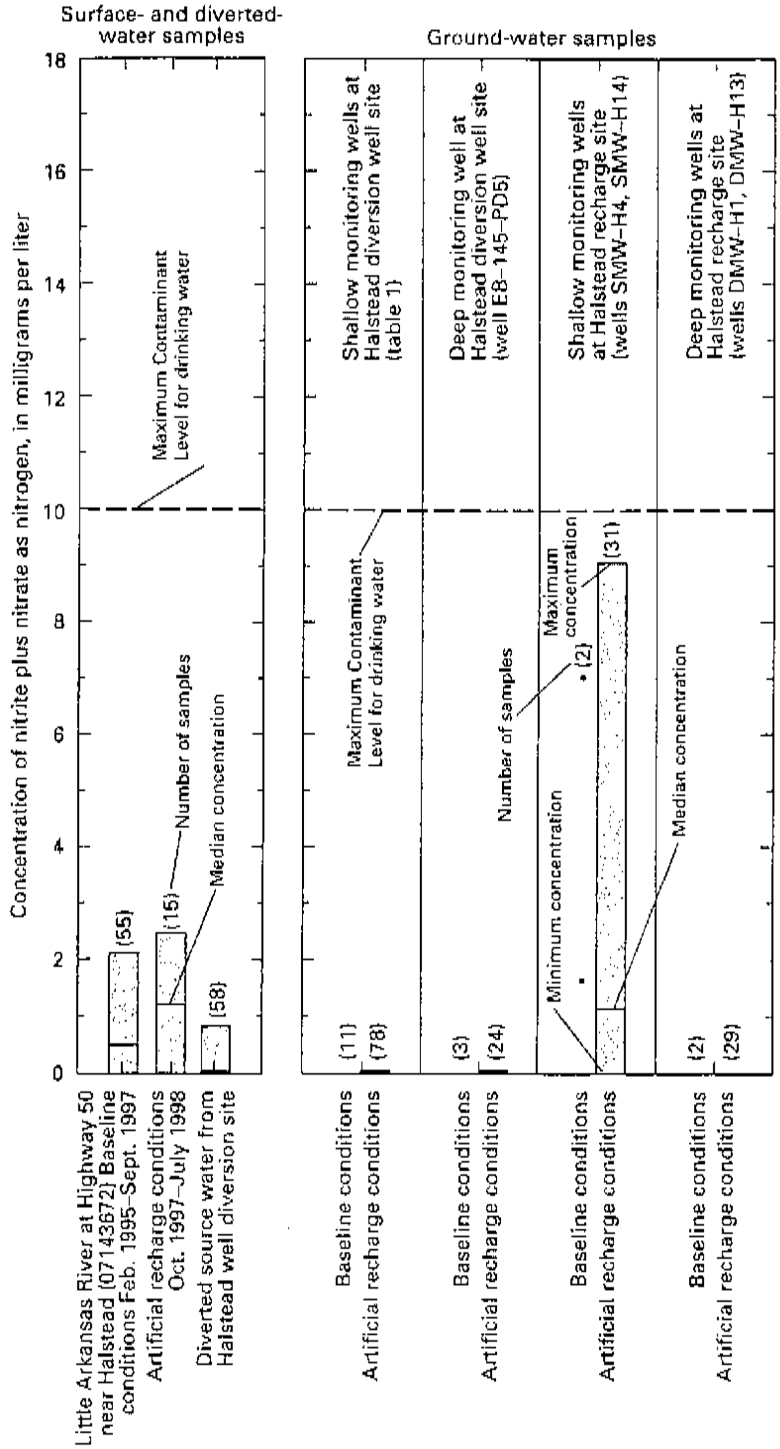


Figure 16. Ranges in nitrite plus nitrate concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Maximum Contaminant Levels from U.S. Environmental Protection Agency (1999).

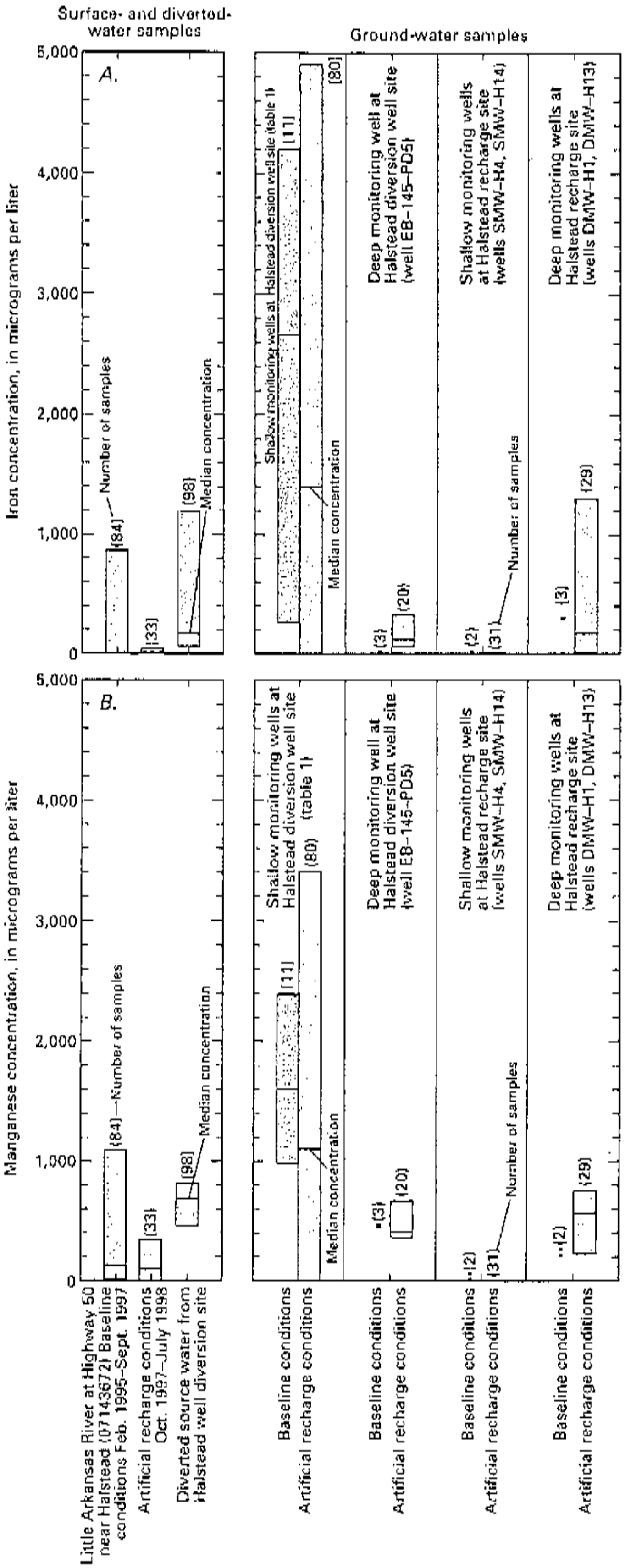


Figure 17. Ranges in *(A)* iron and *(B)* manganese concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Secondary Maximum Contaminant Levels from U.S. Environmental Protection (1999).

the Halstead recharge system. Total coliform bacteria densities in water from the Little Arkansas River at Highway 50 near Halstead ranged from 110 to 53 000 col/100 mL (table 9) from October 1997 through July 1998. At the diversion well site, total coliform bacteria densities in water from shallow monitoring wells ranged from less than 2 to 14 col/100 mL. Total coliform bacteria densities in water from the diversion well ranged from less than 1 to less than 100 col/100 mL. In water from shallow monitoring wells SMW–H4 and SMW–H14 at the Halstead recharge site, densities ranged from less than 1 to 4 col/100 mL during artificial recharge conditions; the range before recharge was less than 1 to 1 col/100 mL (table 9). In water from deep monitoring wells, one sample collected in July 1997 from well DMW–H13 had a bacteria density of 80 col/100 mL; however, most of the other samples had bacteria densities less than 1 col/100 mL, approximating baseline conditions. In addition to the filtering effect of the clay layers, small dissolved-oxygen content is a possible factor causing decreased bacterial densities in ground water from the Halstead recharge system during artificial recharge conditions.

From October 1997 through July 1998, atrazine concentrations in surface water from the Little Arkansas River at Highway 50 near Halstead ranged from less than 0.10 to 12 μg/L (by ELISA) during artificial recharge conditions. At the Halstead diversion well site, concentrations in ground water were less than 0.001 to 2.2 μg/L (by GC/MS) (table 10, fig. 18). Water from the deep monitoring well at the diversion well site (well EB–145–PD5, fig. 3) showed an increase in atrazine concentrations since recharge activities began, presumably because of surface water from the river being induced into the ground water. Atrazine concentrations in water from the diversion well ranged from less than 0.006 to 0.09 μg/L (by GC/MS). Atrazine concentrations in water from the shallow and deep monitoring wells at the Halstead recharge site ranged from less than 0.001 to 0.09 μg/L (by GC/MS) during artificial recharge conditions. Concentrations of atrazine in water from the monitoring wells generally were no larger than those in water from the diversion well and did not exceed the largest baseline concentration of 0.14 μg/L in water from shallow monitoring well SMW–H4. However, baseline atrazine concentrations were higher in water from the shallow monitoring wells at the Halstead recharge site than in water from the diversion well, indicating that shallow ground water may be affected by pesticides applied on nearby fields.

Concentrations of sodium, chloride, nitrite plus nitrate, bacteria, and atrazine were generally larger in surface water from the Little Arkansas River at Highway 50 near Halstead than in ground water from the diversion well site before and after recharge. From the examination of baseline- and artificial-recharge data collected from the Halstead surface-water site (station 07143672, fig. 1) and diversion well site, several additional observations were made with respect to chloride and atrazine. First, there were seasonal fluctuations in both chloride and atrazine (figs. 19 and 20) in surface water. However, these seasonal fluctuations did not coincide. Chloride concentrations were largest during the winter (fig. 19), whereas atrazine concentrations were largest in spring and early summer (fig. 20). Second, there is a time lag in seasonal fluctuations between chloride and atrazine concentrations in the surface water and chloride and atrazine concentrations in the ground water, although significant changes in the chloride concentrations in water from some wells are not evident from the data. During periods of extended pumping, the time lag appears to have decreased possibly because of increased hydraulic gradient between surface and ground water near the diversion well site. Finally, chloride and atrazine concentrations in samples from the deep monitoring wells at the Halstead recharge site increased after recharge began to values approximating that of the recharge water (figs. 19 and 20).

Sedgwick Recharge System

Effects on Ground-Water Levels

Samples were collected and preliminary testing at the Sedgwick recharge site began in October 1997. Testing was intermittent until April 1998, when recharge operations began. As of July 31, 1998, about 31 Mgal of water had been recharged to the *Equus* Beds aquifer at the Sedgwick site. All of the water at this site was recharged through basins (fig. 5). The effect of the recharge activity on water levels can be

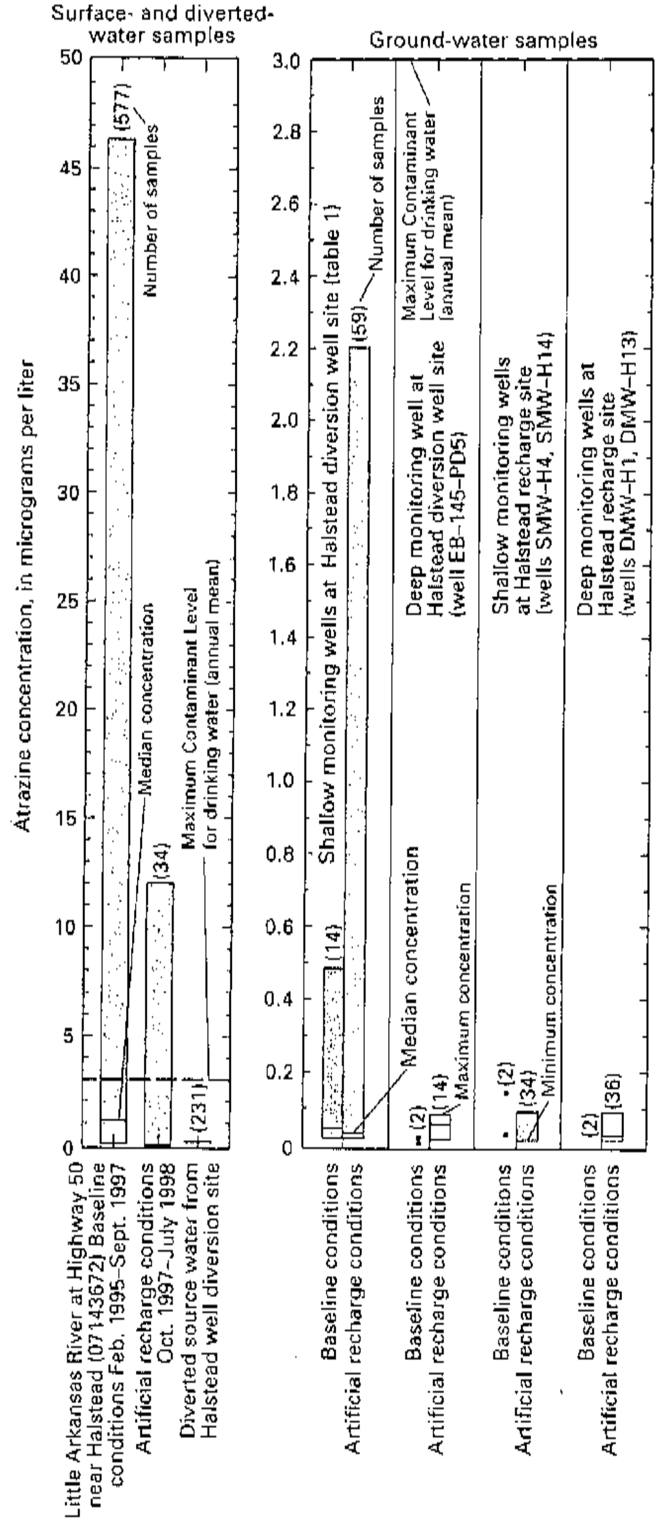


Figure 18. Ranges in atrazine concentrations in surface-, diverted-, and ground-water samples during baseline conditions and recharge conditions. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

evaluated by examining water-level data from the monitoring wells (SMW–S11, SMW–S13, DMW–S10, and DMW–S14, fig. 21). During artificial recharge, all four monitoring wells showed initial increases in water levels; however, water levels receded after about 2 months of recharge.

Effects on Ground-Water Quality

Samples of recharge water were collected daily, and monitoring wells were sampled monthly. Because the source water diverted from the Little Arkansas River is treated prior to pumping into the basins, most physical properties, such as turbidity and suspended solids, generally had smaller concentrations in the recharge water than in the surface water from the Sedgwick site (station 07144100, fig. 1).

Sodium concentrations in water from the Little Arkansas River at Sedgwick (station 07144100, fig. 1) ranged from 11 to 110 mg/L from October 1997 through July 1998 (table 5). Sodium concentrations in treated source water diverted from the Little Arkansas River ranged from 21 to 81 mg/L. From April through July 1998, in water from shallow monitoring wells SMW–S11 and SMW–S13 at the Sedgwick recharge site, the range in sodium concentrations before recharge was 14 to 58 mg/L. After recharge activities began, the range was 20 to 50 mg/L. In water from deep wells DMW–S10 and DMW–S14, the range in sodium concentrations during baseline conditions was 83 to 93 mg/L; during artificial recharge conditions, the range was 75 to 86 mg/L. The median sodium concentration in water from these deep wells was actually smaller during artificial recharge conditions (81 mg/L) than during baseline conditions (86 mg/L). With respect to sodium, it would appear that the recharge activities at the Sedgwick site do not have a substantial effect on the water quality of the *Equus* Beds aquifer (fig. 22). However, there was not enough water recharged at this site to determine annual fluctuations that may occur.

From October 1997 through July 1998, chloride concentrations in treated source water diverted from the Little Arkansas River for recharge ranged from 26 to 180 mg/L. Chloride concentrations in water from shallow monitoring wells SMW–S11 and SMW–S13 ranged from 10 to 78 mg/L (fig. 23). From April through July 1998, water from deep monitoring well DMW–S14 showed a decrease in chloride concentrations since recharge began, whereas chloride concentrations in water from deep monitoring well DMW–S10 remain unchanged (table 5). Generally, chloride concentrations in the source water were not affected by treatment (fig. 24). Once the recharged water infiltrated into the aquifer, the concentrations of chloride in water from nearby monitoring wells generally did not change substantially from what they were prior to recharge (fig. 24).

From October 1997 through July 1998, nitrite plus nitrate concentrations in water from the Little Arkansas River at Sedgwick ranged from less than 0.02 to 2.9 mg/L (table 6). In the treated source water diverted from the Little Arkansas River, nitrite plus nitrate concentrations ranged from 0.13 to 1.8 mg/L. In water from shallow monitoring wells SMW–S11 and SMW–S13, nitrite plus nitrate concentrations ranged from 1.8 to 13 mg/L prior to artificial recharge; during artificial recharge conditions, the range in concentrations in water from these wells was 1.2 to 15 mg/L (fig. 25). The large concentrations of nitrite plus nitrate in water from these shallow wells (larger than source-water concentrations) may be the result, at least partly, of fertilizer application on nearby fields. During baseline conditions in water from deep monitoring wells DMW–S10 and DMW–S14, nitrite plus nitrate concentrations ranged from 0.05 to 2.7 mg/L; during artificial recharge conditions, the range was 0.16 to 4.6 mg/L. The median concentrations in water from these deep wells were 0.09 mg/L before recharge and 2.0 mg/L after recharge. Even though the median nitrite plus nitrate concentrations in water from these wells increased during artificial recharge conditions, there were not enough samples collected prior to recharge to define the seasonal variability of the nitrite plus nitrate concentrations, and the increased median concentrations may be a reflection of unknown seasonal variability. Baseline conditions occurred from June 1997 through February 1998, and artificial recharge conditions occurred from April through July 1998, making definition of seasonal variability difficult at this site.

Although iron and manganese caused infiltration problems at the Halstead recharge site because they precipitated on the filter fabric in the recharge trench, they were not a factor at the Sedgwick site. The largest iron concentration detected in water from monitoring wells at the Sedgwick recharge site after recharge began was 68 µg/L. In fact, iron was not detected in most monitoring well samples. Manganese concentrations in ground water at the Sedgwick recharge site either decreased or did not change (fig. 26). Because

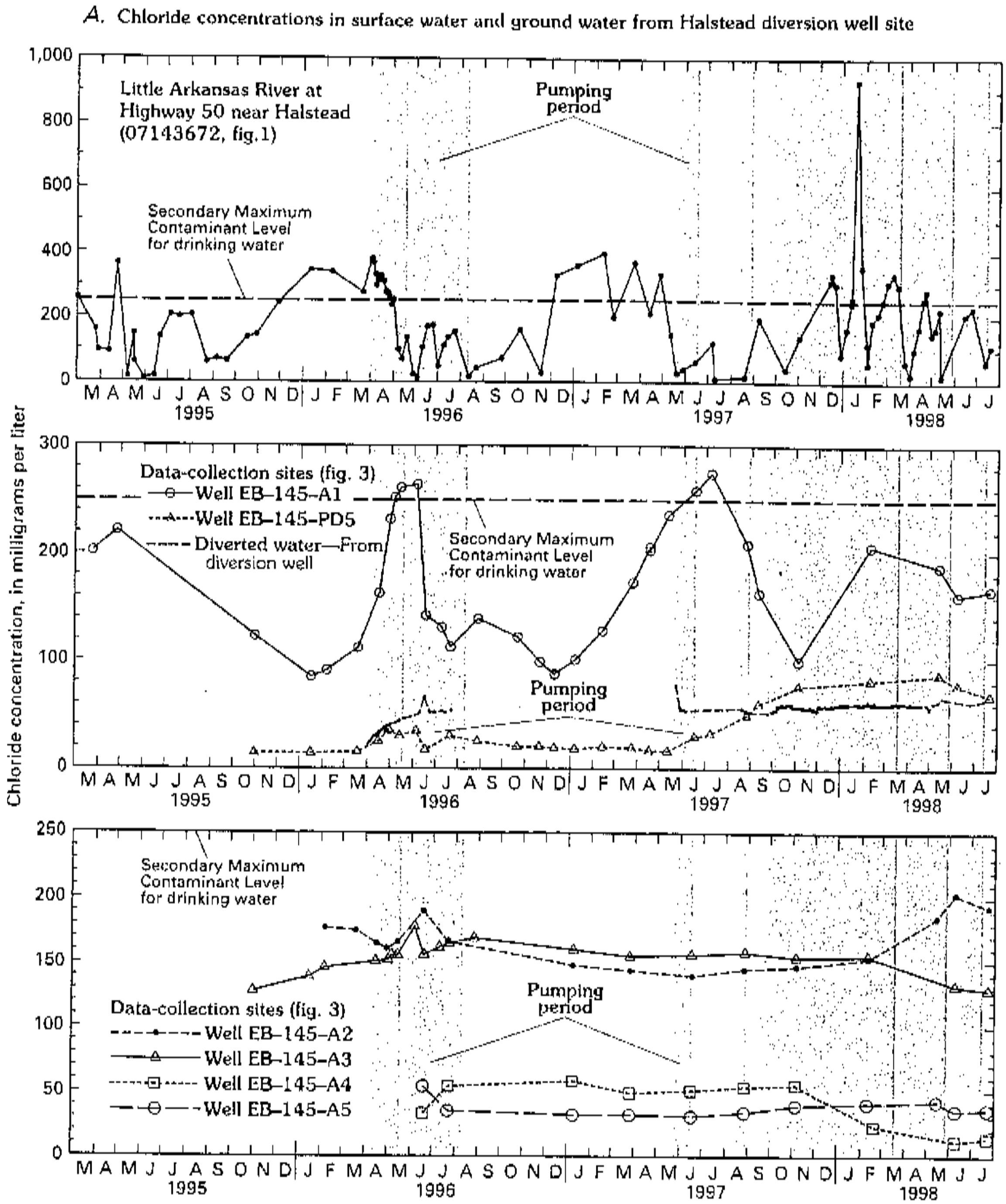


Figure 19. Comparison of chloride concentrations *(A)* in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and *(B)* in diversion well water and ground water from Halstead recharge site, May 1997–July 1998. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

concentrations of iron and manganese were generally larger during baseline conditions at the Sedgwick recharge site, iron and manganese may have precipitated during artificial recharge conditions and that precipitation may have been caused by the rapid infiltration of oxygenated recharge water.

From October 1997 through July 1998, TOC concentrations in water from the Little Arkansas River near Sedgwick ranged from 5.7 to 7.3 mg/L. Treated diverted water had slightly smaller concentrations during the same period—from 4.3 to 5.9 mg/L. In general, water from monitoring wells (SMW–S11, SMW–S13, *B.* Chloride concentrations in diversion well water and ground water from Halstead recharge site

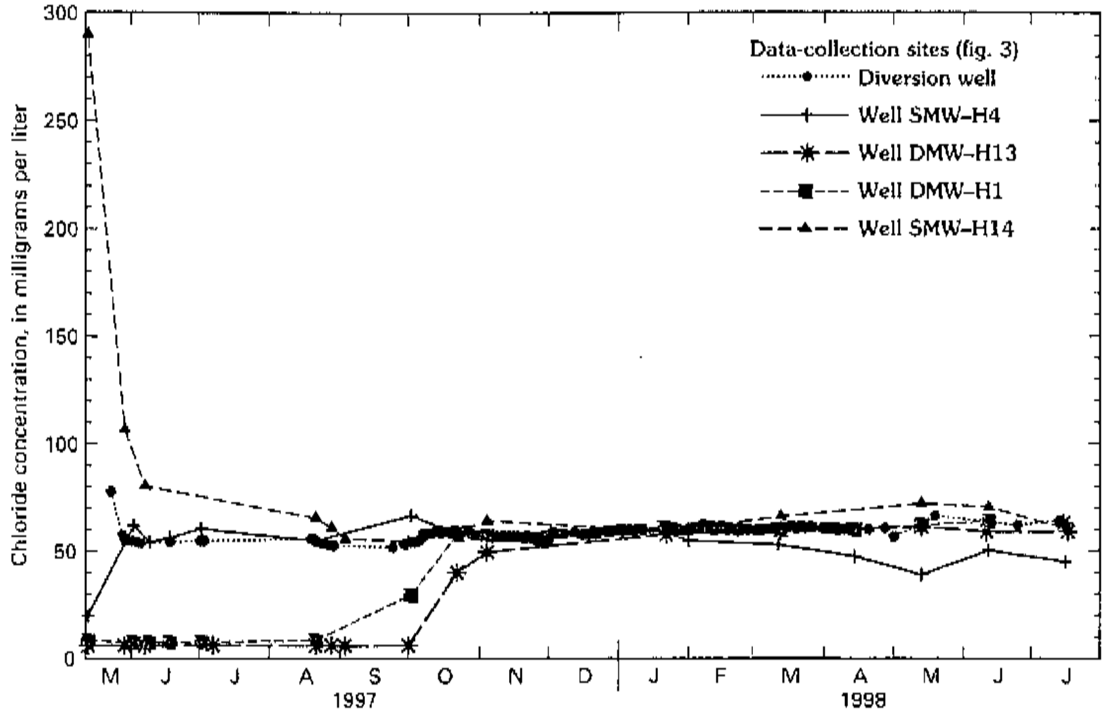


Figure 19. Comparison of chloride concentrations *(A)* in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and *(B)* in diversion well water and ground water from Halstead recharge site, May 1997–July 1998. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999)—Continued.

DMW–S10, and DMW–S14) at the Sedgwick recharge site had smaller TOC concentrations during artificial recharge conditions than during baseline conditions.

From October 1997 through July 1998, total coliform bacteria in water from the Little Arkansas River near Sedgwick ranged from 30 to 27,000 col/100 mL. Total coliform bacteria ranged from less than 1 to 400 col/100 mL in treated source water diverted from the Little Arkansas River. However, densities of total coliform bacteria were not a significant concern in water from monitoring wells at the Sedgwick recharge site with only one detection (of 1 col/100 mL) after recharge began. Total coliform bacteria may become a concern during longer periods of recharge.

Atrazine, because of its frequent use on row crops in the study area and its potential effects on water quality, has been monitored frequently since February 1995 in water from the Little Arkansas River (Christensen and Ziegler, 1998a). Atrazine concentrations in surface water typically are larger in the spring and summer when herbicides are applied and when excessive rains cause greater runoff to streams (Goolsby and others, 1997). In treated source water from the Little Arkansas River, atrazine concentrations determined by ELISA ranged from less than 0.1 to 6.8 μg/L (fig. 27). The maximum atrazine concentration (determined by GC/MS) detected in water from shallow monitoring wells SMW–S11 and SMW–S13 was 0.36 μg/L, exceeding the baseline maximum concentration of 0.1 μg/L (fig. 27). Atrazine was not detected in water from the deep monitoring wells at the site. The addition of PAC to the treated source water was effective in decreasing the concentrations of atrazine to concentrations similar to baseline concentrations; therefore, concentrations of atrazine in water from nearby monitoring wells were similar to what they were prior to recharge, with the exception of atrazine concentrations in water from well SMW–S11. The seasonal variation in atrazine concentrations in water from the Little Arkansas River near Sedgwick and water from shallow monitoring wells SMW–S11 and SMW–S13 is shown in figure 28.

Documentation of the preliminary effects of artificial recharge at the Sedgwick site are important because of the large differences between constituent concentrations in the surface water and the baseline

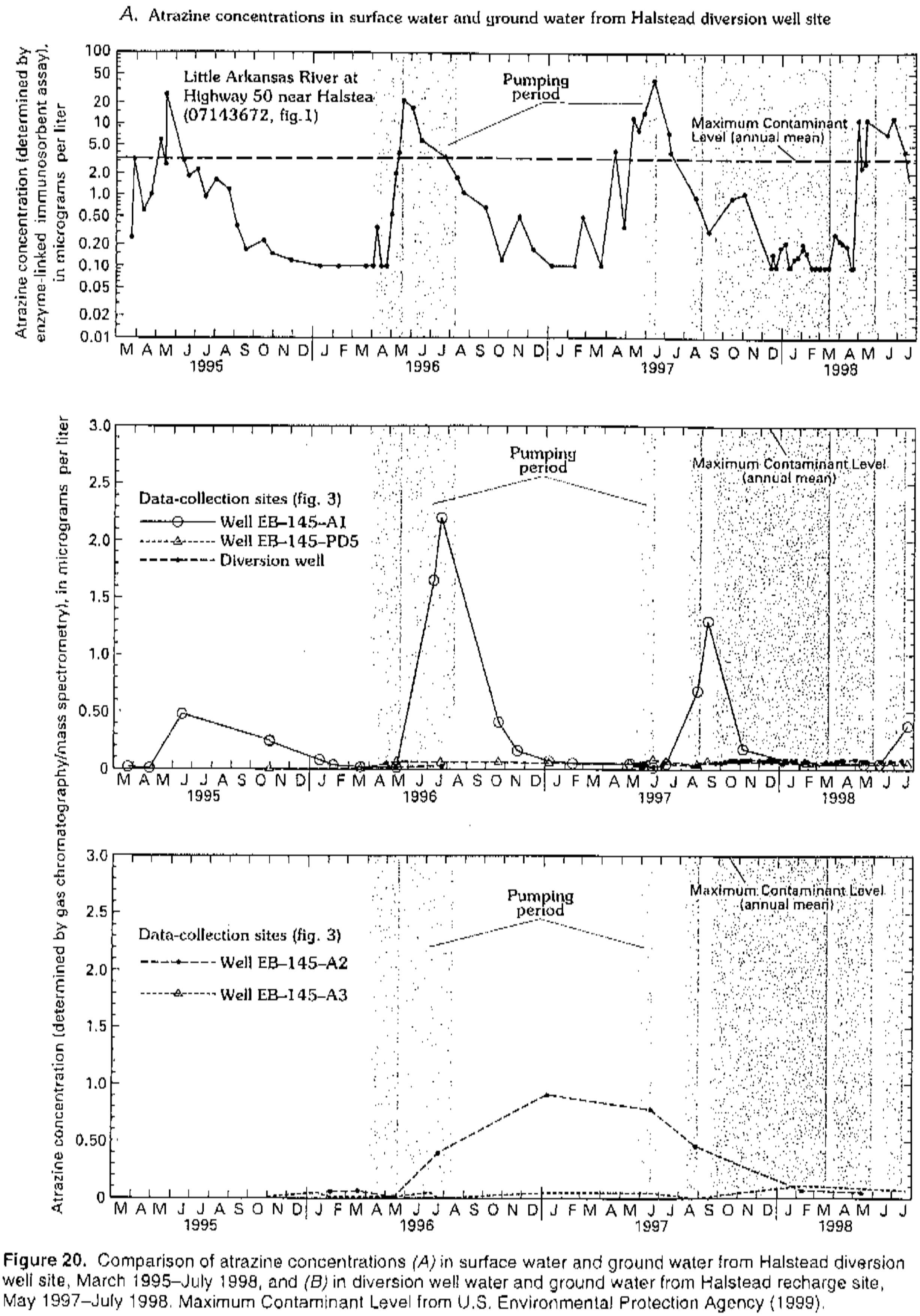


Figure 20. Comparison of atrazine concentrations *(A)* in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and *(B)* in diversion well water and ground water from Halstead recharge site, May 1997–July 1998. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

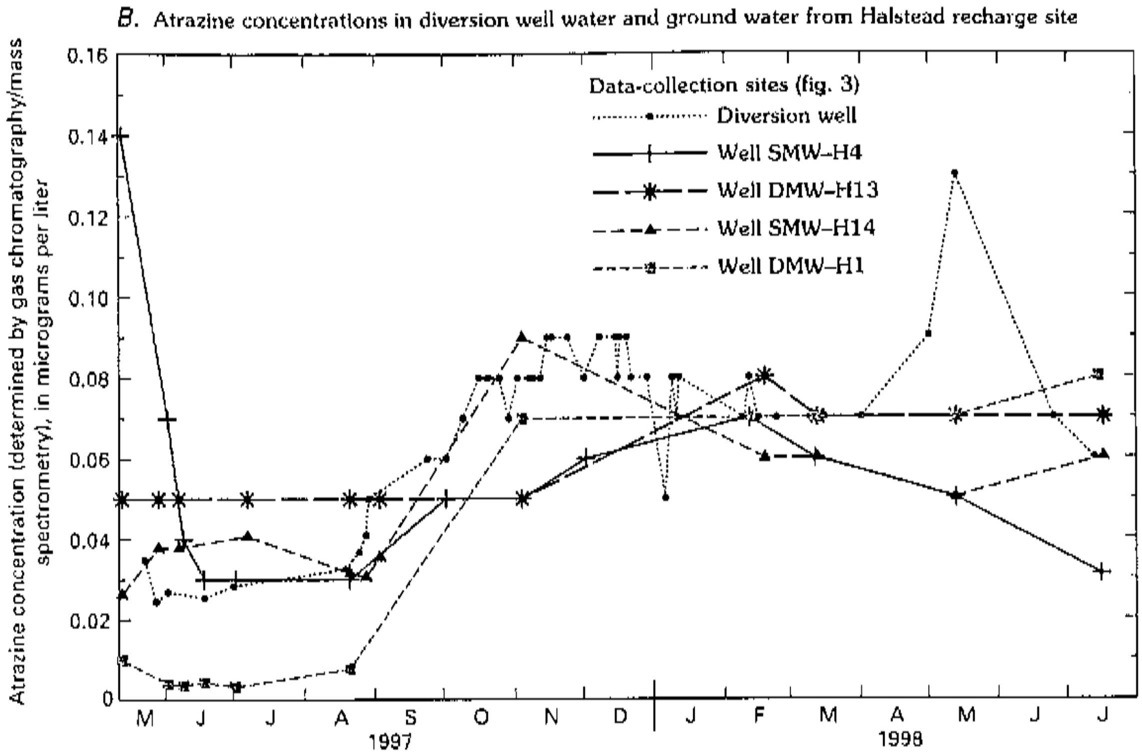


Figure 20. Comparison of atrazine concentrations *(A)* in surface water and ground water from Halstead diversion well site, March 1995–July 1998, and *(B)* in diversion well water and ground water from Halstead recharge site, May 1997–July 1998. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999)—Continued.

water-quality conditions of the receiving aquifer water. Sodium, chloride, bacteria, and atrazine were detected in larger concentrations in untreated surface water than in the receiving ground water. This is especially true in the case of bacteria and atrazine where surface-water concentrations were many times that of baseline ground-water concentrations.

Although sodium concentrations and bacteria densities in ground water have not shown definitive changes since artificial recharge began, concentrations of chloride and atrazine have increased in some of the monitoring wells at the Sedgwick recharge site. However, at this site, only 31 Mgal of water have been recharged compared with 307 Mgal recharged at the Halstead site. Continued monitoring at the Sedgwick site during further recharge operations will help ensure that the large concentrations of certain constituents in the source water do not adversely affect the quality of the receiving ground water.

COMPATIBILITY OF SOURCE WATER FOR ARTIFICIAL RECHARGE

Compatibility of source water for artificial recharge was determined by comparing major-ion and trace-element concentrations in source water and receiving ground water and by evaluating the potential for adverse chemical reactions. Stiff diagrams (Stiff, 1951) of mean concentrations of the major ions during baseline and artificial recharge conditions are shown in figure 29. The shape of the Stiff diagram was used to indicate if there were differences in the chemistry of the source water and that of the receiving ground water. For example, the addition of source water with small concentrations of calcium and bicarbonate to ground water with large concentrations of these constituents may dilute the existing large concentrations. Alternatively, depending on water chemistry, the source water may cause more calcium and bicarbonate to be dissolved from the aquifer material, which then, with further changing water chemistry, may lead to plugging of the aquifer material as the water flows downgradient, thereby limiting recharge.

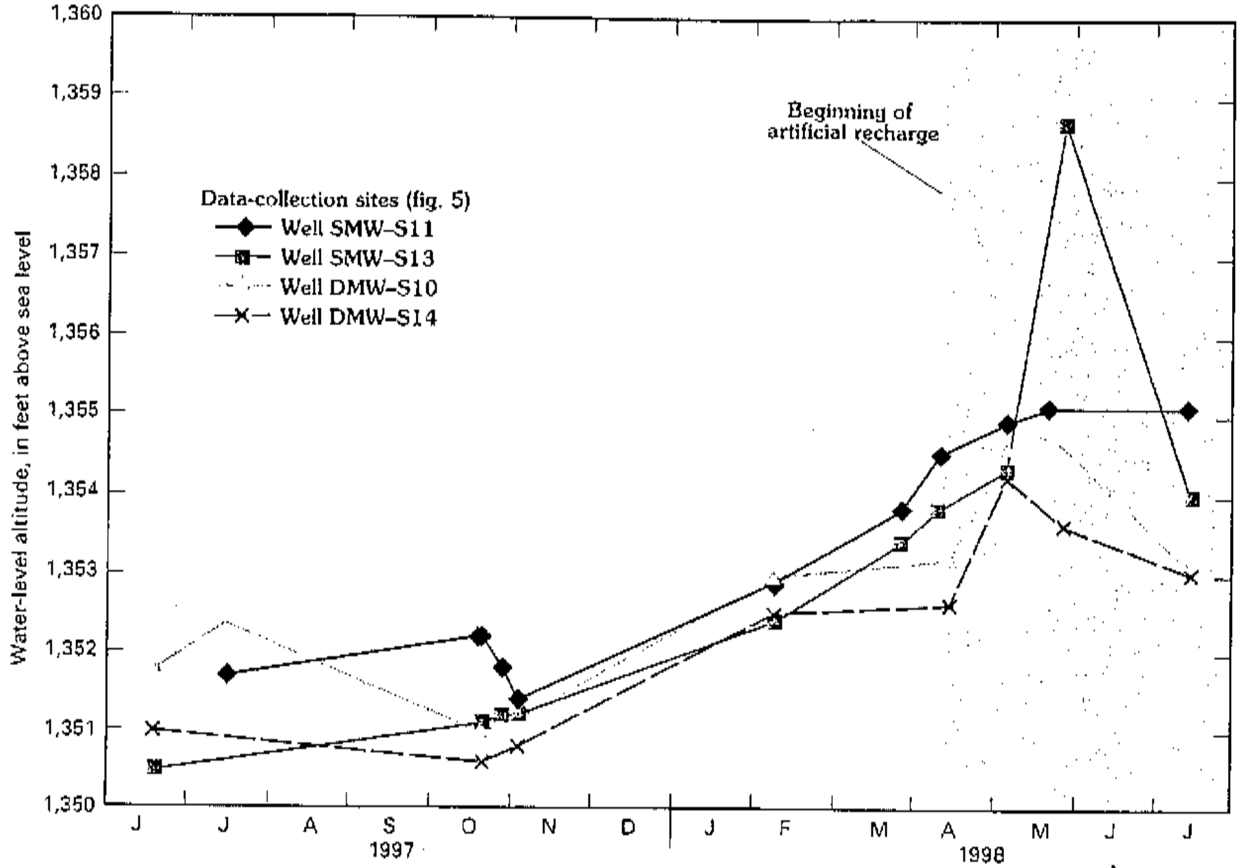


Figure 21. Water levels in the *Equus* Beds aquifer at Sedgwick recharge site, June 1997–July 1998.

The Stiff diagrams of the water from Little Arkansas River at Highway 50 near Halstead (station 07143672, fig. 1) are very similar to the Stiff diagrams of water from shallow monitoring wells at the Halstead diversion well site during baseline and artificial recharge conditions (fig. 29*A*). Stiff diagrams for water from the deep well, however, show smaller chloride plus fluoride concentrations compared to water from the shallow monitoring wells during baseline and artificial recharge conditions. The Stiff diagram for water from the deep well is similar to the Stiff diagram for source water from the diversion well. The Stiff diagram for source water also is similar to that for water from monitoring wells at the Halstead recharge site, although samples from the deep ground water at the recharge site had smaller concentrations of chloride (fig. 29*B*). The Stiff diagrams of water from the shallow monitoring wells at the Halstead recharge site during baseline conditions differ from source water from the diversion well and each other; however, during artificial recharge conditions, the Stiff diagram of water from the shallow monitoring wells is similar to that for the source water. Smaller increases in chloride concentrations in water from the deep wells at the Halstead diversion well site and from deep monitoring wells at the Halstead recharge site also are illustrated in figures 29*A* and 29*B*.

The Stiff diagram for water from the Little Arkansas River near Sedgwick is nearly identical to that for the treated source water used for recharge (fig. 29*C*). At the Sedgwick recharge site, shallow ground water has much smaller concentrations of chloride plus fluoride than concentrations of chloride plus fluoride in treated source water. The treated source water also has larger concentrations of bicarbonate plus carbonate that, when combined with the large calcium concentrations in shallow ground water, could lead to chemical precipitation of calcium carbonate. The precipitation of calcium carbonate may cause some plugging of aquifer materials. Shallow ground water at the Sedgwick site has much larger concentrations of sulfate compared to other ground water. Larger sulfate concentrations in the shallow ground water may be an indication of oxidation of sulfide minerals in the aquifer material, consistent with an unconfined aquifer. The part of the *Equus* Beds aquifer into which the

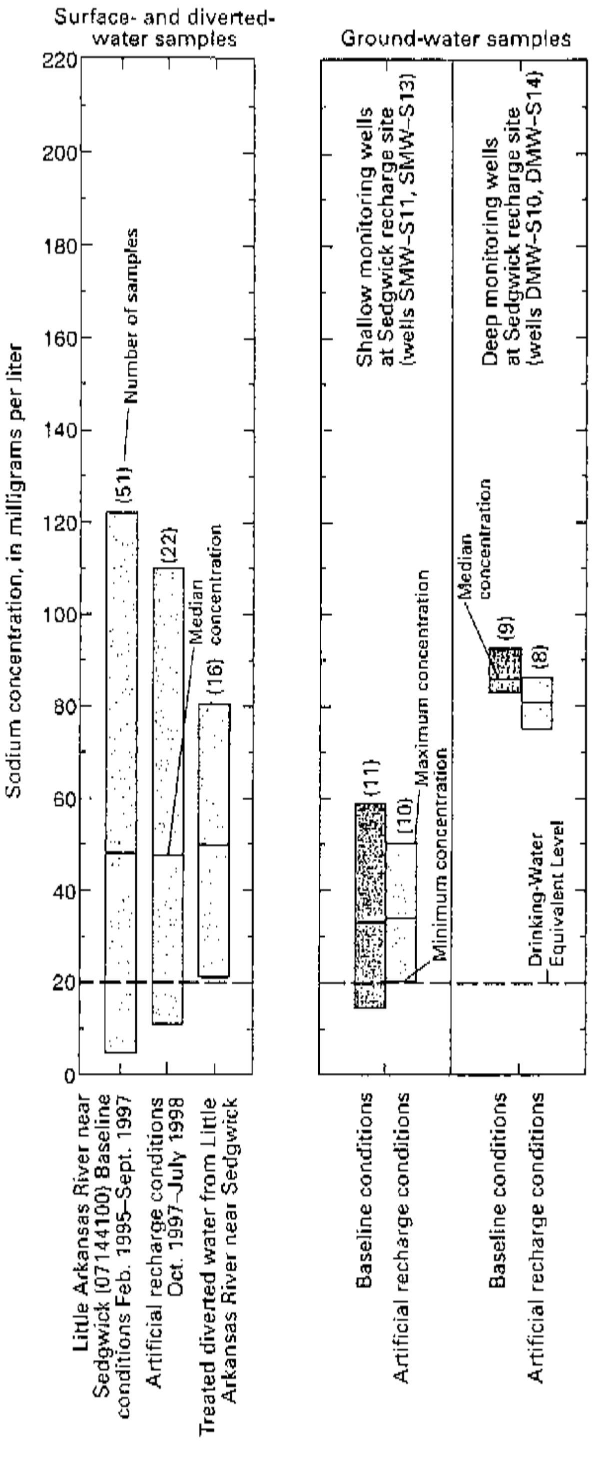


Figure 22. Ranges in sodium concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Drinking-Water Equivalent Level from U.S. Environmental Protection Agency (1999).

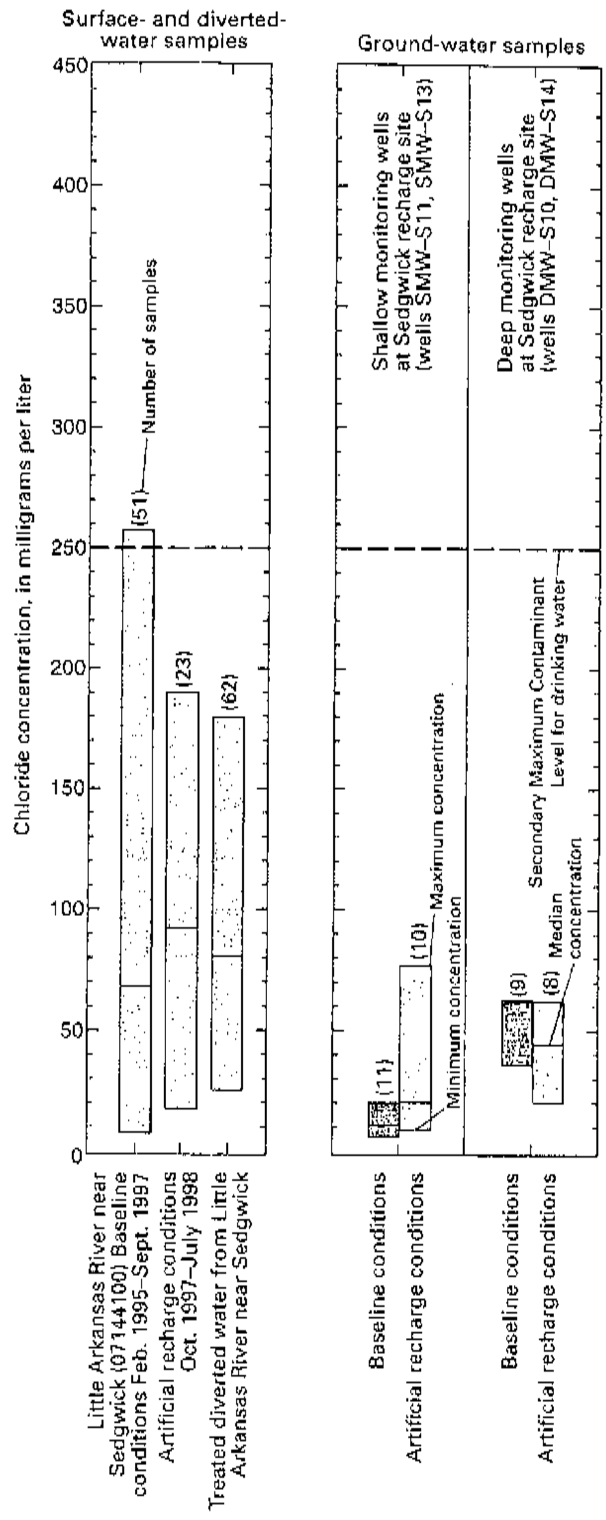


Figure 23. Ranges in chloride concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

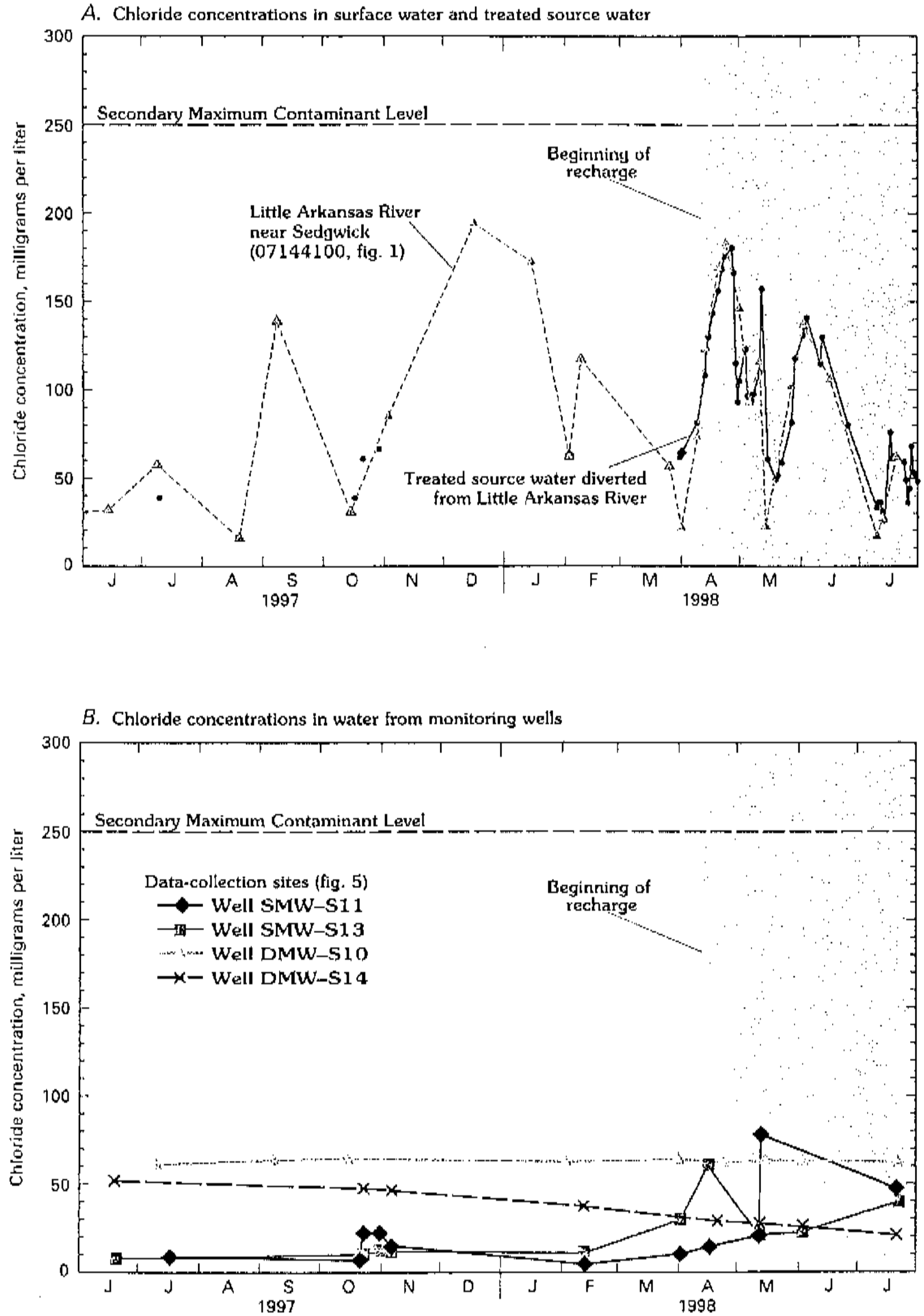


Figure 24. Comparison of chloride concentrations in *(A)* surface water and treated source water from Little Arkansas River and in *(B)* water from monitoring wells at Sedgwick recharge site, June 1997–July 1998. Secondary Maximum Contaminant Level from U.S. Environmental Protection Agency (1999). Data-collection sites shown in figure 5.

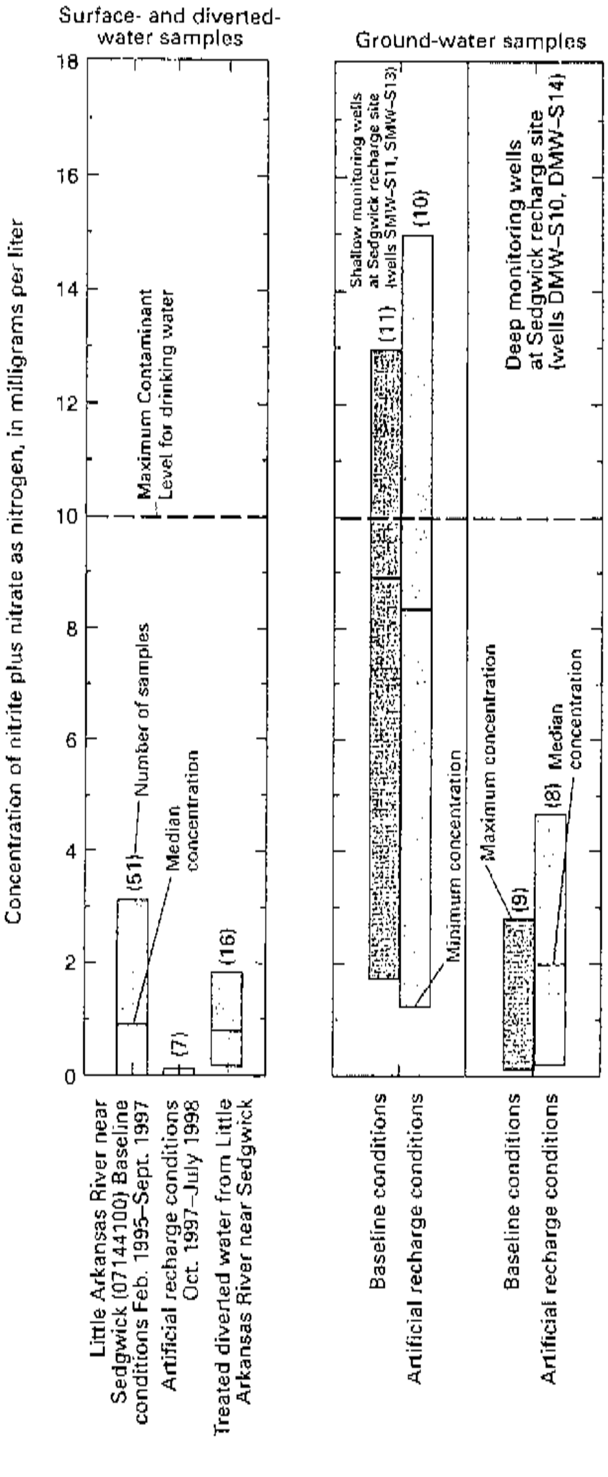


Figure 25. Ranges in nitrite plus nitrate concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

shallow wells at the Sedgwick recharge site are drilled is generally unconfined (fig. 6), as compared to the Halstead recharge site, where the shallow part of the aquifer is confined by a clay layer (fig. 4). The median dissolved-oxygen concentration in water from the shallow wells at the Halstead recharge site is smaller than the median dissolved- oxygen concentration in water from the shallow wells at the Sedgwick recharge site (table 4), indicating that there is more oxygen available at the Sedgwick recharge site to oxidize the sulfide minerals in the aquifer material. There are other possibilities, however, for the larger sulfate concentrations in water from shallow wells at the Sedgwick recharge site, such as the dissolution of gypsum.

In addition to the major ions evaluated using Stiff diagrams, trace elements such as iron and manganese also were examined. The oxidation-reduction (redox) potential of water determines whether redox-sensitive chemicals such as iron and manganese remain in solution or are precipitated by the addition of oxygen. Redox potential for iron and manganese in water samples were not determined during the study, but dissolved-oxygen concentrations in the ground water from the Halstead diversion well site and the Halstead recharge site were small. It is, therefore, unlikely that redox conditions will chemically precipitate iron and manganese and cause plugging of the aquifer material when source water is injected through the recharge well. However, when the source water is exposed to atmospheric oxygen, precipitates can form and may cause plugging of the recharge basins and trench as happened at the Halstead recharge trench. Surface water from the Little Arkansas River has large concentrations of dissolved oxygen that, when introduced into the ground water at the Sedgwick recharge site, could cause chemical precipitation of iron and manganese that could plug the aquifer material. As noted previously, large dissolved-oxygen concentrations in water from the shallow monitoring wells at the Sedgwick recharge site, during both baseline and artificial recharge conditions, are consistent with unconfined conditions. Concentrations of iron and manganese are significantly smaller after recharge (table 7), indicating that precipitation of iron and manganese may have occurred as a result of the addition of highly oxidized

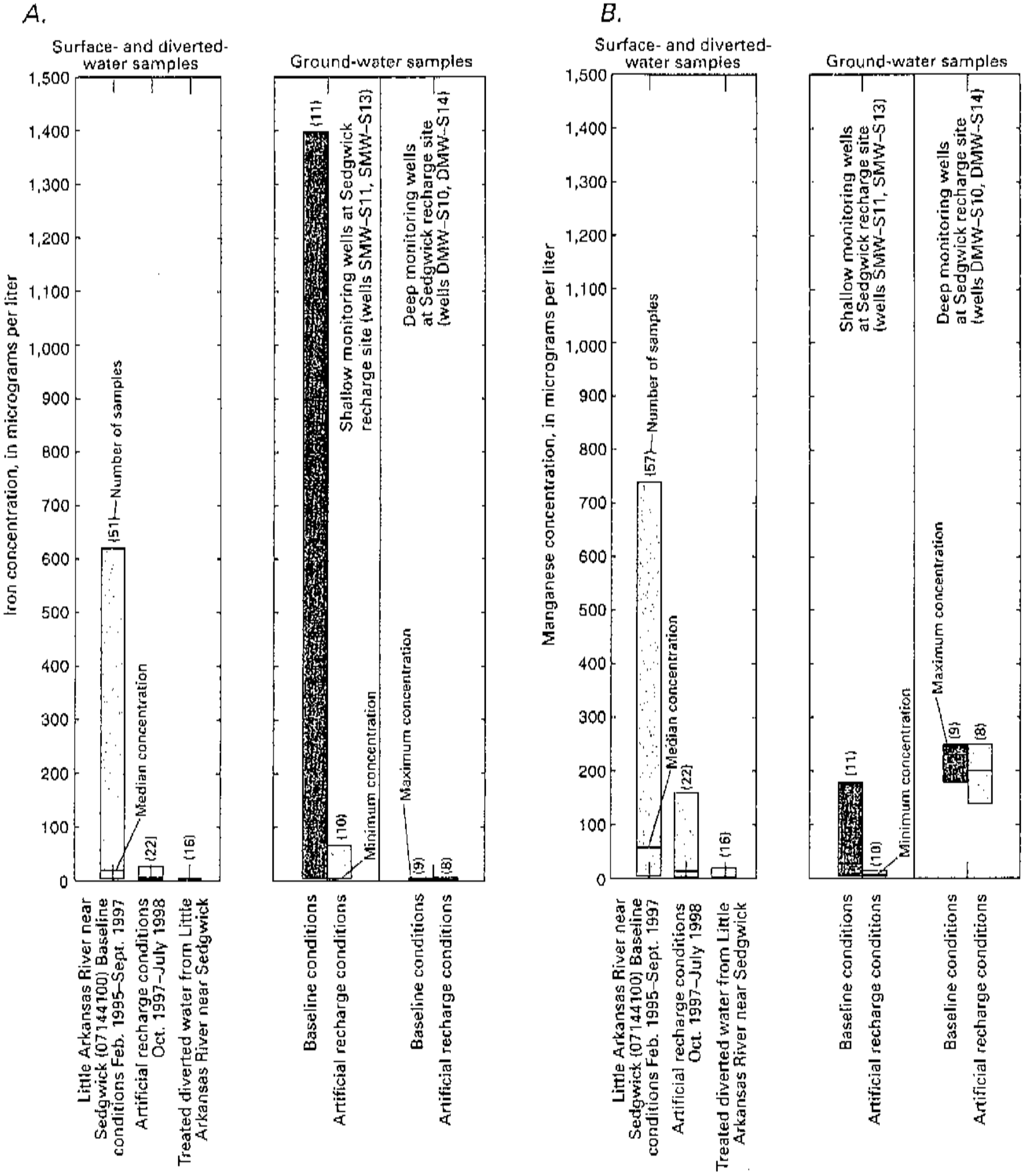


Figure 26. Ranges in *(A)* iron and *(B)* manganese concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Secondary Maximum Contaminant Levels from U.S. Environmental Protection Agency (1999).

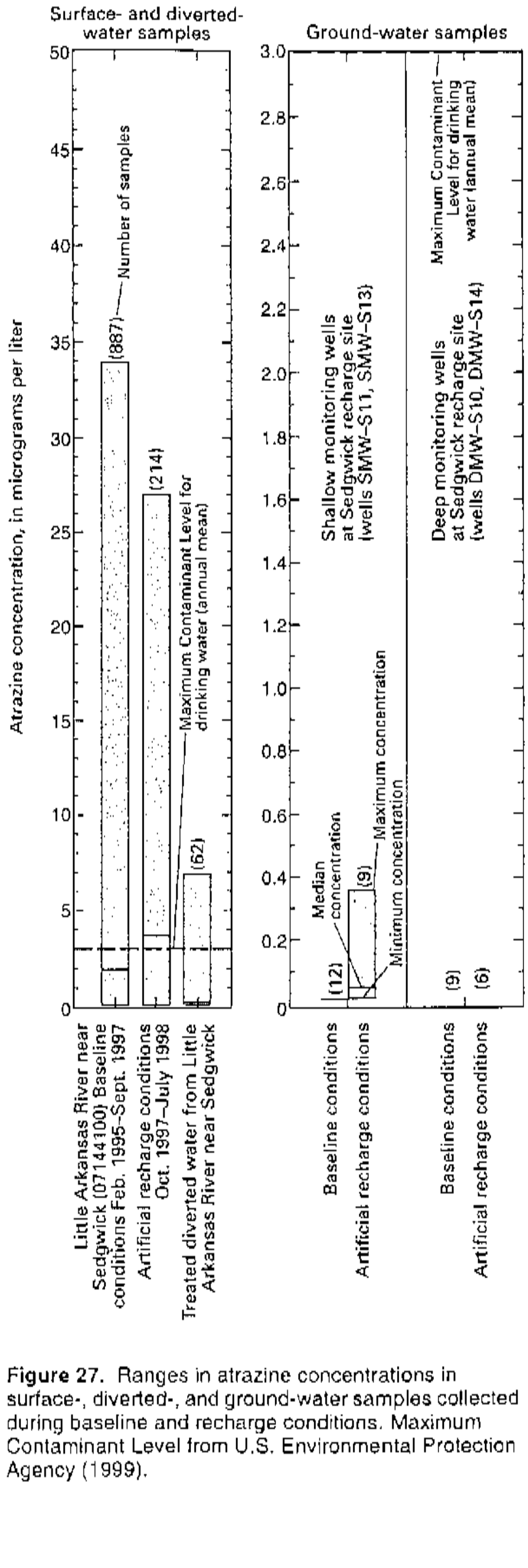


Figure 27. Ranges in atrazine concentrations in surface-, diverted-, and ground-water samples collected during baseline and recharge conditions. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

treated surface water from the Little Arkansas River near Sedgwick. There has been no decrease, however, in the rapid infiltration of treated source water into the *Equus* Beds aquifer through the recharge basins at the Sedgwick recharge site. Therefore, substantial plugging of the aquifer material due to chemical precipitation of iron and manganese at this site appears unlikely.

Physical properties of water, such as turbidity and temperature, also may affect recharge activities by contributing to the plugging of aquifer materials with sediment and dissolved minerals. Turbidity and water temperature of source and ground water at the Halstead diversion well and recharge sites are very similar and have minimal effect on aquifer plugging. At the Sedgwick site, however, even with sediment in the treated source water removed, turbidities and sediment in the treated source water are larger than those of the receiving ground water and could eventually cause plugging of the aquifer material. The primary effect of turbidity differences will be reduced infiltration at the Sedgwick site. Temperature differences in source and receiving ground water at the Sedgwick site can be large (greater than 10 °C), could result in increased chemical rates of reaction causing some dissolution of minerals from the aquifer material or precipitation of minerals from the source water, and may affect permeability.

CONSTITUENTS OF CONCERN FOR FUTURE MONITORING

It was determined from the baseline water-quality monitoring that constituents of concern for artificial recharge activity were sodium, chloride, nitrite plus nitrate, iron, manganese, total coliform bacteria, and atrazine. Sodium and chloride were a concern as related to monitoring frequency and future recharge operations because of the possibility of contamination of the *Equus* Beds aquifer both from source water with large sodium and chloride concentrations and from natural and anthropogenic sources west and northwest of the Wichita well field. Nitrite plus nitrate is a concern mainly because of its effect on human health but also because it is subject to large seasonal variability; the largest baseline concentrations of nitrite plus nitrate occurred during periods when recharge operations were likely. Iron and manganese were a concern

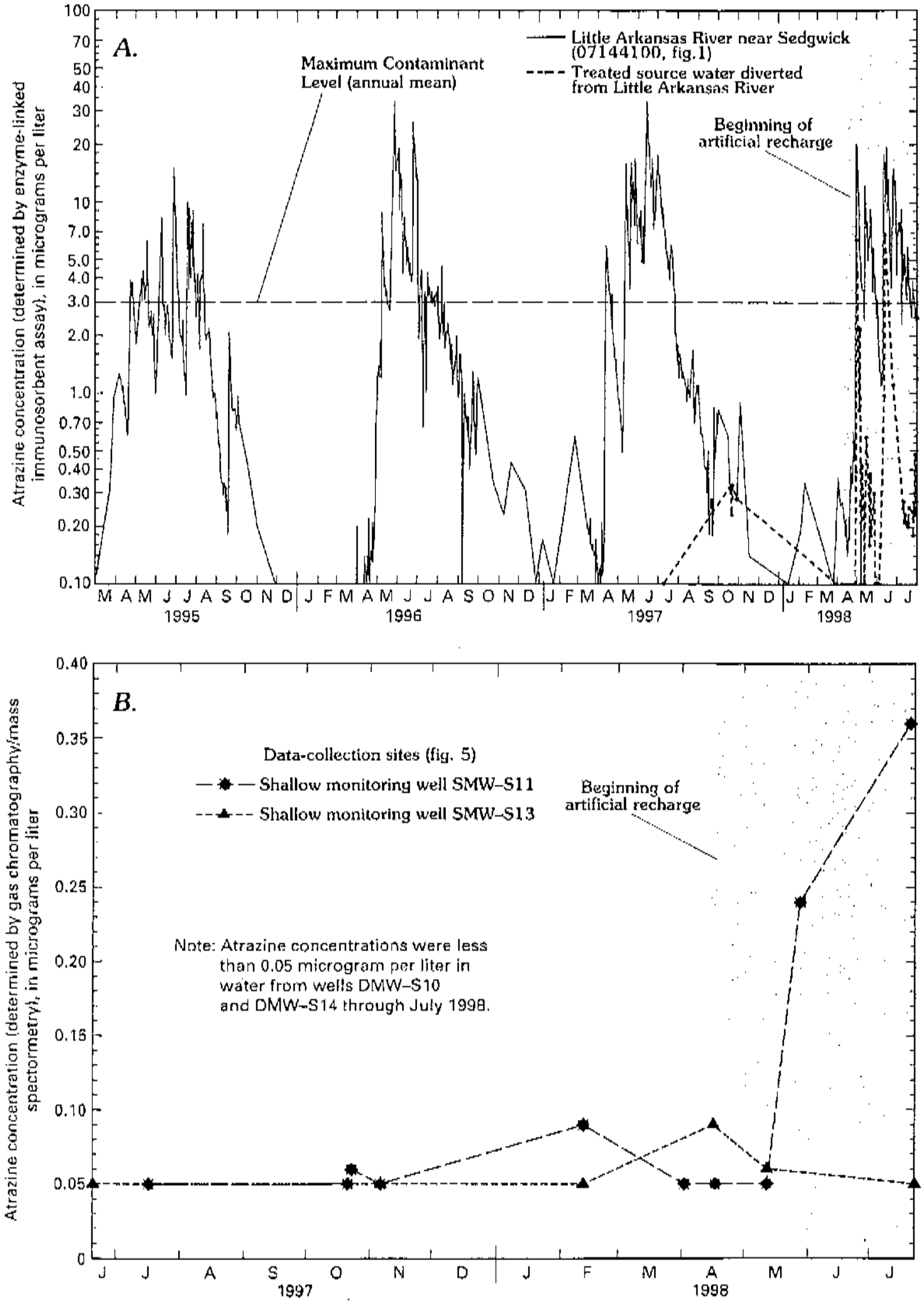


Figure 28. Comparison of atrazine concentrations in *(A)* surface water and treated source water from Little Arkansas River, March 1995–July 1998, and in *(B)* water from monitoring wells at Sedgwick recharge site, June 1997–July 1998. Maximum Contaminant Level from U.S. Environmental Protection Agency (1999).

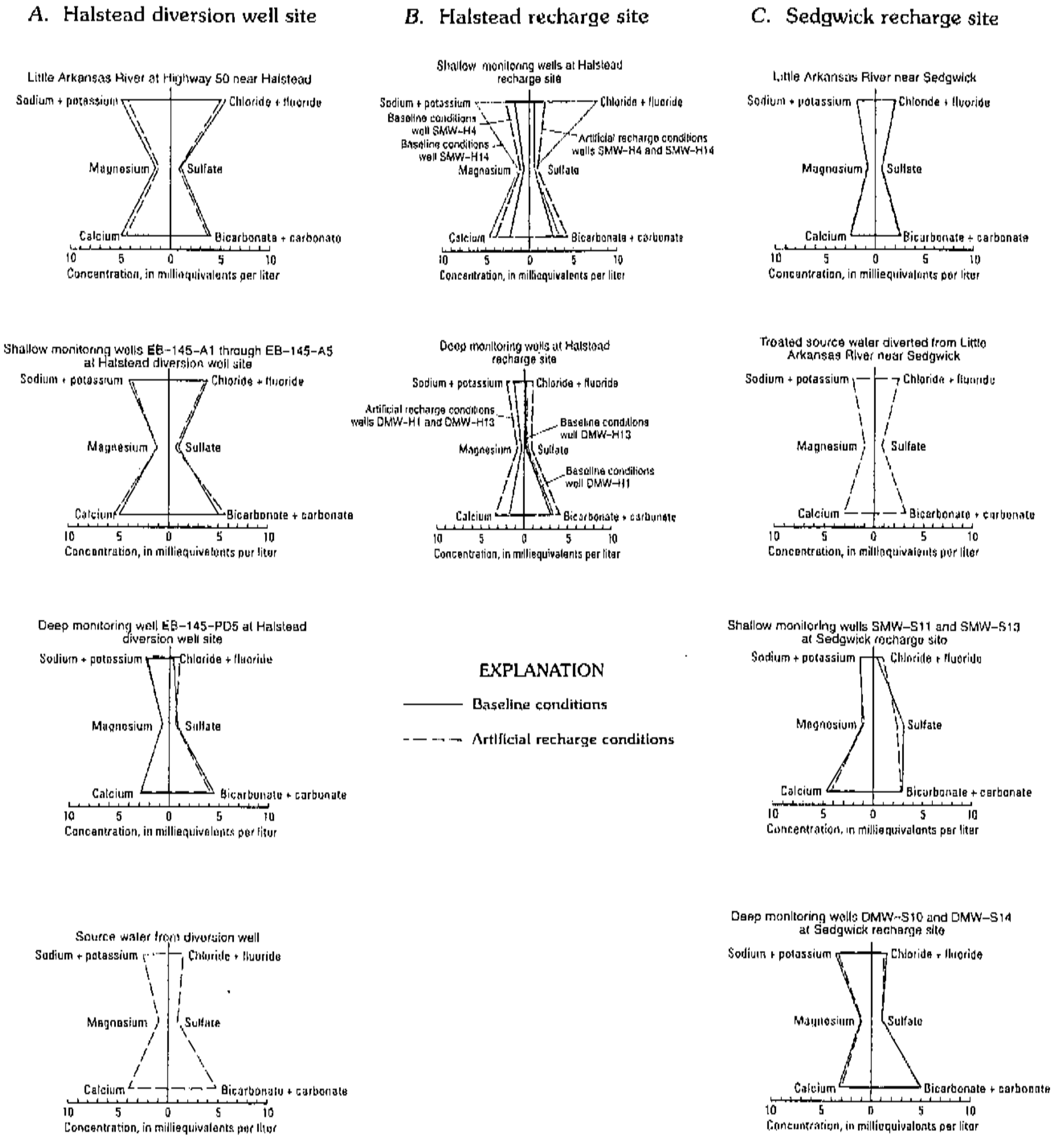


Figure 29. Mean concentrations of major ions in surface and ground water during baseline and recharge conditions at *(A)* Halstead diversion well site, *(B)* Halstead recharge site, and *(C)* Sedgwick recharge site. Location of data-collection sites shown in figures 1 and 2.

because of their tendency to precipitate and thus interfere with the recharge process. Total coliform bacteria were a concern because of large baseline detections in surface water. Atrazine was a concern because of its frequent detection above its MCL and the occurrence of large concentrations during times when recharge was likely. Alachlor, cyanazine, metolachlor, and propazine were also a concern because of occasional detections that exceeded 20 percent of their respective MCL or HAL.

The constituents of concern were monitored during recharge activities on a more frequent basis than the other constituents listed in tables 4–10. However, monitoring of other inorganic and organic constituents was continued to determine an adequate baseline for concentrations of these constituents. It was determined from baseline monitoring that chloride and atrazine concentrations frequently exceeded regulatory criteria in the surface water that was used as a source for recharge. Chloride and atrazine were monitored frequently because of their large and variable concentrations in surface water and because they can be used to indicate when recharge activities need adjustment in treatment or need to be discontinued. Monitoring frequency is also an important consideration in future recharge operations as seasonal definition of constituent concentrations is necessary to describe effects of future recharge.

Monitoring for the constituents of concern in source water was conducted at least every 5 days immediately after recharge began at the Halstead and Sedgwick sites. The data collected at the Halstead recharge site (figs. 19 and 20) illustrate that changes in quality of source and receiving ground water are very gradual and probably could be defined adequately with monthly samples of source water and quarterly samples of receiving ground water. However, at the Sedgwick site, monitoring of selected constituents—primarily chloride, bacteria, and atrazine—would benefit from more frequent sampling because of the large variability in these constituent concentrations. Increased monitoring frequency in the shallow monitoring wells would improve definition of the effects of source water on shallow ground water. Quarterly sampling probably is sufficient for deep ground water at the Sedgwick site because recharge activities seem to have little effect on water quality in the deep monitoring wells and because quarterly sampling is sufficient to define seasonal variability.

Real-time monitoring of source water potentially can improve the effectiveness of the current monitoring program for the *Equus* Beds Ground-Water Recharge Demonstration Project and is important to the maintenance of good quality water in the *Equus* Beds aquifer. Real-time data are recorded hourly and transmitted every 4 hours to the USGS office in Lawrence, Kansas, and displayed on the Internet at http://ks.water.usgs.gov. With real-time monitoring, an undesirable level of a constituent in source water can be identified almost immediately and action taken to either treat the water before recharge or the decision could be made not to recharge until water-quality conditions improved.

To achieve real-time water-quality monitoring, it would be necessary to use surrogates for the constituents of primary concern, chloride and atrazine. A surrogate is a physical property or properties that are monitored continually in-stream that may be substituted for a particular water-quality constituent for which continual data are not available. For example, specific conductance, which is currently being monitored in real time for source water, may be determined by an in-stream probe and could be used as a surrogate for the analysis of chloride concentrations in source water. A comparison of specific conductance and chloride concentrations in water from the Little Arkansas River at Highway 50 near Halstead shows a direct relation with a correlation coefficient (r^2) of 0.92 (fig. 30*A*). Source water from the Halstead diversion well shows a similar relation between specific conductance and chloride (fig. 30*B*).

The slopes of the regression lines for the relations between specific conductance and chloride in water from the Sedgwick recharge site are different than those for water from the Halstead recharge site. A comparison of specific conductance and chloride in water from the Sedgwick site shows a slope of 0.14 for both the surface water and treated source water (figs. 30*C* and 30*D*). The slopes of the relations for the Halstead site are 0.19 for water diverted from the Little Arkansas River at Highway 50 near Halstead and 0.13 for water from the Halstead diversion well. The correlation coefficients are also slightly different between the two sites. The surface water and treated source water for the Sedgwick site have correlation coefficients of 0.85 and 0.83, respectively. The surface water and diversion well water for the Halstead sites show slightly better correlation coefficients of 0.92 and 0.91, respectively. The difference in the relation between the Halstead site and the Sedgwick site indicates that the computation of chloride concentrations, made on the basis of measured values of its surrogate (specific conductance), would need to be site specific and that the accuracy of the computed values also would need to be site specific.

The ELISA screen for triazine herbicides could be used as a surrogate for atrazine to provide real-time monitoring of source water. There is about a 2-day turnaround for ELISA analysis compared with about 40 days for GC/MS analysis. In figure 31, 191 pairs of water samples from the Halstead and Sedgwick

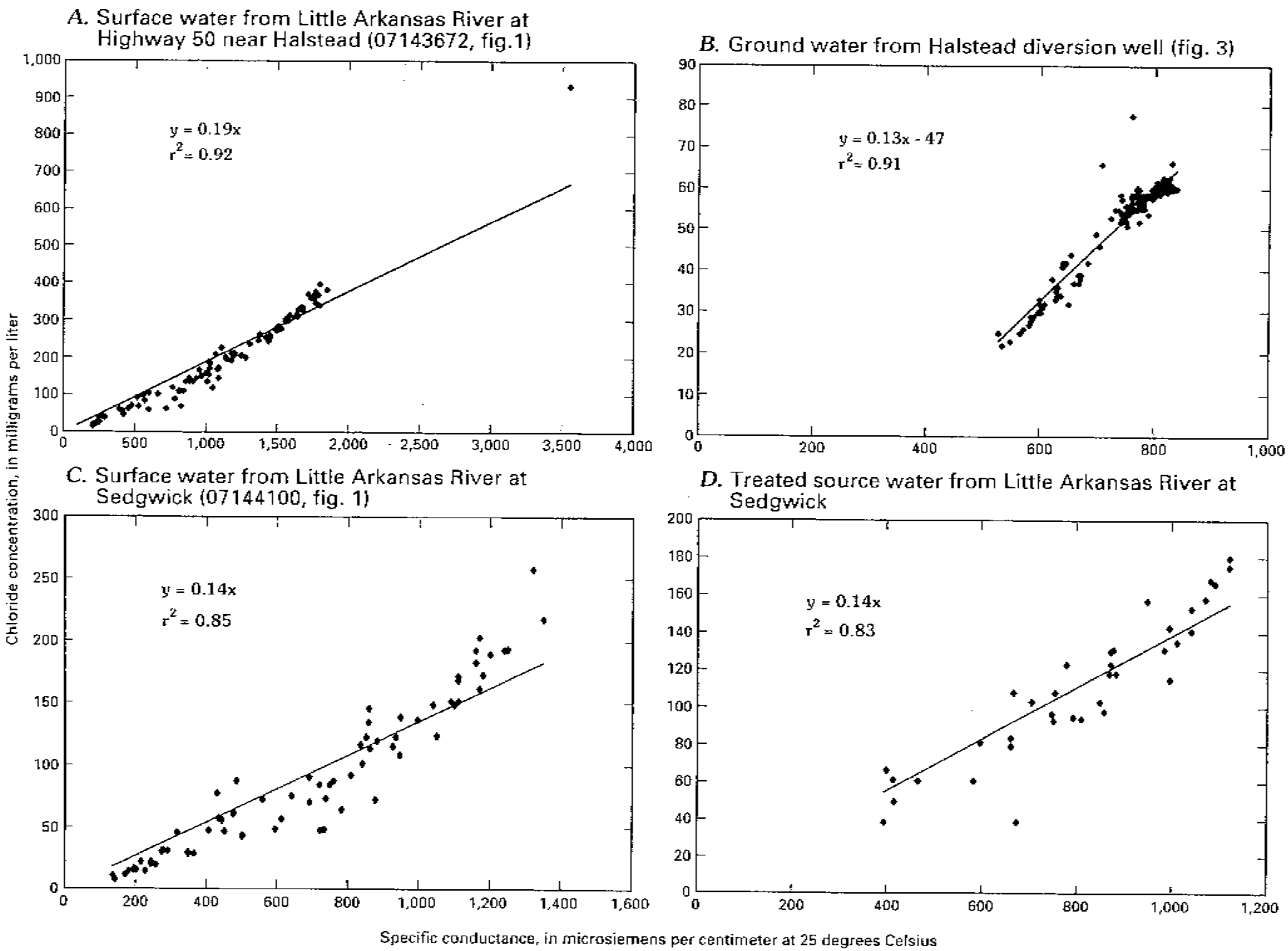


Figure 30. Relations between specific conductance and chloride concentrations in *(A)* surface water from Little Arkansas River at Highway 50 near Halstead, *(B)* ground water from Halstead diversion well, and *(C)* surface water and *(D)* treated source water from Little Arkansas River at Sedgwick, March 1995–July 1998.

surface-water monitoring sites (stations 07143672 and 07144100, fig. 1) were analyzed for triazine herbicides by ELISA and for atrazine by GC/MS. The relation between triazine-herbicide concentrations determined by ELISA and atrazine concentrations determined by GC/MS has a correlation coefficient of r^2=0.85. Not only are the results of the two procedures similar, but the ELISA analysis would allow for many samples to be analyzed at low cost.

The use of surrogates enables real-time water-quality monitoring for the constituents of primary concern. This would allow project officials to take appropriate action if the quality of the surface water changes substantially or if SMCLs or MCLs are exceeded.

SUMMARY AND CONCLUSIONS

One of the primary sources of water for the city of Wichita in south-central Kansas is the Wichita well field, completed in the *Equus* Beds aquifer. The *Equus* Beds Ground-Water Recharge Demonstration Project was begun in 1995. The project was designed to investigate the feasibility of artificially recharging the *Equus* Beds aquifer to meet increased demand for water supplies and to protect this important aquifer from saltwater intrusion from natural and anthropogenic sources to the west and northwest. An evaluation of the preliminary effects of artificial recharge on water quality will, in part, determine if a full-scale recharge project is feasible.

The project is a cooperative effort between the city of Wichita and the Bureau of Reclamation, U.S. Department of the Interior. During the project, high flows from the Little Arkansas River are captured and recharged into the *Equus* Beds aquifer through recharge basins, a trench, or a recharge well, located at two recharge sites near Halstead and Sedgwick, Kansas. To document baseline concentrations and compatibility of stream (recharge) and aquifer water, the U.S. Geological Survey collected water samples from February 1995 through August 1998. These samples were analyzed for dissolved solids, total and dissolved inorganic constituents, nutrients, organic and volatile organic compounds, radionuclides, and bacteria.

Determination of baseline water-quality conditions indicated that the constituents of concern were sodium, chloride, nitrite plus nitrate, iron and manganese, total coliform bacteria, and atrazine. Chloride and atrazine were of particular concern because concentrations of these constituents in surface water from the Little Arkansas River frequently exceeded regulatory criteria. The Little Arkansas River is used as the source water for recharge. The U.S. Environmental Protection Agency Secondary Maximum Contaminant Level for chloride is 250 mg/L, and the Maximum Contaminant Level for atrazine is 3.0 µg/L as an annual mean. Baseline concentrations of chloride in surface water ranged from 8.0 to 400 mg/L. Baseline concentrations of atrazine in surface water ranged from less than 0.10 to 46 µg/L. Chloride and atrazine concentrations were large and variable in surface water when compared with the receiving ground water.

From May 1997 through July 1998, a total of about 338 Mgal of water were artificially recharged at the sites near Halstead and Sedgwick, Kansas. At the Halstead recharge site, some increases in concentrations of chloride and atrazine in water from deep monitoring wells were evident after recharge began even though concentrations remained considerably less than the respective SMCL and MCL established by the U.S. Environmental Protection Agency for drinking water. At the Sedgwick recharge site, chloride concentrations decreased after recharge began in water from one of two deep monitoring wells. In water from the other deep well, concentrations of chloride remained unchanged. Atrazine concentrations increased in water from shallow monitoring wells at the Sedgwick site after recharge began. In water samples from deep wells, atrazine concentrations remained less than the MCL or were not detected during artificial recharge conditions. Not enough water has been recharged at the Sedgwick site to date (1999) to determine the overall effects of artificial recharge on receiving ground-water quality. Continued monitoring is necessary to determine long-term effects of artificial recharge at both sites.

Major-ion and trace-element concentrations in source water and receiving ground water were determined to assess the compatibility of the water for artificial recharge. Stiff diagrams of major ions were used to show the similarity or differences in water chemistry between the source water and receiving ground water. Water from both sources were chemically compatible to the receiving aquifer water at both recharge sites. In addition, trace elements were examined along with dissolved-oxygen concentrations to determine whether redox-sensitive chemical constituents would remain in solution or precipitate once source water

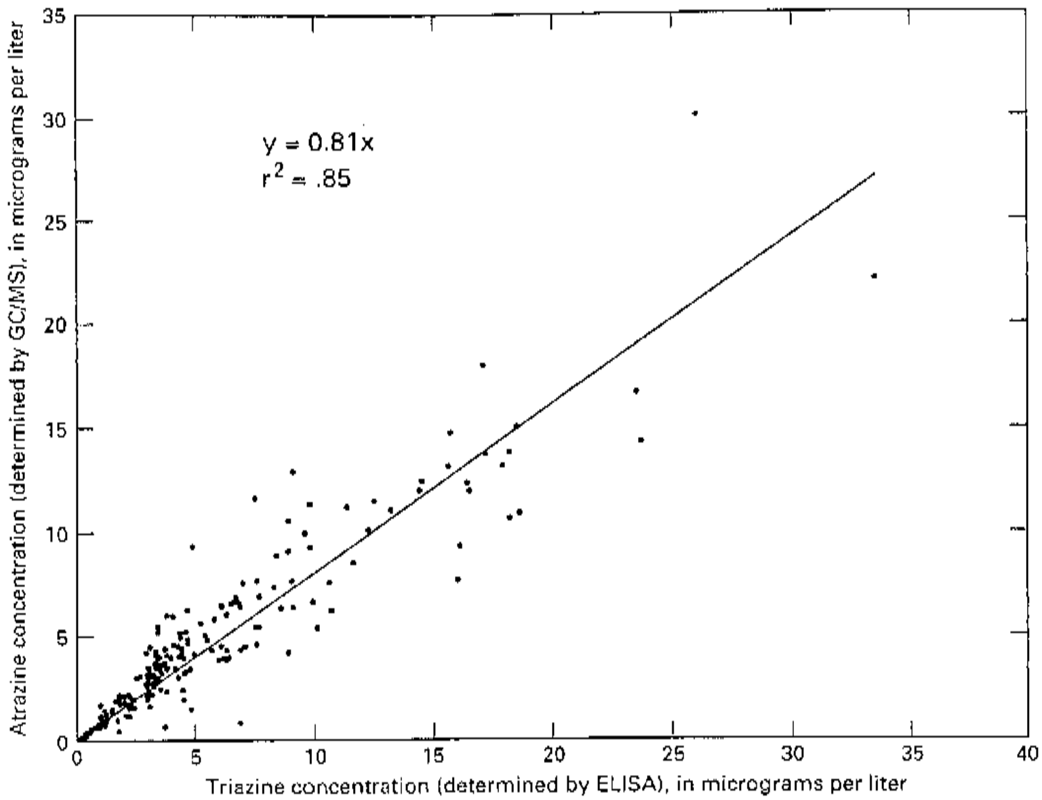


Figure 31. Relation between triazine herbicide concentrations determined by enzyme-linked immunosorbent assay (ELISA) and atrazine concentrations determined by gas chromatography/mass spectrometry (GC/MS) for samples collected from Halstead and Sedgwick surface-water monitoring sites, March 1995–July 1998.

was introduced into the *Equus* Beds aquifer. Major-ion and trace-element concentrations in the source water and receiving water at both recharge sites were similar and probably would not cause detrimental plugging of aquifer materials with the exception of possible iron and manganese precipitation at the Halstead site when source water is exposed to oxygen.

It may be possible to decrease monitoring frequency at the Halstead site because water-quality changes in receiving ground water at this site are very gradual. However, more information is needed at the Sedgwick site. Real-time water-quality monitoring could improve the effectiveness of the current monitoring program used by the *Equus* Beds Ground-Water Recharge Demonstration Project. The use of surrogates for the determination of chloride and atrazine concentrations in source water needs to be site specific and provide a more timely picture of the water quality, thus enabling project officials to alter treatment of water more effectively or to stop artificial recharge until water-quality conditions improve.

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EXHIBIT G

Prepared in cooperation with the
CITY OF WICHITA, KANSAS

Effects of Artificial Recharge on Water Quality in the *Equus* Beds Aquifer, South-Central Kansas, 1995–2000

—*Andrew C. Ziegler, Heather C. Ross, Thomas J. Trombley, and Victoria G. Christensen*

This fact sheet describes the effects of artificial recharge on the Equus Beds aquifer for concentrations of four chemicals of concern—chloride, arsenic, total coliform bacteria, and atrazine. These four chemicals were determined to be of the greatest concern after determining the effects of recharge by comparing the median concentrations of more than 400 chemicals in more than 4,000 samples collected before and after artificial recharge activities began.

Introduction

The water supply for the city of Wichita, south-central Kansas, currently comes from the Wichita well field and Cheney Reservoir (fig. 1). Because these sources are not expected to meet projected city water needs into the 21st century (Warren and others, 1995), artificial recharge of the *Equus* Beds aquifer is being investigated as one alternative to meet future water-supply demands. An additional potential benefit of artificial recharge includes preventing degradation of the water quality of the aquifer by chloride plumes from the Arkansas River to the southwest and the Burrton oil field to the northwest (Ziegler and others, 1999).

In 1995, the *Equus* Beds Ground-Water Recharge Demonstration Project began evaluation of artificial recharge techniques and their effects on water quality in the aquifer. The demonstration project is a cooperative effort among the city of Wichita, Bureau of Reclamation (U.S. Department of the Interior), and the U.S. Geologicial Survey (USGS).

Water from the Little Arkansas River is diverted for artificial recharge when flow in the river exceeds base flow in accordance with the Kansas Department of Agriculture, Division of Water Resources, permit conditions (Burns and McDonnell, 1998). Water is artificially recharged to the *Equus* Beds aquifer, which is part of the High Plains aquifer and consists of alluvial (river-deposited) sediments of sand and gravel interbedded with clay and silt.

At the Halstead diversion well site (fig. 1), water is diverted from the Little Arkansas River by pumping a diversion well completed immediately adjacent to the river that induces the surface water into the well. This diverted source water then is pumped to the Halstead recharge site and recharged to the aquifer by basin, trench, or well. Recharge of the *Equus* Beds aquifer at the Halstead site began in May 1997.

Recharge water for the Sedgwick recharge site is diverted directly from the Little Arkansas River. It is treated to reduce turbidity (the cloudy appearance of water caused by suspended matter) and to remove organic compounds, including the herbicide atrazine, using powder activated carbon (PAC). The diverted water is recharged to the *Equus* Beds aquifer at the Sedgwick site through recharge basins (Ziegler and others, 1999). Recharge of the Equus Beds aquifer at the Sedgwick site began in April 1998.

From 1995–2000, the USGS monitored water-quality conditions during all aspects of the artificial

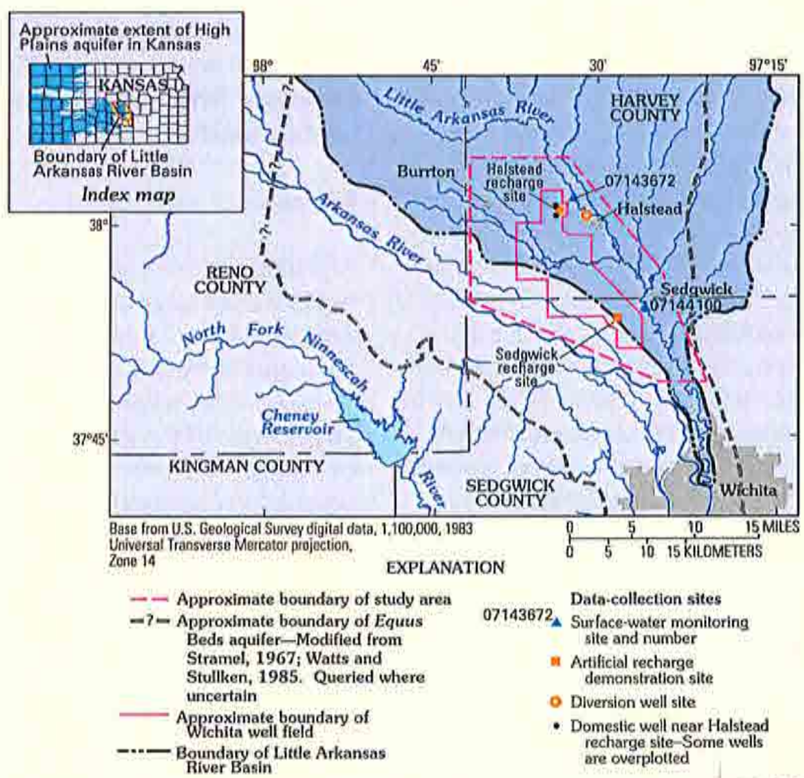
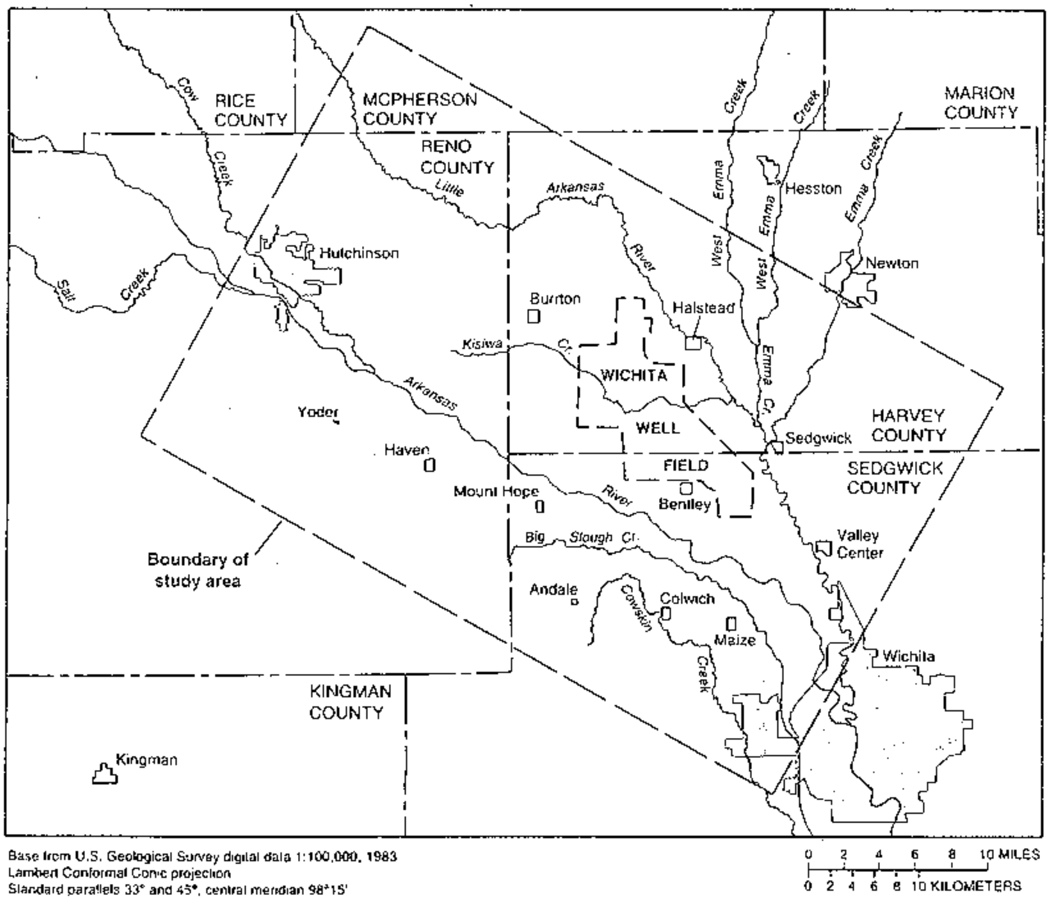


Figure 1. Location of study area near Wichita, south-central Kansas.

EXHIBIT H

Hydrologic and Chemical Interaction of the Arkansas River and the *Equus* Beds Aquifer Between Hutchinson and Wichita, South-Central Kansas

U.S. GEOLOGICAL SURVEY
Water-Resources Investigations Report 95–4191



Prepared in cooperation with the
KANSAS WATER OFFICE,
EQUUS BEDS GROUNDWATER MANAGEMENT DISTRICT NO. 2,
and the BUREAU OF RECLAMATION, U.S. DEPARTMENT OF THE INTERIOR

002007

Hydrologic and Chemical Interaction of the Arkansas River and the *Equus* Beds Aquifer Between Hutchinson and Wichita, South-Central Kansas

By N.C. MYERS, G.D. HARGADINE, and J.B. GILLESPIE

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CONTENTS

Definition of Terms VII
Abstract 1
Introduction 2
  Background 2
  Purpose and Scope 2
  Description of Study Area 2
  Site- and Well-Numbering System 5
  Methods of Investigation 7
  Previous Studies 9
  Acknowledgments 9
Geology 9
Hydrology 10
  Surface Water 10
  Ground Water 14
  Hydraulic Relation Between Rivers and Aquifer 16
  Water Use 16
Chloride in Water 17
  Sources of Chloride 18
  Chloride in Surface Water 19
  Chloride in Ground Water 21
Conceptual Model of the *Equus* Beds Aquifer 22
  Boundaries 22
  Aquifer Properties 23
  Recharge 23
  Discharge 27
Simulation of Stream-Aquifer Interaction 27
  Steady-State Ground-Water Flow Model 28
    Model Geometry 28
    Aquifer Properties 28
    Stresses 29
    Calibration of Steady-State Model 34
    Sensitivity Analysis 35
  Transient Ground-Water Flow Model 36
    Aquifer Properties and Stresses 36
    Calibration of Transient Model and Sensitivity Analysis 38
  Water Budgets 41
  Parameters for Simulations of Hypothetical Conditions 42
  Discussion of Ground-Water Flow-Model Results 46
    Steady-State and Transient Simulatious, 1940–89 46
    Transient Simulations, 1990–2019 47
  Particle Tracking 56
    Discussion of Particle-Tracking Results, 1940–89 66
    Discussion of Particle-Tracking Results, 1990–2019 70
Summary 71
References Cited 76
Supplemental Information 82

PLATES

[Plates are in pocket]

1. Hydrogeologic sections *A–A'* through *D–D'* in the vicinity of the Arkansas River between Hutchinson and Wichita, south-central Kansas
2. Hydrogeologic sections *E–E'* through *G–G'* in the vicinity of the Arkansas River between Hutchinson and Wichita, south-central Kansas

FIGURES

1–6. Maps showing:
- 1. Location of stndy area 3
- 2. Location of study area, streams, Wichita well field, cities, and towns 4
- 3. Areal geology, traces of hydrogeologic sections, and section names 5
- 4. Modified Bureau of Land Management land subdivision system used for numbering well or site locations 6
- 5. Location of observation-well clusters 7
- 6. Location of streamflow-gaging stations, surface-water sampling sites, seepage-survey sites, pumping-test sites, and wells for which hydrographs were used to compare measured and simulated hydraulic heads 8

7. Generalized geologic section showing present and projected past extent of the Hutchinson Salt Member 11

8. Map showing altitude of bedrock surface 12

9–12. Graphs showing:
- 9. Measured streamflow in the Arkansas River during seepage surveys 14
- 10. Measured streamflow in the Little Arkansas River during seepage surveys 15
- 11. Monthly mean streamflow for the Arkansas River near Hutchinson, near Maize, and at Wichita, and difference between streamflow near Maize and near Hutchinson 16
- 12. Monthly mean streamflow for the Little Arkansas River at Alta Mills and at Valley Center, and difference between streamflow at Valley Center and at Alta Mills 17

13. Map showing potentiometric surface in the *Equus* beds aquifer, 1940 18

14. Map showing potentiometric surface iu the *Equus* beds aquifer, 1989 19

15–18. Graphs showiug:
- 15. Daily mean river-stage altitude of the Arkansas River near Hutchinson, daily mean water-level altitude in nearby well EB-216-AA, and daily precipitation rate at streamflow-gaging station, August 1989–September 1991 20
- 16. Monthly instantaneous ground-water levels in wells EB-216-AA, EB-216-A, EB-216-B, and EB-216-C and the daily mean river stage of the Arkansas River near Hutchinson for those days when ground-water levels were measured, April 1988–May 1990 21
- 17. Industrial, municipal, agricultural, and total pumpage from the *Equus* beds aquifer in study area for 1940–89 22
- 18. Streamflow aud chloride concentrations in the Arkansas River near Hutchinson, August 1988–July 1991, and in the Arkansas River near Maize, November 1988–July 1991, and chloride loads in the Arkansas River near Hutchinson, August 1988–July 1991, and the Arkansas River near Maize, November 1988–July 1991 24

19–23. Maps showing:
- 19. Natural chloride concentrations and lines of equal conceutration in upper, middle, and lower units of the *Equus* beds aquifer, 1989–90 25
- 20. Model grid, row and column numbers, and boundary conditions for upper, middle, and lower model layers 29
- 21. Distribution of horizontal hydraulic conductivity in upper, middle, and lower model layers 32
- 22. Ground-water recharge rates for steady-state model 35
- 23. Measured and simnlated potentiometric surfaces for middle model layer, 1940, and areas used for comparison of measured and simulated hydraulic heads 37

24. Graphs showing mean absolute deviations of simulated hydraulic heads from accepted model-calibration heads for changes in evapotranspiration, streambed conductance, streamflow, recharge, and hydraulic conductivity 38

25. Maps showing model cells where pumpage from any of the three model layers was simulated in steady-state model and in transient model for 1940–52, 1953–58, 1959–63, 1964–70, 1971–79, and 1980–89 39

26. Graph showing simulated pumpage used in transient model for each model layer 46

27. Map showing measured and simulated potentiometric surface for 1989 in the middle model layer and areas used for comparison of measured and simulated hydraulic heads 47

28. Graphs showing measured and simulated water-level altitudes for transieut simulation, 1940–89 48

FIGURES—Continued

29–38. Graphs showing:
29. Measured annual average streamflow exceeded 70 percent of time and model-simulated base flow at gaging stations on the Arkansas River near Hutchinson, Arkansas River near Maize, Little Arkansas River at Alta Mills, and Little Arkansas River at Valley Center, 1940–90 50
30. Mean absolute deviation of simulated hydraulic heads from accepted transient-model calibration heads for changes in hydraulic conductivity, recharge, streamflow, storage coefficient, and streambed conductance 51
31. Pumpage specified in model-input data sets and actual simulated pumpage for two simulations of hypothetical conditions 54
32. Simulated cumulative net streamflow gain from or loss to the *Equus* beds aquifer at end of each stress period for the Arkansas River and Little Arkansas River 55
33. Estimated cumulative chloride discharge from Arkansas River to *Equus* beds aquifer at end of each stress period based on simulated streamflow loss from the Arkansas River and assuming a chloride concentration of 630 milligrams per liter 56
34. Estimated water levels at model row-5, column-24 in Wichita well field for simulations assuming no increase in pumpage since 1989, a 1-percent per year increase in pumpage since 1989, a 2-percent per year increase in pumpage since 1989, and a 3-percent per year increase in pumpage since 1989 57
35. Estimated water levels at model row-24, column-23 near Arkansas River for simulations assuming no increase in pumpage since 1989, a 1-percent per year increase in pumpage since 1989, a 2-percent per year increase in pumpage since 1989, and a 3-percent per year increase in pumpage since 1989 59
36. Estimated cumulative net streamflow loss of water from Arkansas River to *Equus* beds aquifer at end of each stress period assuming no increase in pumpage since 1989, a 1-percent per year increase in pumpage since 1989, a 2-percent per year increase in pumpage since 1989, and a 3-percent per year increase in pumpage since 1989 61
37. Estimated cumulative chloride discharge from Arkansas River to *Equus* beds aquifer at end of each stress period assuming a chloride concentration of 630 milligrams per liter and no increase in pumpage since 1989, a 1-percent per year increase in pumpage since 1989, a 2-percent per year increase in pumpage since 1989, and a 3-percent per year increase in pumpage since 1989 63
38. Simulated rate of storage change for stress periods 1–6 and hypothetical stress periods 1–3 66
39. Maps showing simulated distribution of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers 67
40. Maps showing simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming no increase in pumpage since 1989, a 1-percent per year increase in pumpage since 1989, a 2-percent per year increase in pumpage since 1989, and a 3-percent per year increase in pumpage since 1989 70

TABLES

1. Seepage-survey data available for the Arkansas and Little Arkansas Rivers 13
2. Summary of chloride-concentration data collected from Cow Creek and the Arkansas River, August 1988–July 1991 23
3. Simnlated streamflows, for the starting reach of each stream, used in steady-state and transient models 36
4. Simulated streamflow at upstream and downstream ends of streams within steady-state model 38
5. Mean annual precipitation at Hutchinson, Mount Hope, and Wichita climatic stations and for the study area for steady-state and transient model stress periods 39
6. Simulated steady-state and transient ground-water budgets 52
7. Lithologic logs of deep wells at well sites EB-201 through EB-249 82

CONVERSION FACTORS, ABBREVIATIONS, AND VERTICAL DATUM

Multiply	By	To obtain
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
square foot (ft^2)	0.09290	square meter
acre	4,047	square meter
inch per hour (in/hr)	25.4	millimeter per hour
inch per year (in/yr)	25.4	millimeter per year
foot per day (ft/d)	0.3048	meter per day
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year
ton per day (ton/d)	907.2	kilogram per day
ton per day	370.78/(ft^3/s)	milligram per liter (mg/L)
foot squared per day[1] (ft^2/d)	0.09290	meter squared per day
degree Fahrenheit (°F)	°C = (°F-32)5/9	degree Celsius (°C)

[1]The standard unit for transmissivity (T) is cubic foot per day per square foot times foot of aquifer thickness $[(ft^3/d)/ft^2]ft$. To avoid confusion to the nontechnical reader, the mathematical expression has been reduced to foot squared per day (ft^2/d).

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

DEFINITION OF TERMS

Aquifer. A geologic formation, group of formations, or part of a formation that contains sufficient saturated permeable material to yield significant quantities of water to wells or springs.

Equipotential line. A line in a two-dimensional ground-water flow field such that the total hydraulic head is the same for all points along the line.

Evapotranspiration. Water withdrawn from a land area by evaporation from water surfaces and moist soil, and by plant transpiration.

Hydraulic conductivity. The volume of water at the existing kinematic viscosity that will move in unit time under a unit hydraulic gradient through a unit area measured at right angles to the direction of flow. Units of hydraulic conductivity are:

$$\frac{(\text{length}^3/\text{time})}{(\text{length}^2)\ (\text{length/length})}\left(\text{for example, } \frac{(\text{feet}^3/\text{day})}{(\text{feet}^2)\ (\text{feet/feet})}\right)$$

but, as in this report, are commonly simplified and reported as length/time (for example, feet per day).

Hydraulic gradient. Change in total hydraulic head per unit of distauce iu a given direction.

Hydraulic head. Height above a standard datum of the surface of a water column that can be supported by the static pressure at a given point.

Phreatophyte. A plant that can live with its roots in and obtains water from the saturated zone.

Porosity. The ratio of the volume of void spaces in sediment or rock to its total volume.

Potentiometric surface A surface that represents the level to which water will rise in a tightly cased well. More than one potentiometric surface may be required to describe the distribution of hydraulic head if hydraulic head varies appreciably with depth in the aquifer.

Recharge. The processes involved in the addition of water to the zone of saturation.

Saturated thickness. The thickness of the saturated zone in an aquifer.

Specific storage. The volume of water released from or taken into ground-water storage per unit volume of the porons medium per unit change in hydraulic head.

Specific yield. The ratio of the volume of water that saturated rock or sediment will yield by gravity to the volume of the rock or sediment.

Steady state. Condition under which the magnitude and direction of ground-water flow velocities are constant with time.

Storage coefficient. The volume of water an aquifer releases from or takes into storage per unit surface area of the aquifer per unit change in head.

Subcrnp. The areal extent of a truncated rock unit at a buried surface of unconformity. The subcrop would be the surface outcrop if overlying beds were removed.

Transient. Condition under which the magnitude and direction of ground-water flow velocities vary with time.

Transmissivity. The volume of water of the existing kinematic viscosity that will move in unit time through a unit width of the aquifer under a unit hydraulic gradient. Units of transmissivity are:

$$\frac{(\text{length}^3/\text{time})}{(\text{length})\ (\text{length/length})}\left(\text{for example, } \frac{(\text{feet}^3/\text{day})}{(\text{feet})\ (\text{feet/feet})}\right)$$

but, as in this report, are commonly simplified and reported as length^2/time (for example, feet squared per day).

Water table. That surface in a ground-water body at which the water pressure is atmospheric. It is defined by the levels at which water stands in wells that penetrate the water body just far enough to hold standing water.

Hydrologic and Chemical Interaction of the Arkansas River and the *Equus* Beds Aquifer Between Hutchinson and Wichita, South-Central Kansas

By N.C. Myers, G.D. Hargadine, *and* J.B. Gillespie

Abstract

Large chloride concentrations in Arkansas River water have the potential to degrade water quality in the adjacent *Equus* beds aquifer between Hutchinson and Wichita, Kansas. The aquifer is an important source of water for municipal, industrial, agricultural, and domestic uses.

A three-dimensional, finite-difference, ground-water flow-model program (MODFLOW) was used with data from past studies and data collected during 1988–91 to simulate aquifer and stream conditions during the late 1930's, during 1940–89, and during 1990–2019. Results of ground-water flow-model simulations indicated that declining water levels in the *Equus* beds aquifer since the 1940's have caused base flow in the Arkansas and Little Arkansas Rivers to decrease. In 1940, the Arkansas and Little Arkansas Rivers had simulated net base-flow gains within the model area of about 21 and about 67 ft^3/s (cubic feet per second), respectively. By the end of 1989, the Arkansas River had a simulated net base-flow loss of about 52 ft^3/s, and the Little Arkansas River had a net base-flow gain of about 27 ft^3/s. Simulations for 1990–2019 showed that the water-level changes in a selected model cell located in the central part of the Wichita well field could range from -0.2 to -78 feet. Water-level changes in a selected model cell located near the Arkansas River could range from +1.3 to -1.2 feet. In model simulations where only pumpage varied, net base-flow loss from the Arkansas River to the aquifer ranged from about 59 ft^3/s (no increase in pumpage since 1989) to 117 ft^3/s (a 3-percent per year increase in pumpage since 1989) by 2019.

Assuming a chloride concentration of 630 milligrams per liter, the median concentration in Arkansas River water collected during 1988–91, the quantity of chloride discharged from the Arkansas River to the aquifer was estimated to have increased from about 21 tons per day in 1940 to about 100 tons per day in 1989. By 2019, chloride discharge was indicated to range from about 110 tons per day (associated with no increase in pumpage since 1989) to 200 tons per day (associated with a 3-percent per year increase in pumpage since 1989).

A particle-tracking program (MODPATH), which used the results from the flow model, was used to simulate the distribution in the aquifer of chloride from the river during the same time periods. Particle-tracking simulations show that, during 1940–89, the simulated distribution of particles representing chloride from the Arkansas River expanded from relatively narrow bands near the river to a wider distribution within the aquifer and the Wichita well field. Particle-tracking simulations indicate that chloride discharge from the Arkansas River may have reached the edge of the Wichita well field as early as 1963.

INTRODUCTION

Background

Large chloride concentrations in Arkansas River water have the potential to degrade water quality in the adjacent *Equus* beds aquifer. Since 1940, total ground-water withdrawals from the *Equus* beds aquifer in the study area for municipal, industrial, and agricultural uses have increased from about 15,270 to as much as 138,630 acre-ft/yr in 1988 (Spinazola and others, 1985; data on file with the U.S. Geological Survey, Lawrence, Kans.). Many of the wells near the Arkansas River have poorer water quality than wells farther away from the river. Further development of the aquifer could increase infiltration of Arkansas River water to the aquifer, thereby increasing chloride concentrations in the aquifer and decreasing streamflows in the Arkansas River. An understanding of how the stream-aquifer system responds to various system stresses (such as ground-water withdrawals from wells, drought, large or small river flows) could lead to improved management of the available water resources near the Arkansas River. A 4-year study to evaluate the hydrologic and chemical interaction between the river and the aquifer was begun in 1988 as a joint effort of the Kansas Water Office; the Equus Beds Groundwater Management District No. 2 (Halstead, Kansas); the Bureau of Reclamation, U.S. Department of the Interior; and the U.S. Geological Survey (USGS).

The purpose of the 4-year study was to improve understanding of the hydrologic and chemical interaction of the Arkansas River and *Equus* beds (stream-aquifer system) so that water-management agencies can develop strategies to minimize aquifer water-quality degradation and streamflow declines. In this study, "hydrologic interaction" includes the effect of water levels in the *Equus* beds aquifer on flow in the Arkansas and Little Arkansas Rivers, the effect of the Arkansas River on water levels in the aquifer, and the quantity of water exchanged between these rivers and the aquifer. "Chemical interaction" includes the quantity of chloride being discharged from the Arkansas River to the aquifer and the distribution of chloride from the river in the aquifer.

Specific objectives of this study were to develop a detailed understanding of the geology, hydrology, and chloride concentration near the Arkansas River; to use this information to develop a three-dimensional ground-water flow model; to use the ground-water flow model as a tool to help understand the flow between the river and aquifer and horizontal and vertical flow in the aquifer near the river; and to use the ground-water flow model and a particle-tracking program to estimate the horizontal and vertical distribution of chloride from the Arkansas River in the aquifer. It is the intent of the Bureau of Reclamation to use the ground-water flow model developed by the USGS to develop a solute-transport model to simulate chloride transport in the aquifer (Shirley Shadix, Bureau of Reclamation, written commun., 1993).

Purpose and Scope

This report presents the results of a hydrologic and chemical-interaction study of the Arkansas River and *Equus* beds aquifer, and flow-model and particle-tracking simulations of the stream-aquifer system between Hutchinson and Wichita in south-central Kansas. The primary geologic section considered consists of unconsolidated sediments of Pliocene and Pleistocene age (*Equus* bed sediment) that underlie and border the Arkansas River. The *Equus* beds aquifer, which consists of *Equus* beds sediment, is the primary aquifer that is considered. The Permian-age Wellington Formation and Ninnescah Shale, which underlie the unconsolidated sediments, are discussed relative to their effect on ground-water flow and ground-water quality in the unconsolidated sediments. Most of the water-level measurements and water-quality samples for this study were collected from 1986 to 1990; however, model-calibration simulations of ground-water flow cover 1940–89. In addition, model simulations are used to estimate the effects of natural and human-induced stresses on the stream-aquifer system. The sources of chloride and movement of chloride from the Arkansas River into the *Equus* beds aquifer are discussed.

Description of Study Area

The study area is located in south-central Kansas in parts of Reno, Harvey, McPherson, and Sedgwick Counties (fig. 1). The reach of the Arkansas River between Hutchinson and Wichita and the associated unconsolidated sediment (part of the *Equus* beds aquifer) are the focus of this study. However, use of a ground-water flow model necessitated the inclusion of major aquifer stresses, such as the Wichita well field and the Little Arkansas River (fig. 2), and boundaries, such as the contact between Permian-age bedrock and *Equus* beds sediment, in the study area. Principal cities in the area are Hutchinson, Newton, and Wichita. Other towns in the area are shown in figure 2.

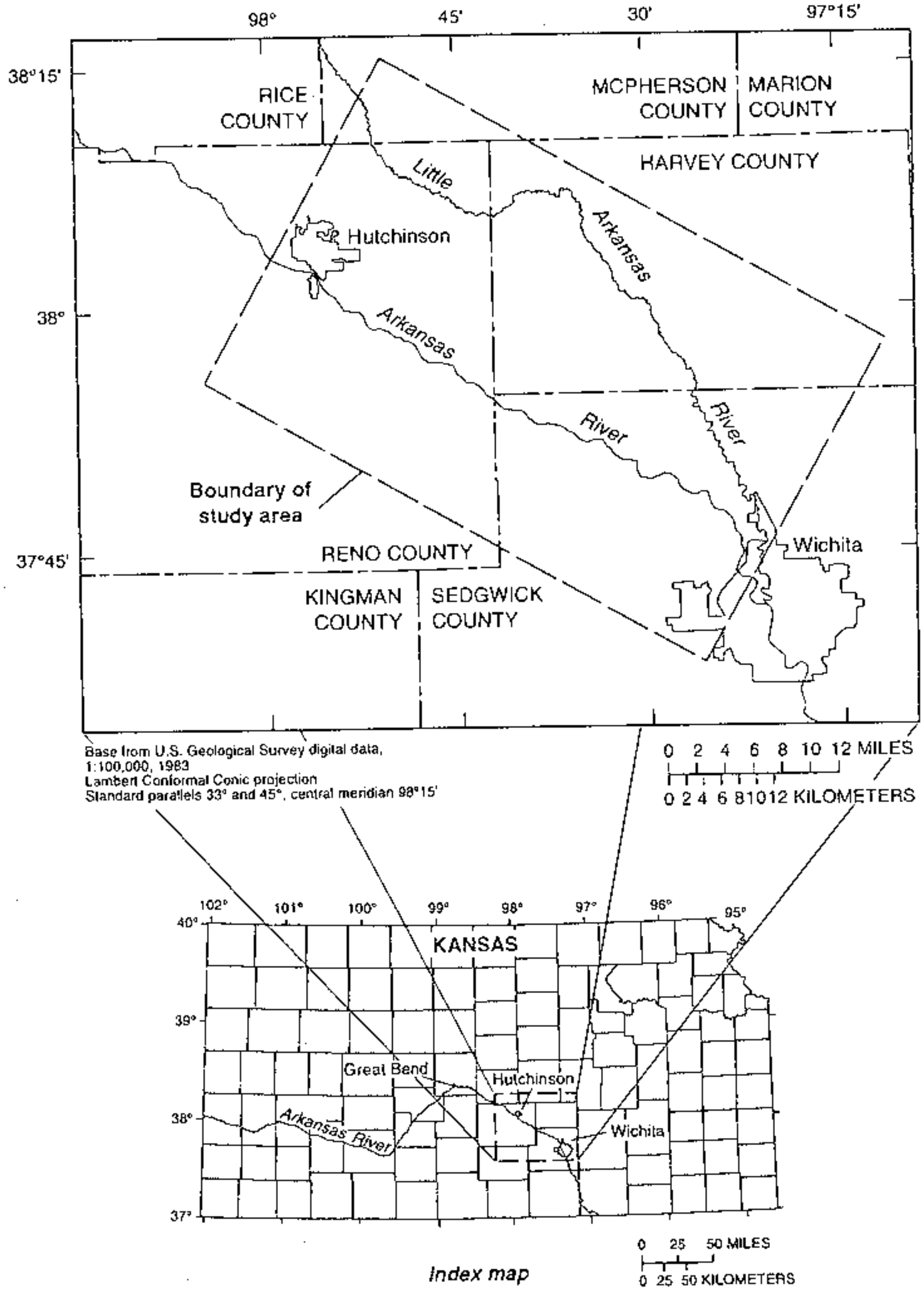


Figure 1. Location of study area.

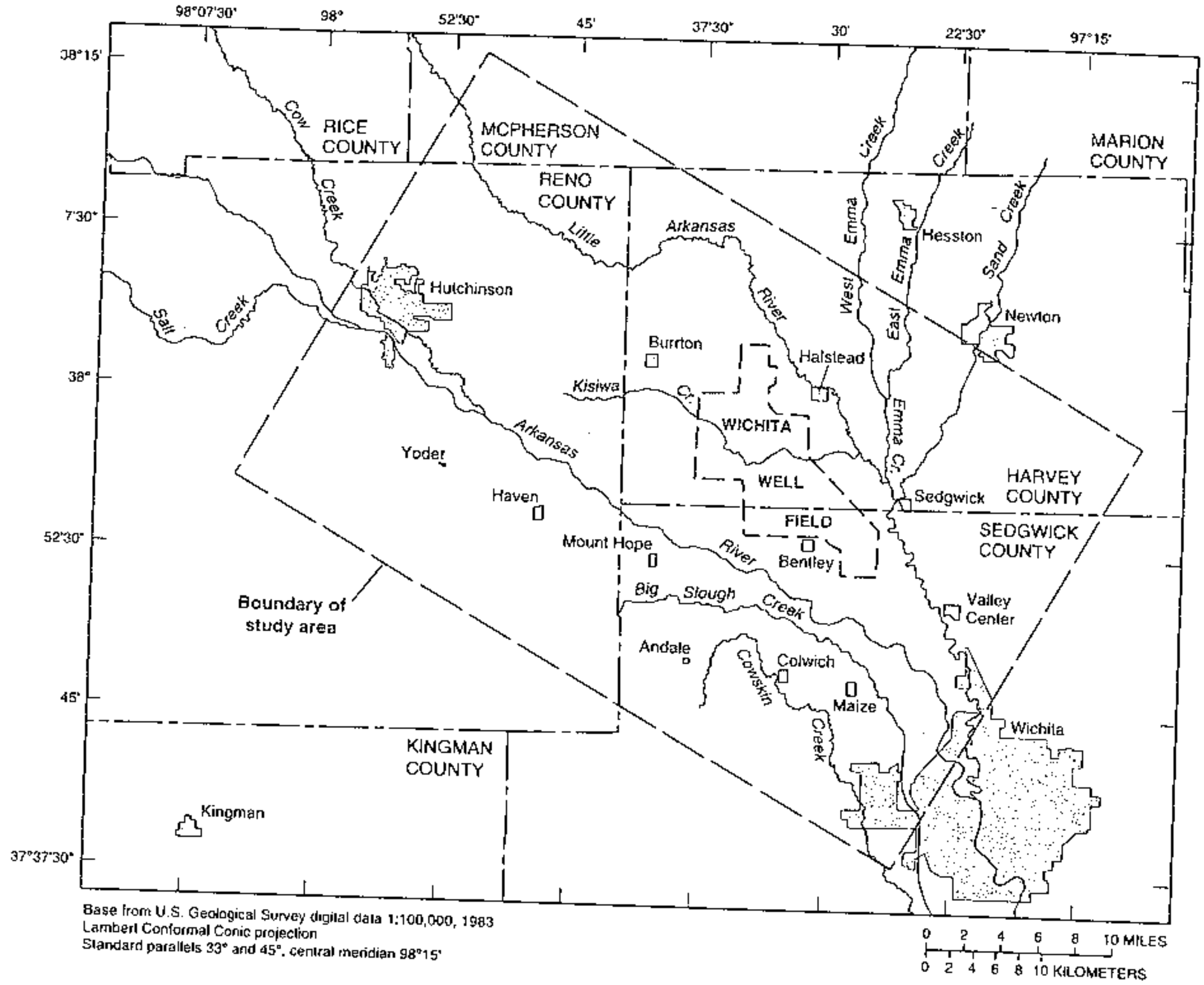


Figure 2. Location of study area, streams, Wichita well field, cities, and towns.

South-central Kansas has a continental climate and is characterized by large variations in seasonal temperatures, moderate precipitation, and windy conditions. Seasonal temperatures range from daily averages of 29.6 °F in January to 81.4 °F in July (National Oceanic and Atmospheric Administration, 1935–89). Temperatures may range from more than 100 °F in the summer to less than -20 °F in the winter. Mean annual precipitation at weather stations near Hutchinson, Monnt Hope, and Wichita is 29.88, 31.65, and 30.48 in., respectively, for 1940–89 (National Oceanic and Atmospheric Administration, 1935–89) or about 30.67 in. for the study area. Most of this precipitation occurs during spring and summer.

The study area lies in the Arkansas River section of the Central Lowland physiographic province (Schoewe, 1949). There is very little topographic relief in the study area except for an area of sand dunes (fig. 3) north and northeast of Hutchinson. For the most part, the land surface slopes gently toward the major streams in the area.

The Arkansas River and the Little Arkansas River are the major streams in the study area (fig. 2). The Arkansas River flows southeast in a fairly straight, slightly braided channel. The Arkansas River channel is entrenched 5 to 10 ft below the general land surface. In contrast, the Little Arkansas River meanders as it flows east and southeast to its confluence with the Arkansas River in Wichita. The channel of the Little Arkansas River is entrenched 15 to 25 ft below the general land surface. Cow Creek and smaller creeks are tributaries to the Arkansas River. Emma and Sand Creeks and smaller creeks are tributaries to the Little Arkansas River.

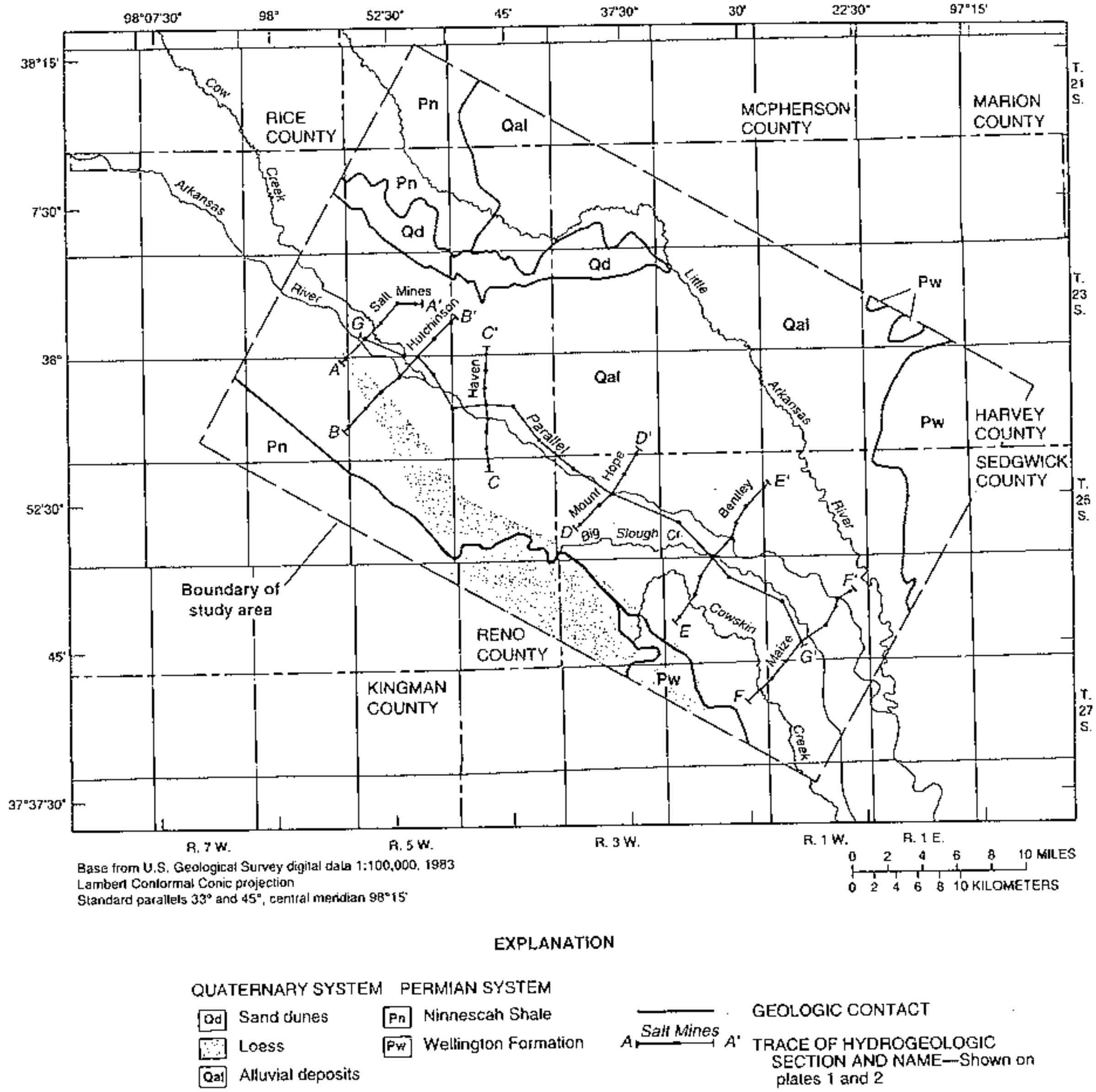


Figure 3. Areal geology, traces of hydrogeologic sections, and section names (geology modified from Kansas Geological Survey, 1964).

Site- and Well-Numbering System

In this report, wells installed during this study or utilized in a well network near the Arkansas River are designated as L-###-I, where L may be EB (*Equus* beds), NAS (North American Salt Company), or USGS-H (U.S. Geological Survey wells drilled near Hutchinson); ### is a site number; and I is a letter or letters designating the relative depth of the well. For example, EB-240-AA, EB-240-A, EB-240-B, and EB-240-C are four wells located at site 240 and have well screens that are shallow (AA), intermediate (A or B), and deep (C). Wells with relative depth designations of A (or any multiple of A's) through C have their screens in *Equus* beds sediment. One well with a depth designation of D (EB-237-D) has its well screen in Permian-age bedrock.

The numbering system for other wells, from which data were used in this study, is based on a modification of the Bureau of Land Management's

(U.S. Department of the Interior) system of land subdivision (fig. 4). The first number indicates the township south (S) of the Kansas-Nebraska State line; the second indicates the range east or west (E or W) of the sixth principal meridian; and the third indicates the section in which the well is located. The first letter following the section number denotes the quarter section or 160-acre tract; a second letter, the quarter-quarter section or 40-acre tract; a third letter, the quarter-quarter-quarter section or 10-acre tract. The 160-acre, 40-acre, and 10-acre tracts are designated A, B, C, and D in a counterclockwise direction beginning in the northeast quarter of the section. Wells or sampling sites in a tract are

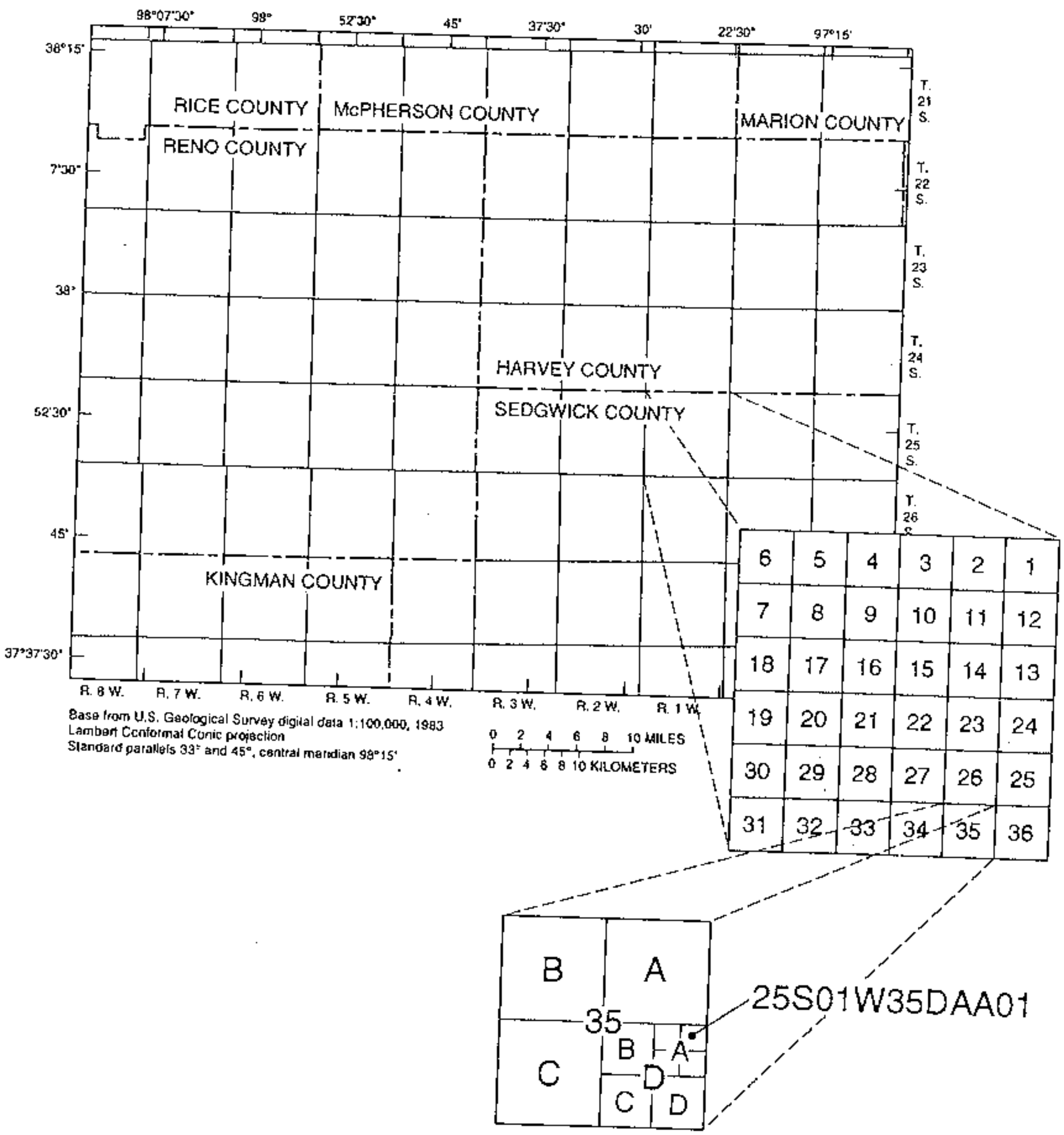


Figure 4. Modified Bureau of Land Management land subdivision system used for numbering well or site locations.

numbered consecutively, beginning with 1, in the order in which the wells or sites were inventoried. For example, 25S01W35DAA01 indicates the first well inventoried in the northeast quarter of the northeast quarter of the southeast quarter of sec. 35, T. 25S., R. 1W. (fig. 4).

Methods of Investigation

Results from previous investigations were used to guide data-collection efforts during this study. A network of 155 clustered (well screens at different depths) observation wells at 55 sites was established by the Bureau of Reclamation, the Kansas Geological Survey, and the USGS along the Arkansas River (fig. 5) during this and previous studies. Four wells at two sites (NAS-1-A, NAS-1-C, NAS-2-A, NAS-2-C) were drilled by the North American Salt Company (Hutchinson, Kansas) and were included in the network. Lithologic logs for the network wells are presented in the "Supplemental Information" section of this report. A streamflow-gaging station was installed on the Arkansas River near Maize, and water-level recorders were placed in five shallow wells (20–35 ft deep) that were drilled near the river (wells EB-204-AA, EB-210-AA, EB-216-AA, EB-221-AA, and EB-232-AA). Two pumping tests (fig. 6) were performed to help refine estimates of aquifer characteristics. Water samples from the observation-well network were collected annually by personnel of the Equus Beds Groundwater Management District No. 2 (GMD2) and were analyzed for major ions by the Kansas Department of Health and Environment (Topeka). During most of the study

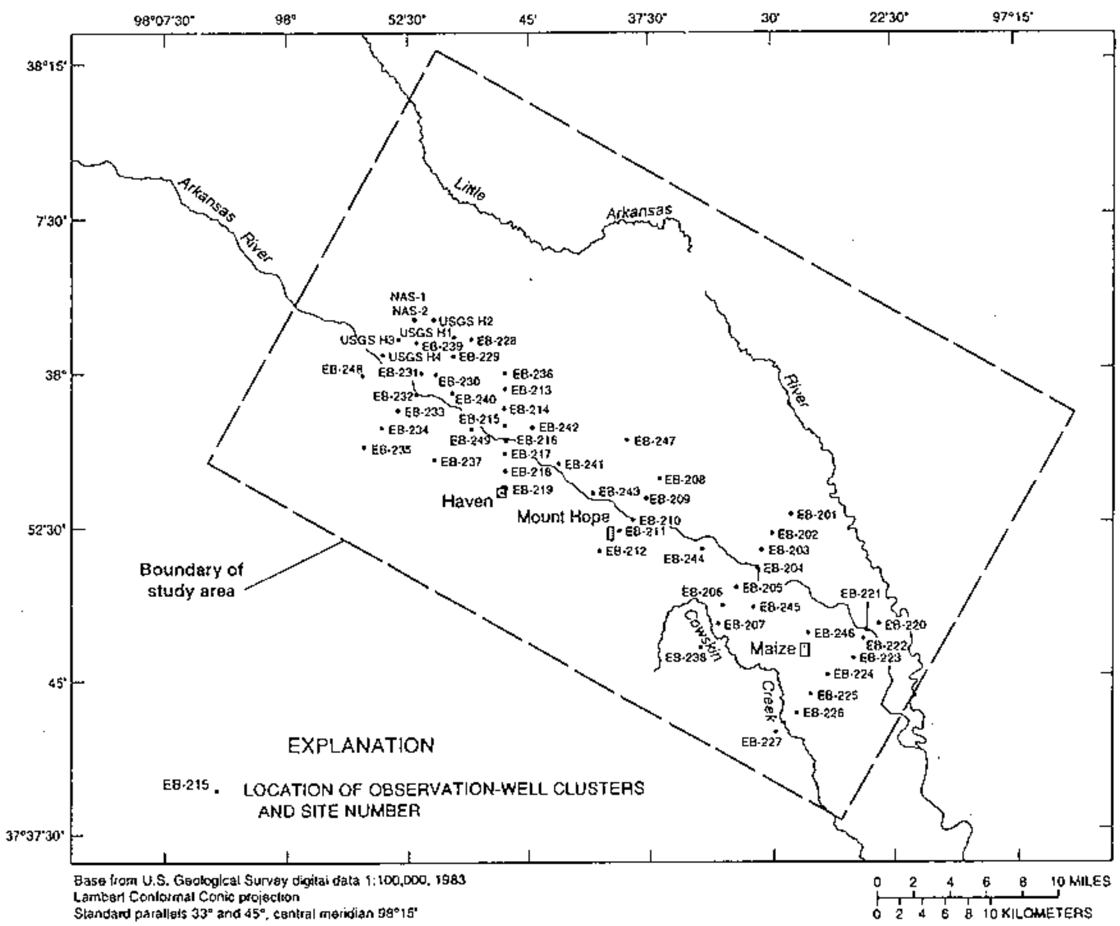


Figure 5. Location of observation-well clusters.

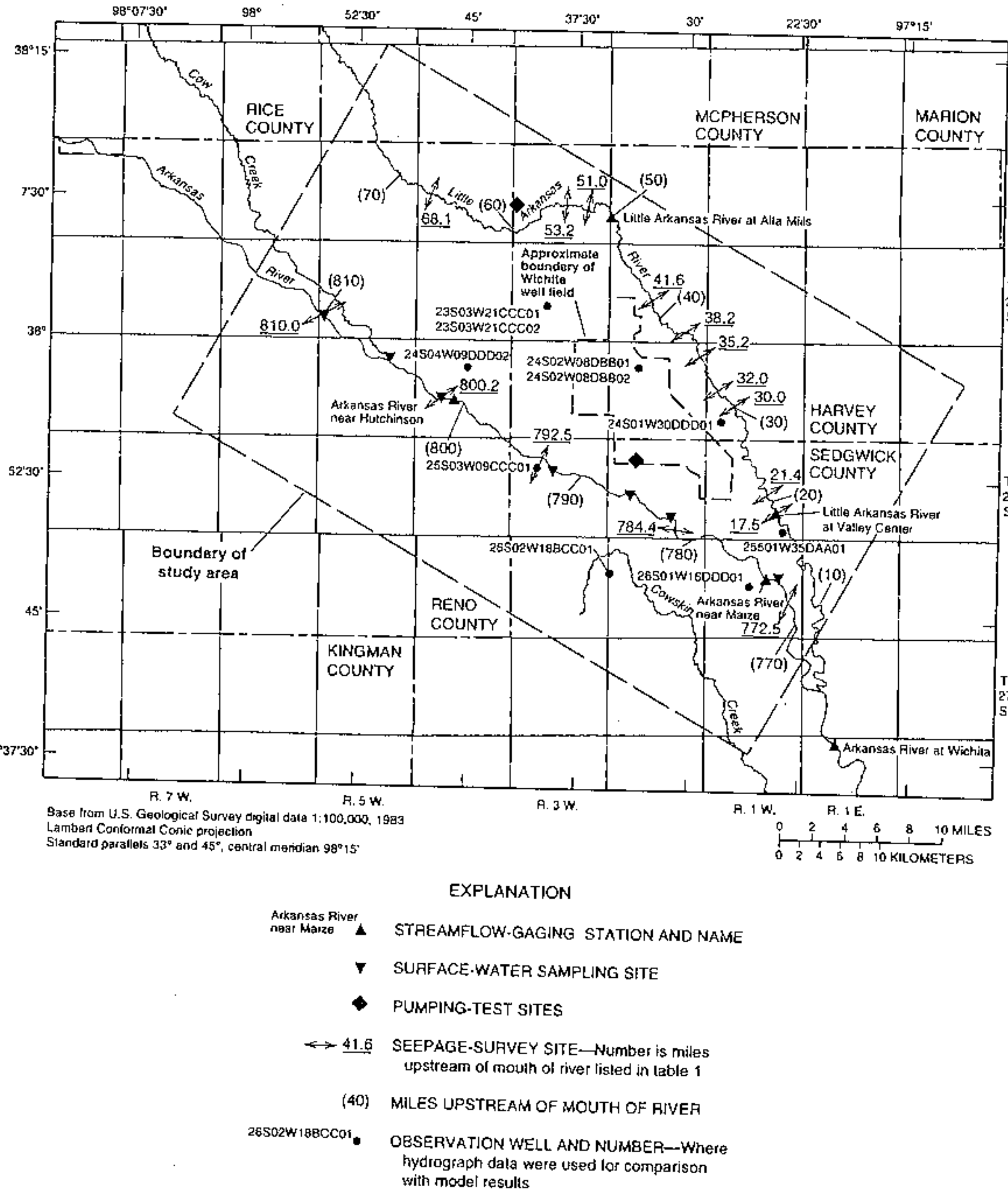


Figure 6. Location of streamflow-gaging stations, surface-water sampling sites, seepage-survey sites, pumping-test sites, and wells for which hydrographs were used to compare measured and simulated hydraulic heads.

period, GMD2 personnel measured, on a monthly basis, water levels in network observation wells and stream stage at selected bridges along the Arkansas River.

Seepage data for the Arkansas and Little Arkansas Rivers were obtained from USGS files in Lawrence, Kansas, for the years prior to this study (before 1988). Two seepage surveys were completed on the Arkansas River for this study during April 1989 and September 1990. The seepage snrveys were done using USGS methods for the measurement and computation of streamflow (Rantz and others, 1982).

Hydraulic-conductivity, recharge, evapotranspiration, and pumpage data for the ground-water flow model were obtained, in large part, from data sets of a single-layer ground-water flow model developed by Spinazola and others (1985). Water-level data collected during 1938–42 and reported in studies by Williams (1946) and Williams and Lohman (1949), and potentiometric-surface maps published by Williams and Lohman (1949) and Spinazola and others (1985) were modified to construct a circa-1940 potentiometric-surface map. A steady-state model was developed to simulate conditions that existed during the late 1930's (pre-1940) (Spinazola and others, 1985). Three model layers were defined on the basis of lithologic data from the observation-well network along the Arkansas River and lithologic logs published by Williams (1946) and Williams and Lohman (1949). Steady-state flow-model data were adjusted during the calibration process to approximate the pre-1940 potentiometric surface. Then, a transient model was developed to simulate conditions in the aquifer from 1940 to 1989. Flow-model aqnifer properties and stresses were refined during calibration of the transient model to approximate streamflows; 1971, 1980, and 1989 potentiometric surfaces; and water-level changes for 10 observation wells in the study area.

Previous Studies

The *Equus* beds aquifer is an important source of water to cities, industries, and farms. Because of the importance of this source of water and because of the occurrence of large chloride concentrations in parts of the aquifer, in streams, and in adjacent rocks, the *Equus* beds aquifer has been the subject of many studies. Williams and Lohman (1949) extensively described the geology and ground-water resources of the *Equus* beds. Williams (1946) described ground-water conditions near Hutchinson. Williams and Lohman (1947), Stramel (1956, 1962a, 1962b, 1967), and Petri and others (1964) have studied the aquifer in the Wichita well field area. Bayne (1956) and Lane and Miller (1965a) described the geology and hydrology of Reno and Sedgwick Counties, respectively. Bevans (1988) described the water resources of Sedgwick County.

Chloride concentrations in the *Equus* beds aquifer have been the subject of several studies. Leonard and Kleinschmidt (1976) studied the occurrence of saline water in the Little Arkansas River Basin, Hathaway and others (1981) studied the chemical quality of irrigation water in the *Equus* beds area, Williams (1946) discussed the origin of large concentrations of chloride in the aquifer near Hutchinson, and Gogel (1981) discussed the potential for discharge of saltwater from Permian-age rocks to the *Equus* beds. Whittemore (1982, 1990) and Whittemore and Basel (1982) identified sources of saltwater brines in the *Equus* beds using chloride-iodide and chloride-bromide ratios.

Investigators in previous ground-water flow and solute-transport modeling studies have simulated the *Equus* beds aquifer as one layer. These studies focused on the overall or specific aspects of the ground-water flow system and on the movement of chloride derived from concentrated sources such as oil-field brine or natural brine from the Wellington Formation. None, however, focused on the interaction between the Arkansas River and *Equus* beds aquifer.

Ground-water flow models of all or part of the *Equus* beds have been developed to determine the long-term safe yield of the aquifer (Green and Pogge, 1977; McElwee and others, 1979) and to describe the ground-water flow in the *Equus* beds aquifer (Spinazola and others, 1985) and between the *Equus* beds aquifer and the underlying Wellington aquifer (Gogel, 1981; Spinazola and others, 1985). Solute-transport models have been used to simulate the movement of chloride in the *Equus* beds aquifer (Sophcleous, 1983; Spinazola, 1985) and to develop water-use management strategies for aquifer reclamation and restoration (Heidari and others, 1986).

Acknowledgments

The authors wish to thank Shirley Shaddix, Bureau of Reclamation, for her time and effort in coordinating this study with the participating agencies. Most of the wells in the observation-well network installed along the Arkansas River were drilled by the Bureau of Reclamation. Mike Dealy and Don Koci, Equns Beds Groundwater Management District No. 2, measured a large number of water levels and collected water samples.

GEOLOGY

The oldest rocks that either crop out in the stndy area or subcrop beneath the *Equus* beds are the

Wellington Formation and Ninnescah Shale of Permian age (fig. 3). The Wellington Formation and Ninnescah Shale together make up the Sumner Group. The Sumner Group is underlain by rocks of the Chase Group. The Wellington Formation and the Ninnescah Shale rocks dip gently to the west. The Wellington Formation underlies the *Equus* beds in most of the study area.

The Wellington Formation is about 700 ft thick (Bayne, 1956) and is divided into the lower anhydrite member, Hutchinson Salt Member, and the npper shale member. The lower anhydrite member consists of about 200 ft of gray shale and silty shale with some carbonate lenses and thin anhydrite beds. The Hutchinson Salt Member consists of salt beds, interbedded halite, anhydrite, and shale that lie about 650 ft below land surface at Hutchinson (Bayne, 1956) and is mined at that location. The salt beds are about 300 ft thick at Hutchinson and reach a maximum thickness of 450 ft to the southwest, but thin rapidly to the northeast (fig. 7) due to dissolution of the salt by freshwater (Gogel, 1981). The upper shale member of the Wellington Formation consists of gray, green, and maroon shale interbedded with thin layers of limestone, dolomite, anhydrite, and gypsum (Bayne, 1956). The upper shale member is about 200 ft thick. The upper shale member is overlain by sediment of Quaternary age in about the eastern two-thirds of the study area. In this area, the upper surface of the upper shale member forms the bedrock surface. In about the western one-third of the study area, the upper shale member is overlain by the Ninnescah Shale.

The Ninnescah Shale crops out (fig. 3) or subcrops in the western part of the study area. It consists of red silty shale, siltstone, and fine-grained sandstone (Bayne, 1956). Gray-green streaks and spots are common in the unit. The maximum thickness is estimated to be 300 ft (Bayne, 1956). Southwest of the study area, the Ninnescah Shale contains a salt bed, but this salt bed has not been identified in the study area. The Ninnescah Shale is overlain by sediment of Quaternary age in about the western one-third of the study area. In this area, the upper surface of the Ninnescah Shale forms the bedrock surface.

Dissolution of the Hutchinson Salt Member and subsequent subsidence and collapse of overlying rock caused as much as 350 ft of Quaternary sediment to accumulate in subsiding areas. Areas of present-day subsidence are indicated by a linear trend of water-filled depressions and sinkholes at land surface (Williams and Lohman, 1949). Because of salt dissolution and pre-Quaternary erosion, the bedrock surface is irregular and is generally lowest where the greatest thickness of salt has been dissolved (fig. 8). The lowest part of the bedrock surface forms a depression, which lies near the present-day course of the Arkansas River. This bedrock-surface map has been updated with data obtained during this study and so may not match exactly the bedrock surface shown in figure 7.

Quaternary deposits occur throughont the study area as alluvial deposits, sand dunes, and loess deposits (fig. 3). The alluvial deposits, known as the *Equus* beds, are from 0 to abont 350 ft thick in the study area. For the purposes of this study, the *Equus* beds sediment was divided into lower, middle, and upper units on the basis of gamma logs and lithologic descriptions; refer to the "Supplemental Information" section, plates 1 and 2, Williams (1946), Williams and Lohman (1949), and Lane and Miller (1965b). The lower and upper units primarily consist of sand and gravel interbedded with clay or silt but may consist primarily of clay with thin sand and gravel layers. Sand in the lower unit, in general, is finer grained than sand in the upper unit. The middle unit primarily consists of clay or silty clay interbedded with sand and gravel but may consist primarily of sand and gravel with thin clay layers. The middle unit generally has more fine-grained material than the lower and upper units.

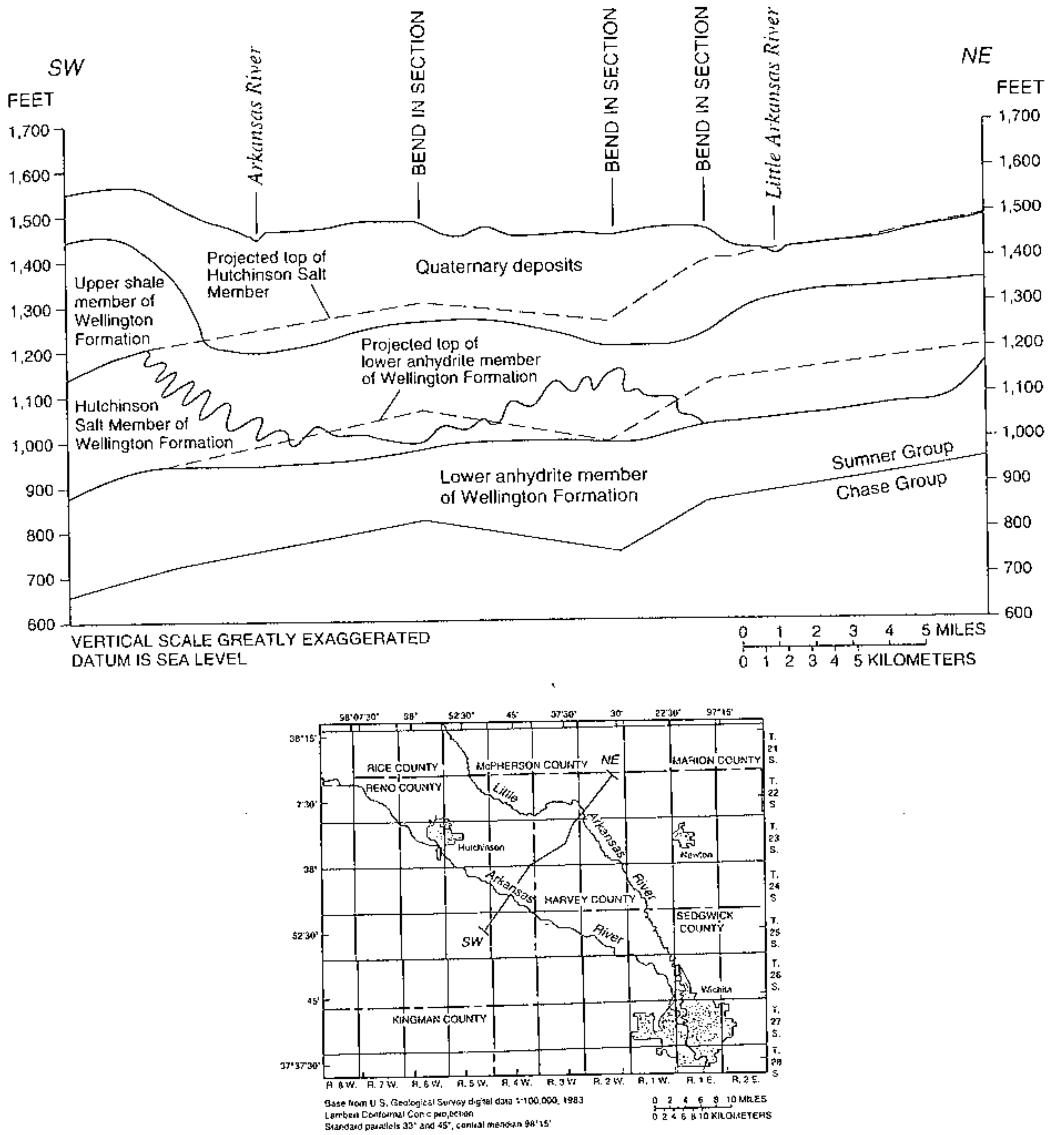
Sand dunes overlie the Ninnescah Shale near Hutchinson and overlie *Equus* beds sediment east of Hutchinson (fig. 3). Dune deposits consist of fine-grained tan sand with interbedded buried soil layers of silt, clay, and organic-material mixtures (Williams, 1946). The maximum thickness of dune sand is about 150 ft, but in most areas, dune sands do not exceed 30 to 50 ft in thickness.

Loess deposits occur primarily on uplands southwest of the Arkansas River (fig. 3). These windblown silt deposits are about 30 ft thick on uplands but thin rapidly towards the Arkansas River (Lane and Miller, 1965b).

HYDROLOGY

Surface Water

The Arkansas River, the Little Arkansas River, and their tributaries constitute the stream-drainage system in the study area. Flow in these streams is maintained primarily by base flow from the adjacent



Index map

Figure 7. Generalized geologic section showing present and projected past extent of the Hutchinson Salt Member (modified from Leonard and Kleinschmidt, 1976).

aquifer. Most of the base flow ultimately is derived from precipitation that recharges this aquifer (Bevans, 1988).

Seepage data collected during this study and from files at the USGS (Lawrence, Kans.) for the Arkansas River (table 1, fig. 9) for the 1980's and

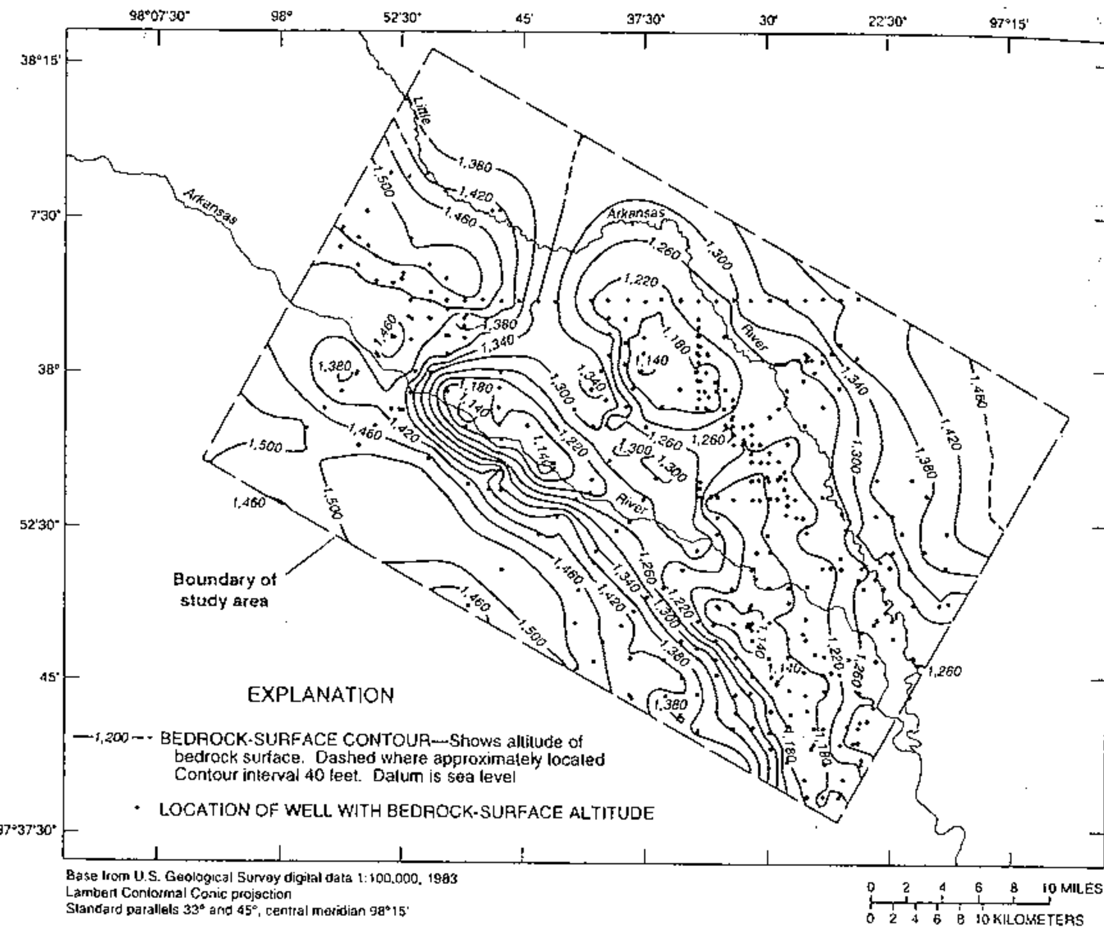


Figure 8. Altitude of bedrock surface (modified from Williams and Lohman, 1949; Lane and Miller, 1965a; and Spinazola and others, 1985).

1990 show that the river gains or loses water in the upper reach and loses water in the lower reach. When the npper reach of the river is gaining, the point at which the river changes from a gaining stream to a losing stream is between the Haven (*C–C'*) and Bentley (*E–E'*) hydrogeologic sections; this point probably moves upstream or downstream with changes in river stage and aquifer stresses.

The Little Arkansas River is primarily a gaining stream within the study area. Seepage data from files of the USGS (Lawrence, Kans.) (table 1, fig. 10) for the 1960's and 1970's show that the rate of gain is largest downstream of river mile 53.2. Some of this gain may be due to contributions from tributaries.

Streamflow hydrographs for the Arkansas River near Hutchinson, the Arkansas River near Maize, the Arkansas River at Wichita, the Little Arkansas River at Alta Mills, and the Little Arkansas River at Valley Center show seasonal streamflow patterns (figs. 11 and 12). Figures 11*A* and 12*A* show the monthly mean streamflow for the period for which there is contemporaneous streamflow data—March 1987–December 1991 for the three gaging stations on the Arkansas River and July 1973–December 1991 for the two gaging stations on the Little Arkansas River. In general, the largest streamflows in both rivers occur during the spring and summer months when most of the annual precipitation occurs. Smaller streamflows occur during the dryer fall and winter months.

Streamflow of the Arkansas River at Maize was less than streamflow of the Arkansas River near Hutchinson about one-half of the time for the

Table 1. Seepage-survey data available for the Arkansas and Little Arkansas Rivers

[Data collected during this study and from files of the U.S. Geological Survey, Lawrence, Kans.]

River mile upstream of mouth (fig. 6)	Date (month/day/ year)	Streamflow (cubic feet per second)	Gain (+) or loss (-) between measuring sites (cubic feet per second)
Arkansas River			
800.2	03/17/81	80	--
792.5	03/17/81	72	-8.0
772.5	03/17/81	67	-5.0
800.2	12/03/87	261	--
792.5	12/03/87	268	+7.0
784.4	12/03/87	282	+14
772.5	12/03/87	261	-21
800.2	04/20/89	81	--
792.5	04/20/89	66	-15
784.4	04/20/89	85	+19
772.5	04/20/89	59	-26
810.0	09/11/90	95	--
800.2	09/11/90	120	+25
792.5	09/11/90	107	-13
784.4	09/12/90	94	-13
772.5	09/12/90	84	-10
Little Arkansas River			
68.1	11/21/68	7.4	--
41.6	11/21/68	29	+21.6
35.2	11/21/68	32	+3.0
17.5	11/21/68	84	+52
68.1	11/13/69	2.6	--
53.2	11/13/69	3.6	+1.0
41.6	11/13/69	15	+11.4
35.2	11/13/69	18	+3.0
17.5	11/13/69	55	+37
68.1	02/20/70	2.9	--
41.6	02/18/70	21	+18.1
35.2	02/19/70	20	-1.0
68.1	09/10/70	.06	--
53.2	09/10/70	.18	+.12
41.6	09/10/70	5.8	+5.62
35.2	09/10/70	6.2	+.40
Little Arkansas River—Continued			
53.2	04/15/71	6.1	--
51.0	04/15/71	13	+6.9
38.2	04/15/71	18	+5.0
32.0	04/15/71	21	+3.0
30.0	04/15/71	22	+1.0
21.4	04/15/71	41	+19
17.5	04/15/71	46	+5.0
68.1	09/29/71	.55	--
53.2	09/30/71	1.4	+.85
41.6	09/30/71	10	+8.6
35.2	09/30/71	11	+1.0
17.5	10/01/71	24	+13
68.1	12/09/71	5.2	--
53.2	12/09/71	7.5	+2.3
41.6	12/10/71	22	+14.5
35.2	12/10/71	24	+2.0
68.1	11/07/72	4.9	--
41.6	11/08/72	7.0	+2.1
38.2	11/08/72	15	+8.0
35.2	11/08/72	16	+1.0
17.5	11/08/72	51	+35

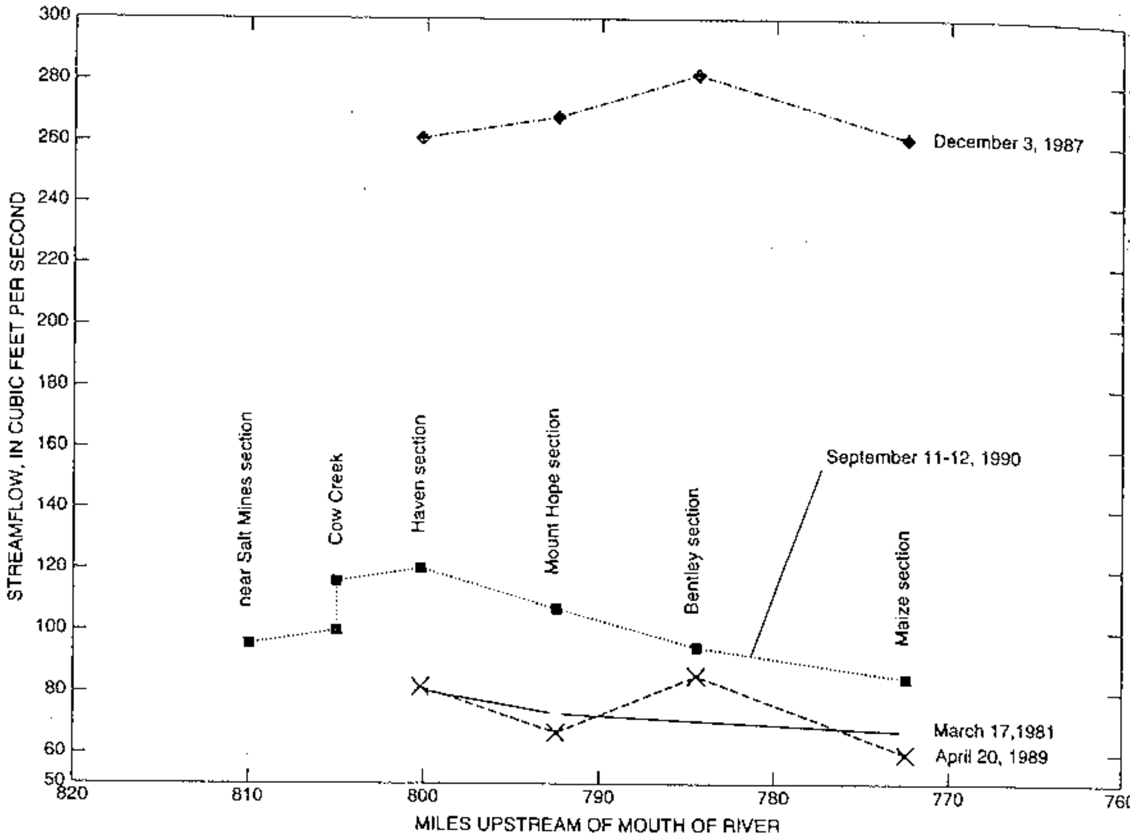


Figure 9. Measured streamflow in the Arkansas River during seepage surveys (seepage-survey sites shown in figure 6; hydrogeologic-section locations shown in figure 3; seepage data collected during this study and from files of the U.S. Geological Survey, Lawrence, Kans.).

contemporaneous period of record (fig. 11*B*). This relation usually occurs during the late-summer to early-spring months when small streamflows occur but may persist into late spring and summer during periods of less-than-normal rainfall, such as occurred in 1991 (fig. 11*B*). Streamflow of the Arkansas River at Wichita usually exceeds the streamflow at Maize or near Hutchinson (fig. 11*A*) because of the additional streamflow from the Little Arkansas River and other tributary streams. Streamflow of the Little Arkansas River at Valley Center rarely was less than streamflow of the Little Arkansas River at Alta Mills for the contemporaneous period of record (fig. 12*B*). The smallest differences in streamflow usually occur during the late-summer to early-spring months.

Ground Water

Equus beds sediment is an important source of ground water because of the generally shallow depth to the water table and the large saturated thickness. Near the Arkansas River, the water table may be as little as 10 ft below land surface. Farther from the river and near the Little Arkansas River, the water table may be at a greater depth below land surface, depending on the altitude of land surface and the amount of water-table decline that has been caused by long-term pumping. Data collected during this study indicate that the maximum saturated thickness within the study area, about 300 ft, is near the course of the Arkansas River and corresponds to the lowest areas of the bedrock surface (fig. 8).

The potentiometric-surface maps for 1940 and 1989 represent the composite potentiometric surface for all three units of the *Equus* beds aquifer. There were insufficient water-level data for some parts of the study area to construct separate potentiometric-surface maps for all three units. In areas where water-level data are available for all three units, such as near the Arkansas River, water-level altitudes in the three units

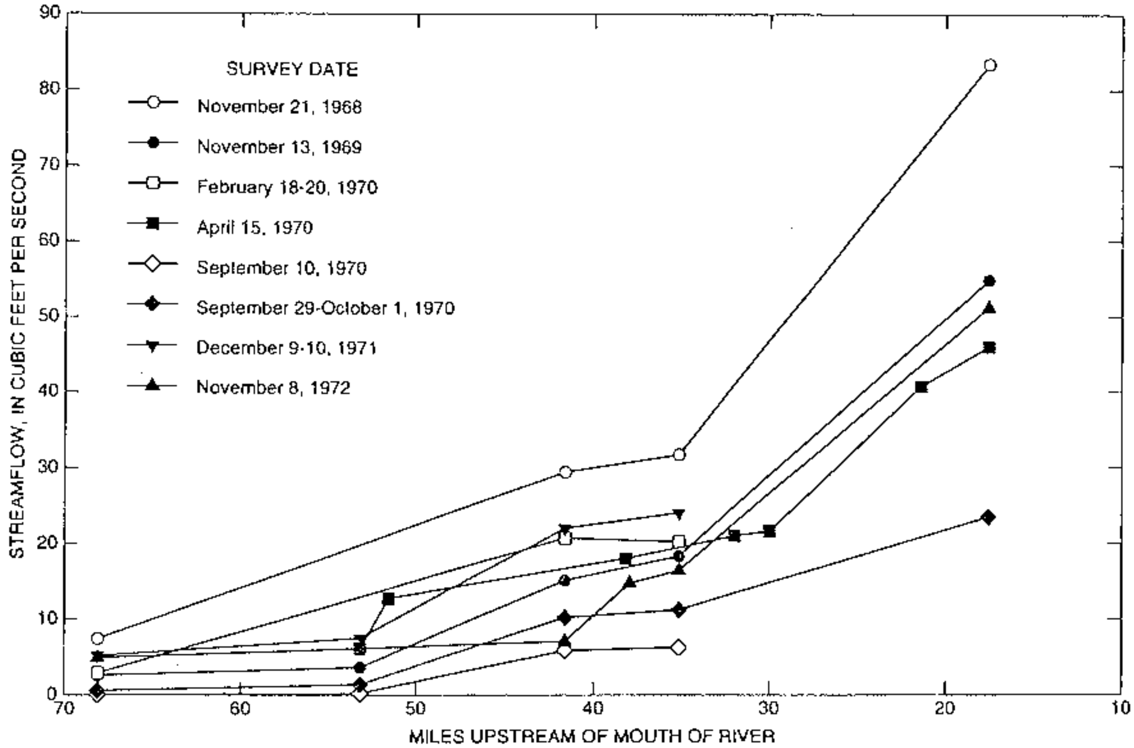


Figure 10. Measured streamflow in the Little Arkansas River during seepage surveys (seepage-survey sites shown in figure 6; seepage data from files of the U.S. Geological Survey, Lawrence, Kans.).

are similar (plates 1 and 2) except in the area of well sites EB-203 and EB-204 (plate 2).

Potentiometric-surface maps for 1940 and 1989 (figs. 13 and 14) indicate that the Arkansas and Little Arkansas Rivers control the direction of ground-water flow in the study area to a large extent. Near the Arkansas River, ground water flows southeast and generally parallels the direction of river flow (figs. 13 and 14). Near the Little Arkansas River, ground water flows towards the river (figs. 13 and 14). Northeasterly flow of ground water occurs southwest of the Arkansas River near Hutchinson (fig. 13). Except for the Wichita well field area, the direction of ground-water flow in 1989 generally is similar to that for 1940. In the Wichita well field area continuous pumping of municipal wells since the early 1940's and seasonal pumping of irrigation wells since the late 1950's have caused ground-water levels to decline as much as 30 ft or more (Lane and Miller, 1965a). Consequently the direction of ground-water flow in 1989 in the Wichita well field area is variable (fig. 14). Water-level data from clustered observation wells along the Arkansas River show that the overall direction of ground-water flow is similar in the upper, middle, and lower units (plates 1 and 2) except in the area of well sites EB-203 and EB-204 (plate 2).

The sand-dune area near Hutchinson contains layers of silt and clay that retard the downward movement of water, as shown by water levels in closely spaced wells that differ by as much as 27 ft (Williams and Lohman, 1949, table 37, wells 375 and 376), and by the presence of interdune ponds (Williams, 1946) and springs (Williams and Lohman, 1949). Nevertheless, the sand dunes are an effective precipitation-capture area and probably recharge the aquifer with a larger percentage of precipitation than other areas in the study area (Williams, 1946). A "ridge" of ground water in *Equus* beds sediment under and near the southern edge of the sand dunes (figs. 13 and 14) indicates that recharge in the sand dunes is larger than in surrounding areas.

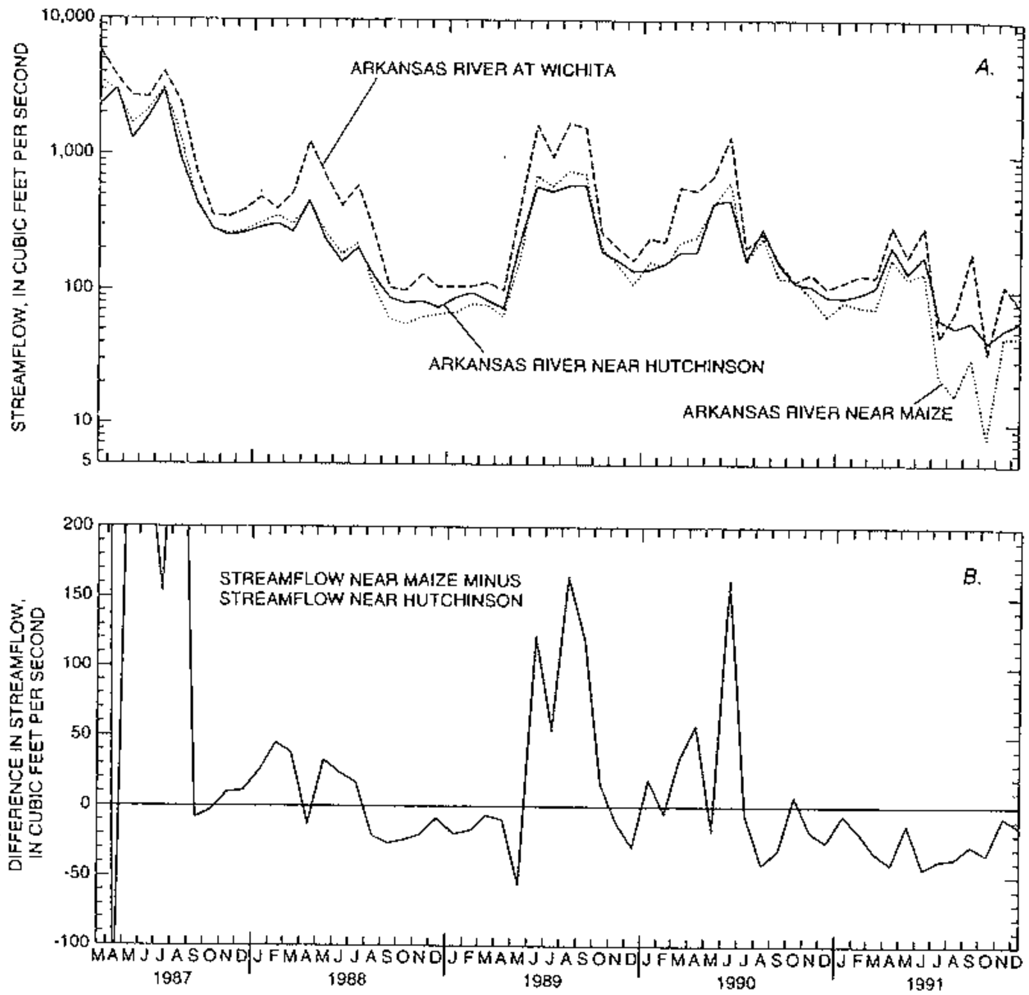


Figure 11. Monthly mean (*A*) streamflow for the Arkansas River near Hutchinson, near Maize, and at Wichita, and (*B*) difference between streamflow near Maize and near Hutchinson.

Hydraulic Relation Between Rivers and Aquifer

The relation of the potentiometric surface to the Arkansas and Little Arkansas Rivers is indicative of the hydraulic conductivity of the sediment near the rivers. Potentiometric-surface contours cross the Arkansas River nearly perpendicular to the channel (figs. 13 and 14). River stage and ground-water levels are very nearly the same because nearby aquifer sediment consists of coarse material that transmits water-level changes readily. Hydrographs from a streamflow-gaging station (Arkansas River near Hutchinson) and a well about 300 ft downstream (well EB-216-AA) show that water-level changes in the river are transmitted very quickly to the nearby well (fig. 15). Water-level changes in the river also are reflected by changes in the water levels in deeper wells at the same site (fig. 16). In contrast, potentiometric-surface contours bend upstream before they cross the Little Arkansas River (figs. 13 and 14). The water level in the aquifer is higher than the water level in the river because nearby aquifer sediment is finer grained and less transmissive than the sediment near the Arkansas River.

Water Use

Withdrawal of water by wells is a significant source of discharge from the *Equus* beds aquifer. Prior to 1940, water withdrawals from the *Equus* beds, which were relatively insignificant in terms of their effects on the aquifer, were mainly for municipal and

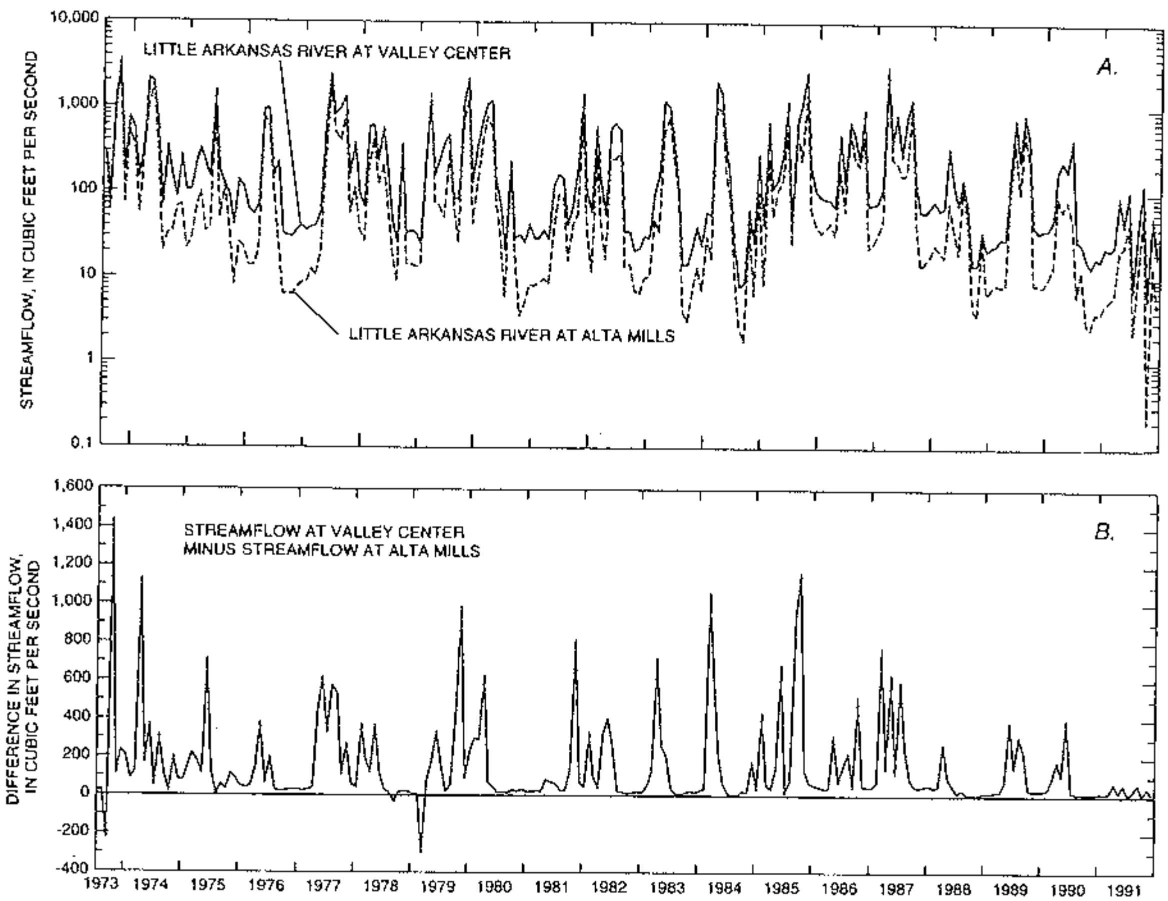


Figure 12. Monthly mean (*A*) streamflow for the Little Arkansas River at Alta Mills and at Valley Center, and (*B*) difference between streamflow at Valley Center and at Alta Mills.

industrial use and occurred near the cities of Hutchinson and Wichita (Spinazola and others, 1985). With the development of 25 wells in the Wichita well field in 1940, municipal water use increased rapidly from 1940 to about 1952 (fig. 17). In 1992, there were 55 municipal wells in the Wichita well field. Water withdrawals from the aquifer were fairly constant throughout the 1950's, but in the late 1950's and early 1960's, agricultural and industrial water uses began increasing. Agricultural water use (primarily irrigation) was fairly uniform in distribution throughout the study area, including the area of the Wichita well field. Industrial water use was limited to local areas. In the mid-1970's agricultural water use increased substantially and has been the single largest use of water since the early 1980's (fig. 17).

Most of the municipal wells in the Wichita well field obtain water from the middle and lower units of the *Equus* beds aquifer. Irrigation wells near the Arkansas River usually obtain water from the upper and middle units because of the large chloride concentrations found in the buried bedrock depression near the course of the river. Irrigation wells farther from the river may obtain water from all three layers. Some industrial wells also may obtain water from all three layers.

CHLORIDE IN WATER

Large concentrations of chloride make water unsuitable for uses such as human and livestock consumption and crop irrigation. The Kansas Department of Health and Environment (1986) has established a Secondary Maximum Contaminant Level (SMCL) of 250 mg/L (milligrams per liter) for

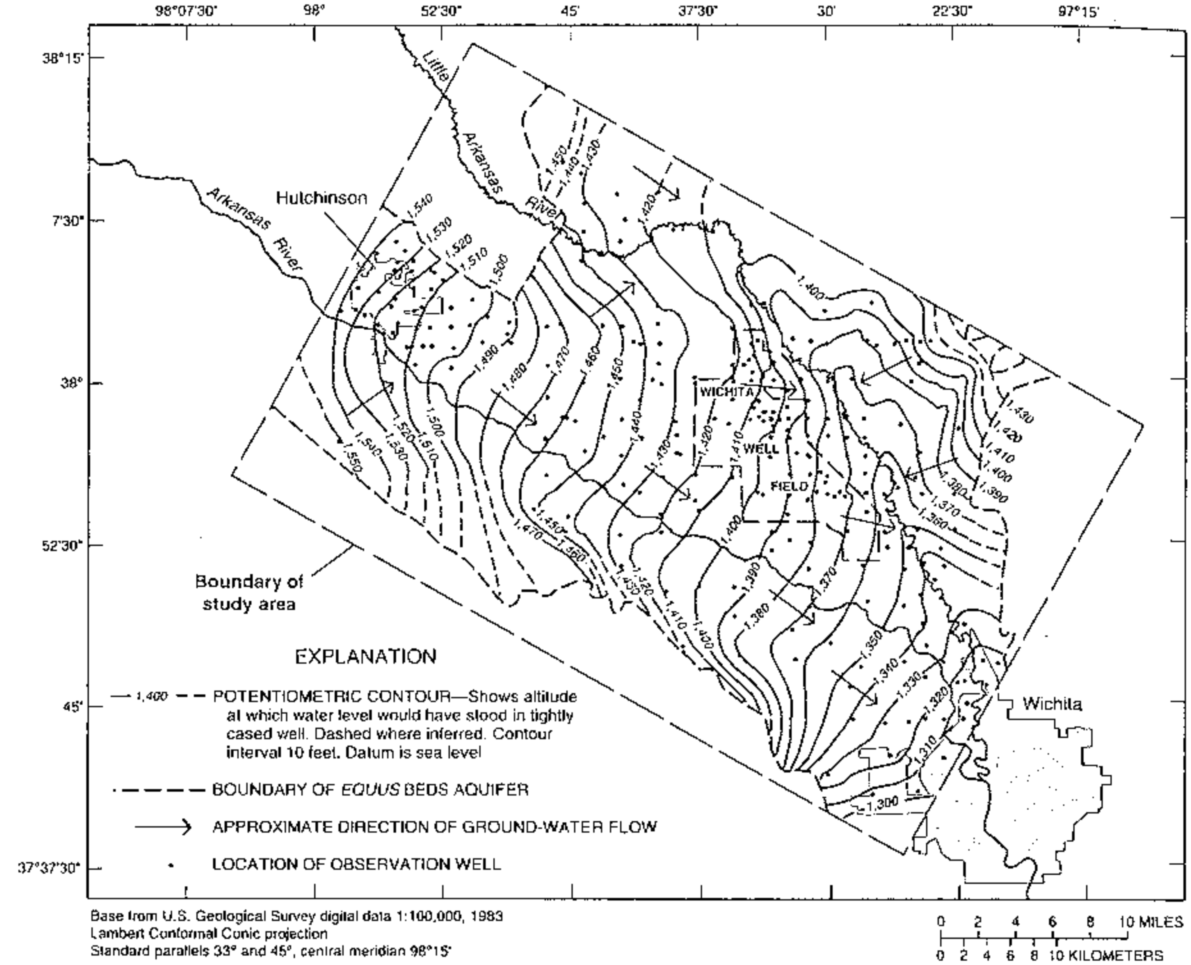


Figure 13. Potentiometric surface in the *Equus* beds aquifer, 1940 (composite of water levels from all three units) (modified from Spinazola and others, 1985).

chloride in drinking water. SMCL's have been established for constituents of water that affect the aesthetic qualities of the water, such as taste, color, or smell. SMCL's are not enforceable.

Sources of Chloride

Two natural sources of chloride and three artificial sources, resulting from human activities, affect ground-water quality in the study area. The two natural sources of chloride are Arkansas River water and saline ground water from the Wellington Formation (Gogel, 1981). The three artificial sources of chloride are brine from oil-field activities, brine from salt-mining activities, and evaporation-pan brine from salt-refining activities. In addition, Williams and Lohman (1949) noted large chloride concentrations in Arkansas River water downstream of a sewage outlet near Hutchinson, probably derived from salt-mining and salt-refining activities.

Because of the multiple sources of the chloride, it was useful to distinguish the naturally derived chloride from the artificially derived chloride. In this way chloride in ground water that came from the Arkansas River could be identified more easily and tracked. During this study, samples of water from the observation-well network along the Arkansas River were analyzed by Whittemore (1990) who used chloride-iodide and chloride-bromide ratios to distinguish chloride from oil-field, salt-refining, and natural sources. Chloride from salt-mining activities was indistinguishable from chloride from natural

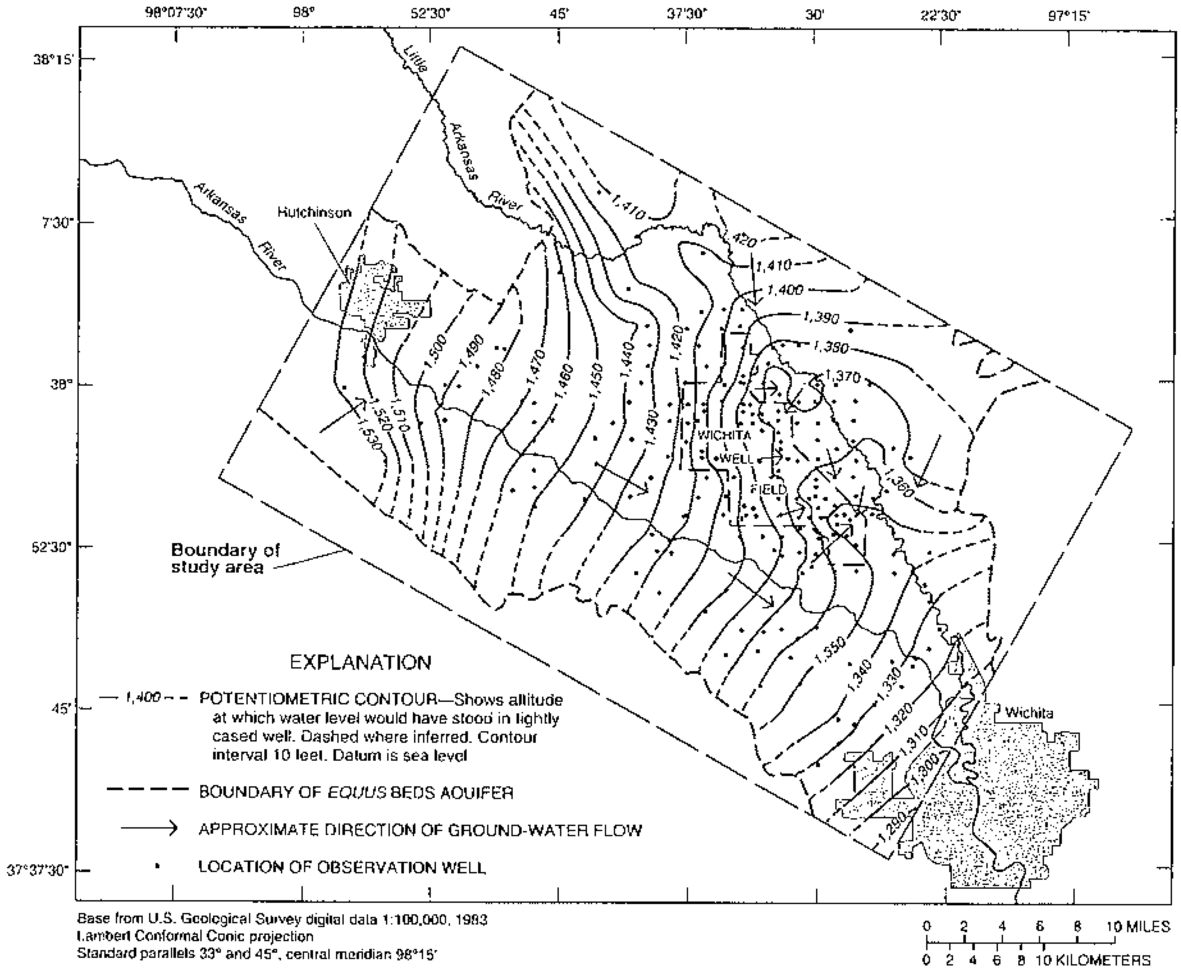


Figure 14. Potentiometric surface in the *Equus* beds aquifer, 1989 (composite of water levels from all three units) (data from the U.S. Geological Survey WATSTORE data base).

sources. However, chloride from mining activities most likely would be found in the vicinity of the salt mines near Hutchinson. Using mixing curves, Whittemore (1990) also determined the concentrations of chloride in samples that were derived from oil-field, salt-refining, and natural sources.

Chloride in Surface Water

Chloride concentrations in Arkansas River water increase downstream of Great Bend, Kans. (fig. 1). Much of the chloride probably comes from salt marshes on tributaries to the Arkansas River upstream of Hutchinson (Williams, 1946). Within the study area, water from the Arkansas River is classified as brackish or salty (Williams, 1946). Williams and Lohman (1949) reported that concentrations of chloride in Arkansas River water samples collected during the winter of 1934–35 at two sampling sites near Hutchinson ranged from 392 to 460 and 750 to 1,895 mg/L. The largest chloride concentrations were downstream of the sewage outlet near Hutchinson (Williams and Lohman, 1949). Chloride concentrations were as large as 1,400 mg/L and were generally larger than 1,000 mg/L during low river flows in the fall of 1937 (Williams and Lohman, 1949).

Samples of Arkansas River water were collected during this study at sampling sites along the river (fig. 6) near Hutchinson, Haven, Mount Hope, Bentley, and Maize. Median chloride concentrations for each of the five sites ranged from 620 to 640 mg/L (table 2). The median chloride concentration for all

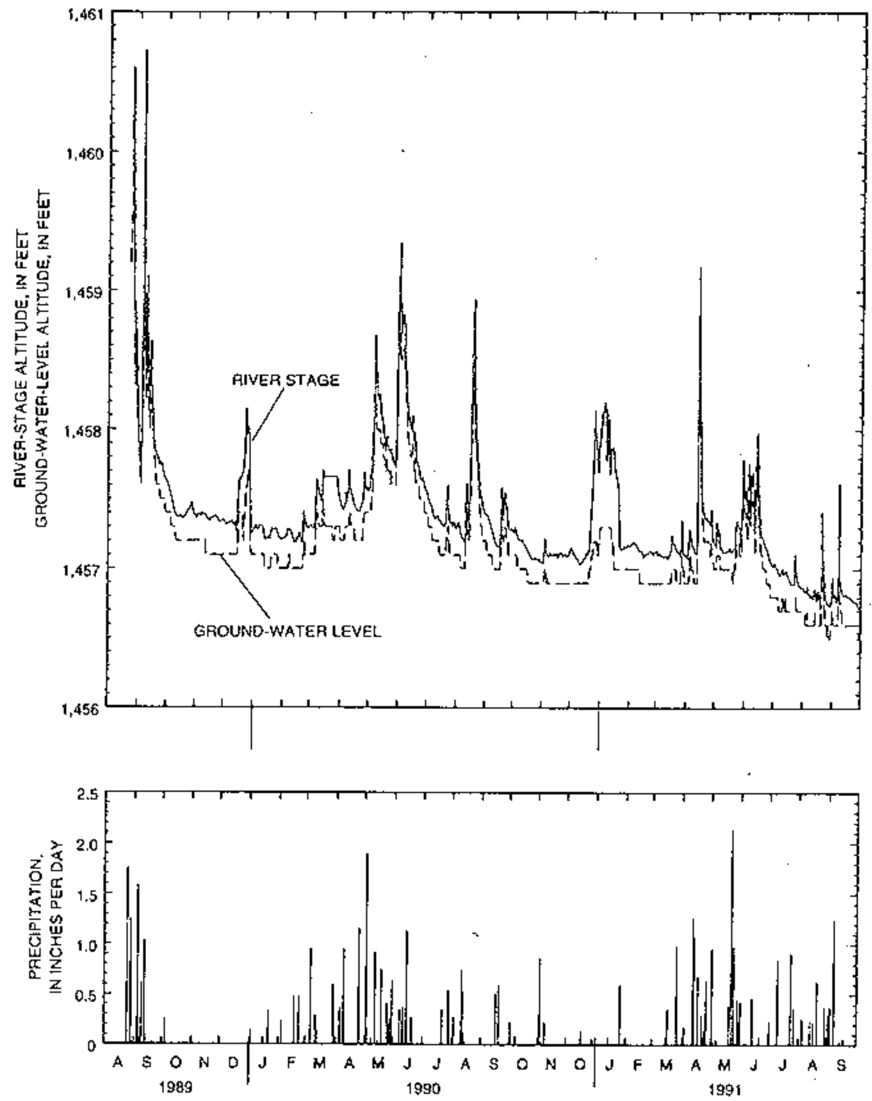


Figure 15. Daily mean river-stage altitude of the Arkansas River near Hutchinson (gaging station 07143330), daily mean water-level altitude in nearby well EB-216-AA, and daily precipitation rate at streamflow-gaging station, August 1989–September 1991. Well EB-216-AA was completed in the upper unit of the *Equus* beds aquifer and is perforated from 20 to 30 feet below land surface.

175 samples was 630 mg/L (table 2). Figures 18*A* and 18*B* show that generally there is an inverse relation between flow in the river and chloride concentration. Chloride loads in the river (fig. 18*C*) are a function of flow and concentration, and fluctuate depending primarily on streamflow but also on the chloride concentration of the stream water.

The Little Arkansas River also is known to have had salty water although generally not in as large concentrations as in the Arkansas River. Leonard and Kleinschmidt (1976) reported that chloride concentrations in water samples collected during 1960–72 at Valley Center ranged from 56 to 220 mg/L. The maximum chloride concentrations in the Little

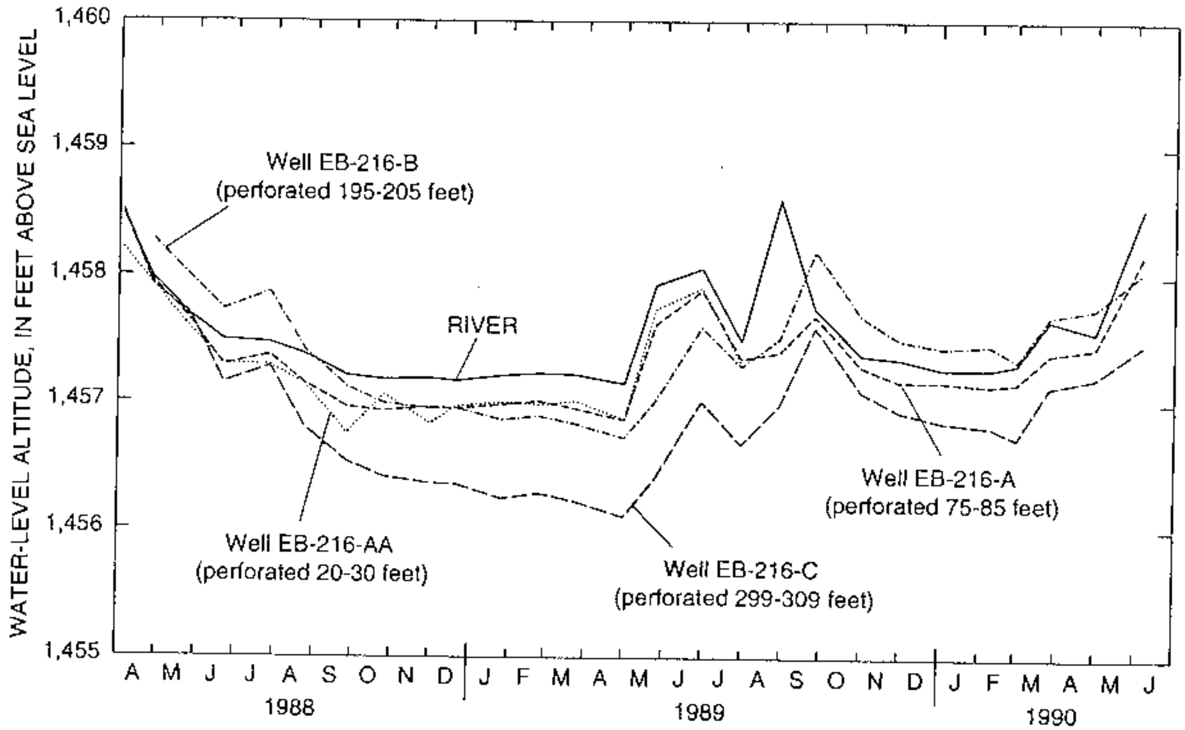


Figure 16. Monthly instantaneous ground-water levels in wells EB-216-AA, EB-216-A, EB-216-B, and EB-216-C and the daily mean river stage of the Arkansas River near Hutchinson for those days when ground-water levels were measured, April 1988–May 1990.

Arkansas River occurred near the mouths of tributaries draining oil-field areas (Leonard and Kleinschmidt, 1976).

Chloride in Ground Water

Williams (1946) found that chloride concentrations in wells within 1 mi of the Arkansas River in the vicinity of Hutchinson ranged from "... a few hundred..." to 1,200 mg/L. He also found that, with the exception of ground-water wells in oil fields, the chloride concentrations in ground water were progressively smaller farther from the river. In general, this is the pattern for the entire river reach from Hutchinson to Wichita. Hathaway and others (1981, fig. 9) showed that chloride concentrations generally ranged from 75 to 250 mg/L in an area along the Arkansas River, and concentrations exceeded 250 mg/L at some locations. Hathaway and others (1981) also stated that a comparison of data from Williams and Lohman (1949) and data from their study "...suggests that pumpage of wells in the Wichita well field area has produced little apparent change in chloride concentration levels of waters in this region."

Chloride analyses of water samples collected during this study (fig. 19) show a pattern similar to that found by Hathaway and others (1981). The concentrations shown in figure 19 represent the concentrations of natural chloride in ground water based on geochemical identifications by Whittemore (1990). Thus, in areas where chloride from oil-field and salt-refining brines is present, the chloride concentration shown in figure 19, which represents the natural chloride concentration, is some fraction of the total concentration of dissolved chloride. The largest concentrations of natural chloride were found in the lower unit of the *Equus* beds aquifer (fig. 19*C*). These large concentrations may result from subsurface flow

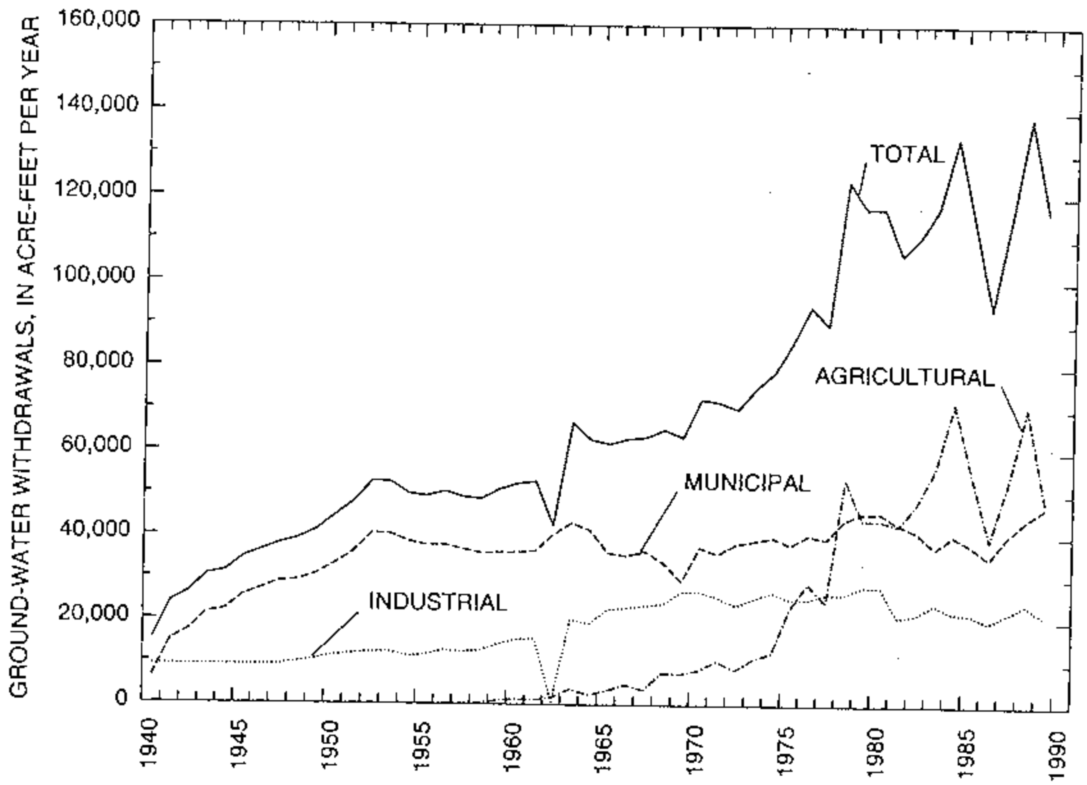


Figure 17. Industrial, municipal, agricultural, and total pumpage from the *Equus* beds aquifer in study area for 1940–89 (modified from Spinazola and others, 1985). Data for 1980–89 were estimated from records of pumpage in Harvey, Reno, and Sedgwick Counties (data obtained from Kansas Department of Agriculture, Division of Water Resources, Topeka, Kans.).

of saline water from zones of active salt dissolution in the Wellington Formation. Concentrations of natural chloride in the upper and middle units are similar to each other (figs. 19*A* and 19*B*). The primary source of this natural chloride probably is the Arkansas River. Other sources could be mined-salt brine (indistinguishable from other natural sources) that has not been concentrated by evaporation or saline water from the Wellington Formation. In the upper unit, the areal distribution of natural chloride concentrations reflects the gaining and losing reaches of the Arkansas River. During periods of base flow, the upstream part of the reach between Hutchinson and Wichita may be gaining or losing water, and the downstream part is losing water. Thus, in the upper unit, water from the river has penetrated a greater distance into the aquifer (fig. 19*A*) in the downstream part of the reach. Some of the natural chloride in the upper unit near Hutchinson may have originated from saline water in Cow Creek (table 2).

CONCEPTUAL MODEL OF THE *EQUUS* BEDS AQUIFER

A conceptual model of the *Equus* beds aquifer, including boundaries, aquifer properties, ground-water recharge, and ground-water discharge, was useful to guide the development of the ground-water flow model and for later evaluation of results from the ground-water flow model.

Boundaries

Within the study area, shale of the Wellington Formation and Ninnescah Shale acts as a low-permeability barrier to ground-water flow. This shale exists beneath all of the study area and crops out on the southwest side and in the northwest and northeast corners of the study area. The sediments that comprise the *Equus* beds aquifer extend beyond the study area along the Little Arkansas and Arkansas River Valleys.

Table 2. Summary of chloride-concentration data collected from Cow Creek and the Arkansas River, August 1988–July 1991

Sampling-site name and number (fig. 6)	Number of samples	Chloride concentration, in milligrams per liter			
		Mean	Median	Minimum	Maximum
Cow Creek near confluence with Arkansas River 375906097503900	22	533	495	380	740
Arkansas River at Hutchinson 375903097515700	27	621	640	340	1,100
Arkansas River near Hutchinson 07143330	40	634	640	190	1,100
Arkansas River near Mount Hope 375343097394000	29	610	620	380	1,000
Arkansas River 4 miles northeast of Colwich 375032097305500	30	619	635	240	1,100
Arkansas River near Maize 07143375	49	590	620	140	1,100
Arkansas River, all sites	**175**	**613**	**630**	**140**	**1,100**

Aquifer Properties

Aquifer properties include horizontal and vertical hydraulic conductivity and specific yield. Horizontal hydraulic-conductivity values calculated from transmissivities reported by Reed and Burnett (1985) ranged from 55 to 1,000 ft/d. Hydraulic conductivities calculated from two pumping tests done during this study were 50 and 1,200 ft/d. Vertical-to-horizontal hydraulic-condnctivity ratios from aquifer tests reported by Reed and Burnett (1985) and from two pumping tests done during this study ranged from 0.0006 to 0.22. Specific yield of fine- to coarse-grained materials, which comprise the *Equus* beds allnvial sediments, typically range from 0.1 to 0.35 (Fetter, 1988) and were calculated to range from 0.08 to 0.34 for aquifer tests reported by Williams and Lohman (1949) and Reed and Burnett (1985). Spinazola and others (1985) used a specific yield of 0.15 in their gound-water flow model of the *Equus* beds. Storage coefficients calculated by Reed and Burnett (1985) and from two pumping tests done during this study ranged from 0.0004 to 0.16.

Recharge

Recharge to the *Equus* beds aquifer is from snbsurface inflow, precipitation, streamflow losses, and irrigation return flow. On the basis of the 1989 potentiometric surface (fig. 14), subsurface inflow probably occurs along parts of the northwest and northeast sides of the study area. An estimate of the inflow across the northwest side, assuming a hydraulic gradient of 0.0011, a hydraulic conductivity of 450 ft/d, and an inflow area of 3,836,000 ft^2, is about 22 ft^3/s. Estimated inflow across the northeast side of the study area, assuming a hydranlic gradient perpendicular to the study-area boundary of about

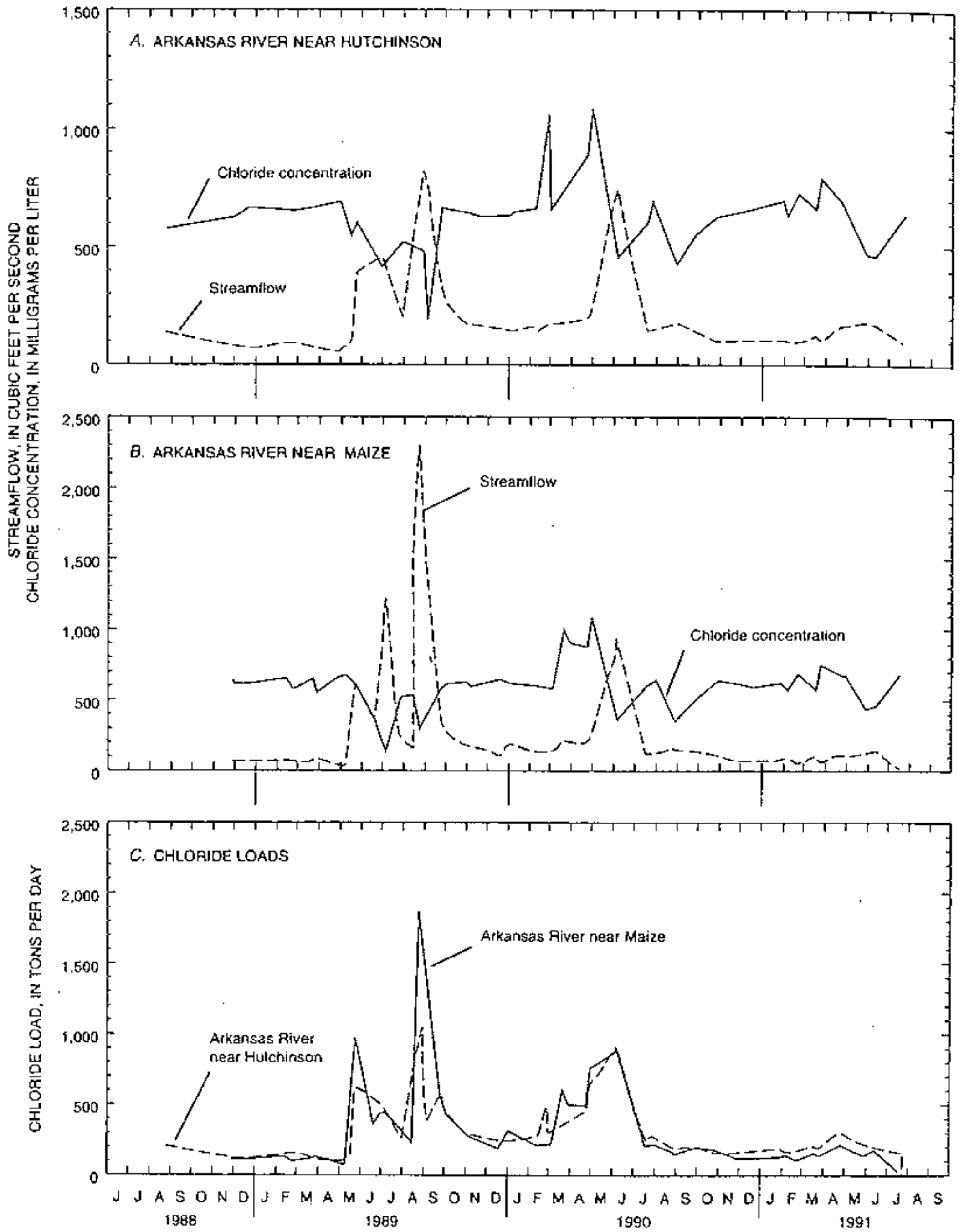


Figure 18. Streamflow and chloride concentrations in (*A*) the Arkansas River near Hutchinson, August 1988–July 1991, and (*B*) in the Arkansas River near Maize, November 1988–July 1991, and (*C*) chloride loads in the Arkansas River near Hutchinson, August 1988–July 1991, and the Arkansas River near Maize, November 1988–July 1991. Load calculated from streamflow and chloride-concentration data (chloride-concentration data from Equus Beds Groundwater Management District No. 2, Kansas Geological Survey, and USGS; data on file with the U.S. Geological Survey, Lawrence, Kans.).

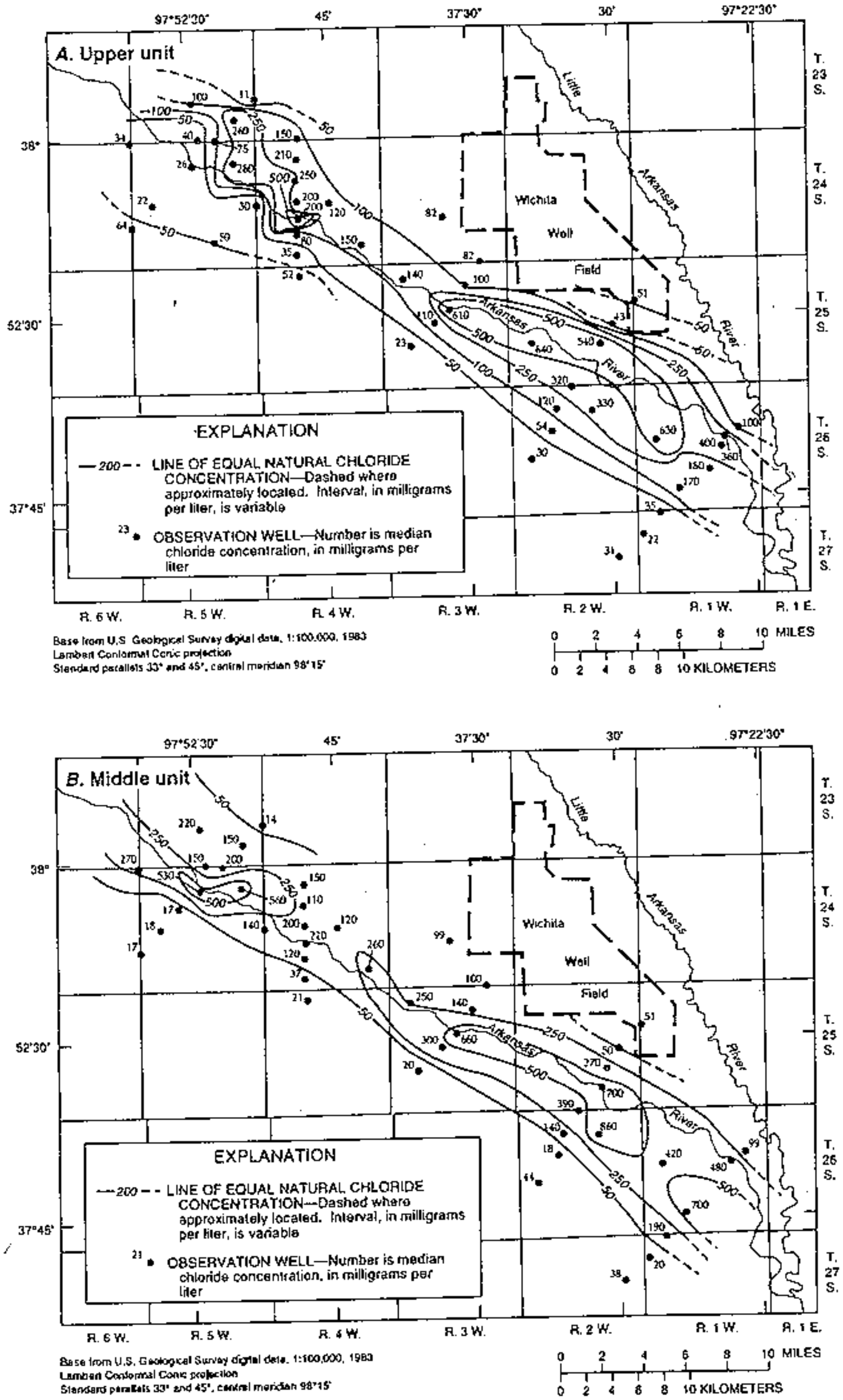


Figure 19. Natural chloride concentrations and lines of equal concentration in (*A*) upper, (*B*) middle, and (*C*) lower units of the *Equus* beds aquifer, 1989–90 [data from Whittemore (1990) and on file with the U.S. Geological Survey, Lawrence, Kans.].

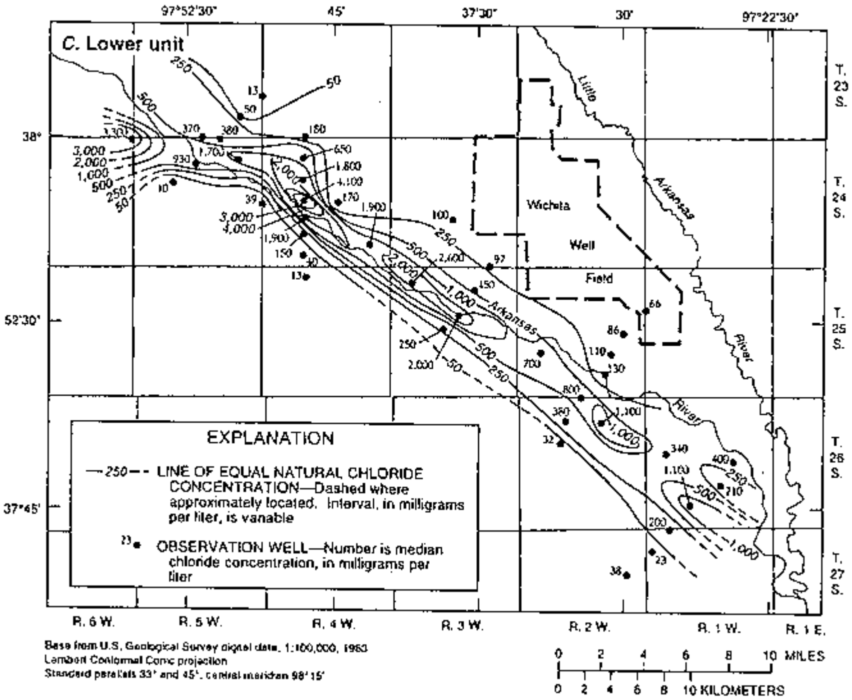


Figure 19. Natural chloride concentrations and lines of equal concentration in (*A*) upper, (*B*) middle, and (*C*) lower units of the *Equus* beds aquifer, 1989–90 [data from Whittemore (1990) and on file with the U.S. Geological Survey, Lawrence, Kans.]—Continued.

0.0012, a hydraulic conductivity of 100 ft/d, and an inflow area of 6,595,000 ft^2, is about 9 ft^3/s. Total subsurface inflow for 1989 conditions thus is estimated to have been about 31 ft^3/s.

Recharge from precipitation occurs over all of the study area except where shale crops out. The amount of water reaching the saturated zone of the aquifer over the long term would be the total amount of precipitation minus the sum of surface runoff and evapotranspiration from the unsaturated zone, assuming no net change in subsurface storage. Results of soil-moisture water-balance computations for an area near Newton indicate that the mean annual recharge to the aquifer ranges from 0.44 to 6.02 in. (about 1.4 to 20 percent of the 1940–89 mean annual precipitation for the study area), depending on the type of vegetative cover (Dugan and Peckenpaugh, 1985, table 3). Assuming a recharge area within the study area of about 438,300 acres, the mean annual recharge thus could range from about 22 to 304 ft^3/s.

Recharge from streams occurs primarily along the Arkansas River. Results of seepage surveys (table 1) show that streamflow losses for the Arkansas River between river mile 800.2 and river mile 772.5 ranged from 0 to 36 ft^3/s, or an average per mile streamflow loss of 0 to 1.3 ft^3/s. Extrapolating this rate of loss to the entire reach of the Arkansas River within the study area, 48.2 river miles, estimated losses could range from 0 to about 63 ft^3/s. Losses probably are less than the maximum of 63 ft^3/s because some reaches of the Arkansas River gain streamflow. Estimates of streamflow loss prior to 1988 would be difficult to make because the gaging station near Maize was installed in 1988 and because of changing ground-water withdrawal patterns and amounts.

Irrigation water would recharge the aquifer if the amount of water applied exceeded the consumptive irrigation requirements of crops. During 1989 and 1990, the mean depth of irrigation water applied to crops within the study area was 10.8 and 13.6 in., respectively, for a 2-year mean of 12.2 in. (Kansas Department of Agriculture and Kansas Water Office, 1990). Assuming that some of this water would evaporate before infiltrating the ground, less than 12.2 in. of water actually would be available to the irrigated crops. Consumptive irrigation requirements of row crops and alfalfa near Newton were calculated to be 11.41 and 13.68 in/yr, respectively (Dugan and Peckenpaugh, 1985, table 3). On the basis of this data and assuming appropriate timing of applications, the amount of irrigation return flow to the aquifer would be negligible.

Discharge

Discharge from the aquifer is from subsurface outflow, evapotranspiration, streamflow gains, and gronnd-water withdrawals from wells. On the basis of the 1989 potentiometric surface (fig. 14), subsurface outflow probably occnrs along parts of the northeast and southeast sides of the study area. Estimated outflow along the northeast side, assuming a hydraulic gradient perpendicular to the study-area boundary of 0.0011, a hydraulic conductivity of 150 ft/d, and an outflow area of 5,240,000 ft^2, is about 10 ft^3/s. Estimated outflow along the southeast side, assnming a hydraulic gradient of 0.0012, a hydraulic conductivity of 500 ft/d, and an outflow area of 4,371,000 ft^2, is about 30 ft^3/s. Total subsurface outflow for 1989 conditions thus is estimated to have been about 40 ft^3/s.

Discharge by evapotranspiration occurs over all the study area. Evapotranspiration discharge may be separated into two components—that which comes from the unsaturated zone and that which comes from the saturated zone by phreatophytic consnmption. Evapotranspiration from the unsaturated zone, although strictly speaking not a discharge from the aquifer, intercepts water that might have otherwise percolated down to the saturated zone. In this conceptual model, evapotranspiration from the unsaturated zone is accounted for in the recharge amount (see preceding "Recharge" section). No information on phreatophyte evapotranspiration rates is available for the study area. However, phreatophytic consumption probably occurs near streams and lakes where the water table is close to the land surface. Evapotranspiration discharges from both the saturated and unsaturated zones vary seasonally due to climate and plant demands.

Discharge to streams occurs primarily along the Little Arkansas River. Results of seepage surveys of the Little Arkansas River completed during the late 1960's and early 1970's (table 1) show that streamflow gains between river mile 68.1 and river mile 17.5 ranged from 23.45 to 76.6 ft^3/s, or an average per mile of 0.46 to 1.5 ft^3/s. By extrapolating this rate of gain to the entire reach of the Little Arkansas River within the study area, 54.0 river miles, estimated gains could range from about 25 to 81 ft^3/s. The range in streamflow gain in 1989 probably was less because ground-water withdrawals from the aquifer near the Little Arkansas River have increased since the early 1970's, consequently causing a decrease in the hydraulic gradient towards the river and a decrease in streamflow gain.

Discharge from the aquifer by ground-water withdrawals from wells occurs throughout the stndy area. Municipal and industrial ground-water withdrawals occur in localized areas, whereas irrigation ground-water withdrawals in 1989 were distributed fairly uniformly over the study area. Total gronnd-water withdrawals dnring 1989 in the model area were about 116,265 acre-ft/yr or about 160.6 ft^3/s.

SIMULATION OF STREAM-AQUIFER INTERACTION

A three-dimensional, finite-difference, flow-model program, MODFLOW (McDonald and Harbaugh, 1988), was used to simulate ground-water flow, surface-water flow, and stream-aquifer interaction. It is common to speak of both the computer program and the data sets that represent the stream-aquifer system as "models." In this report, the computer program will be referred to as the flow-model program or flow-model modules, and the data sets, which represent the stream-aquifer system, will be referred to as the ground-water flow model or model.

Ground-water flow models, by definition, are simplifications of the actual stream-aquifer system and embody certain assumptions. Assumptions for MODFLOW are:

(1) The density of water is uniform, is that of freshwater, and is not affected by temperature; the solute-concentration effect is neglible.

(2) Aquifer properties and stresses are distributed uniformly within a model cell and are constant during a stress period.

(3) The effects of aquifer stresses beyond the model boundaries are neglible.

(4) Tops and bottoms of model cells are horizontal, and the sides of cells are vertical.

(5) Stream leakage to and from the aquifer is vertical.

McDonald and Harbaugh (1988) discuss model theory, mathematical treatment of simulated conditions, and solution techniques used in the MODFLOW program.

The ground-water flow model was developed in two stages. First, a steady-state flow model was developed to simulate aquifer conditions existing in the study area during the late 1930's. (Prior to 1940, there were no major pumpage centers in the study area.) Second, a transient model was developed to simulate aquifer conditions for 1940–89. The hydraulic-head distribution generated by the steady-state model was used as the initial hydraulic-head distribution for the transient model. The transient model also was used for simulations using hypothetical conditions for 1990–2019. Data sets for the steady-state and transient (1940–89) models are available on magnetic tape or disk from the USGS in Lawrence, Kans.

Steady-State Ground-Water Flow Model

The steady-state ground-water flow model can be described in terms of its geometry, simulated aquifer properties, and simulated stresses. The steady-state model was assigned one stress period during which the geometry, aquifer properties, and stresses were held constant, and the aquifer-storage change was assumed to be zero.

Model Geometry

Model geometry was determined by the focus of the study, the area of interest, natural boundaries, aquifer thickness, and aquifer stratigraphy. Because the focus of this study was the Arkansas River and the adjacent *Equus* beds aquifer in the area between Hutchinson and Wichita, a model grid was laid out with rows parallel to the river, and with a small grid spacing near the river (fig. 20). The model consists of 34 rows, 42 columns, and three layers for a total of 4,284 model cells. The upper, middle, and lower model layers correspond to the upper, middle, and lower units of the *Equus* beds aquifer. The row and column spacings for all three model layers were identical. The model grid was made large enough to take advantage of natural barriers to ground-water flow, such as the contact between shale and *Equus* beds sediment, and to encompass the Wichita well field and the Little Arkansas River, which are major stresses in the ground-water flow system. Row spacing was varied to provide detail near the river and yet to minimize the total number of grid cells in the model. Northeast from the Arkansas River, row spacings were 1,000, 2,000, 5,000, and 10,000 ft. Southwest from the river, row spacings were 1,000, 2,000, and 5,000 ft. Column spacing was 5,000 ft.

Various boundaries affect the geometry of the three model layers. No-flow boundaries were simulated, with no-flow cells, where shale provides a natural boundary to ground-water flow southwest of the river from near Hutchinson to near Wichita, northeast of Hutchinson, and north of Wichita (fig. 20). A no-flow boundary also was simulated beneath the *Equus* beds aquifer where shale is considered a relatively impermeable boundary to ground-water flow. Stream cells in the model are where the flux of water to or from the stream is dependent on the difference between hydraulic head in the stream and aquifer and the vertical hydraulic conductivity of the streambed. Stream cells were used to simulate perennial streams (fig. 20*A*). Constant-head cells (constant-head boundaries) were used to simulate ground-water flow into or out of the model area where the *Equus* beds aquifer extends laterally beyond model limits (fig. 20). Water-level data collected during this study indicate that there is little vertical hydraulic-head gradient near the constant-head boundaries. Accordingly, constant-head values used in model simulations were the same in all three layers for a given row-and-column location in the model. The saturated thickness of the aquifer determined the overall thickness simulated by the model, and the stratigraphy determined the thickness simulated for each of the three model layers.

Aquifer Properties

Aquifer properties defined for the steady-state flow model were horizontal and vertical hydraulic conductivity. Specific yield and storage coefficient were assumed to be zero in the steady-state simulation. The model developed by Spinazola and others (1985), which represented the *Equus* beds aquifer with a single layer, was the primary source of aquifer-property data. Aquifer-test data from Reed and Burnett (1985) and from this study were used to refine aquifer properties for the three model layers. The distribution of horizontal hydraulic conductivity in the model layers is shown in figure 21.

Vertical hydraulic conductivity was specified in the model in terms of a vertical conductance. First, vertical hydraulic conductivity was calculated by

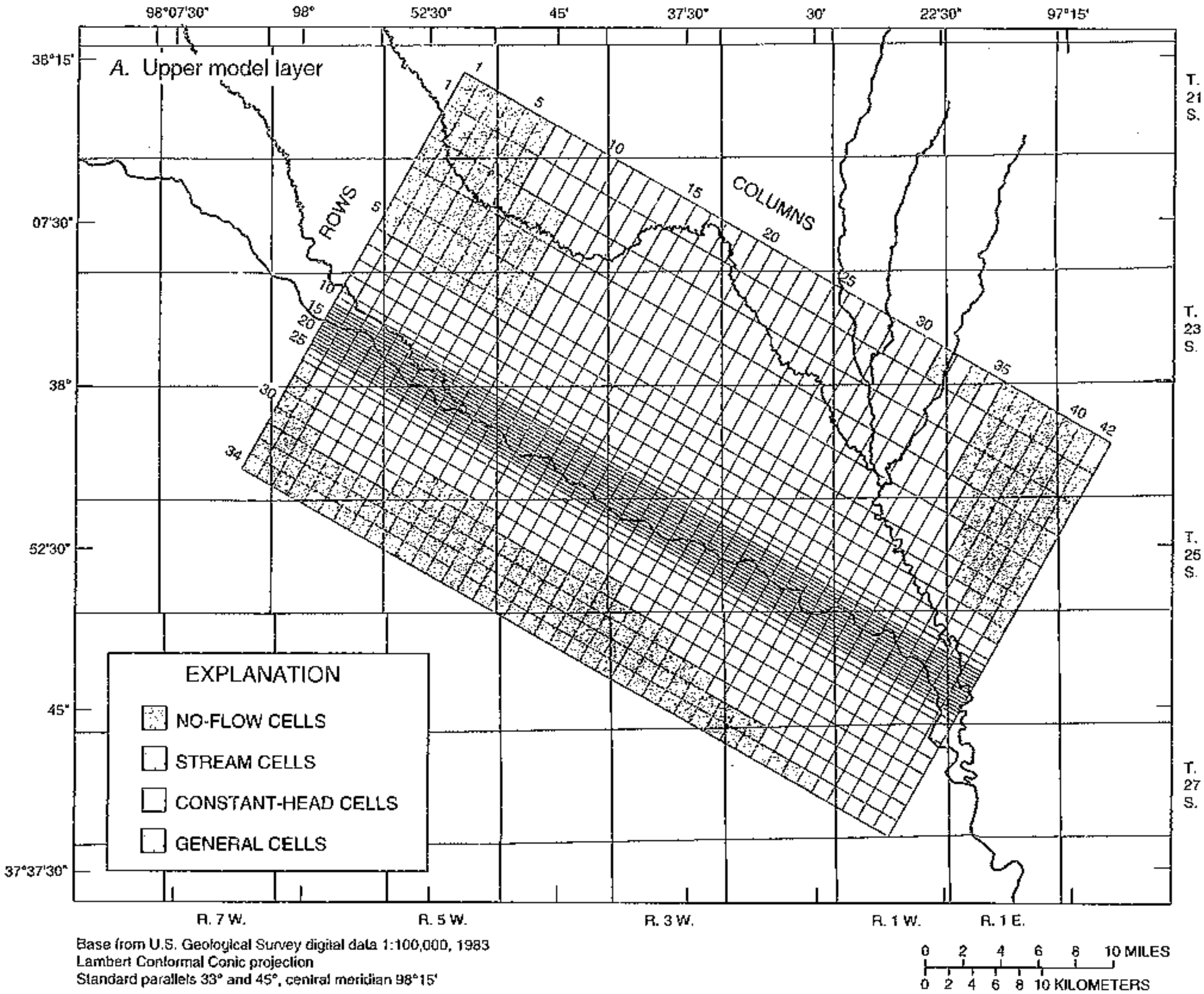


Figure 20. Model grid, row and column numbers, and boundary conditions for (*A*) upper, (*B*) middle, and (*C*) lower model layers.

multiplying an assumed vertical-to-horizontal hydraulic-conductivity ratio (K_v/K_h) of 0.005 times the horizontal hydraulic conductivity for each model cell. Vertical hydraulic conductivity then was used to calculate the vertical conductance between model cells using the formula (McDonald and Harbaugh, 1988, p. 5–13):

$$vcont = \frac{1}{\left(\frac{0.5 x z_k}{K_{v_k}}\right) + \left(\frac{0.5 x z_{k+1}}{K_{v_{k+1}}}\right)}, \qquad (1)$$

where

- $vcont$ = vertical conductance, in day^{-1};
- k = 1, 2, 3 for upper, middle, or lower model layer, respectively;
- z_k = thickness of model cell in layer K, in feet;
- z_{k+1} = thickness of model cell in layer k+1, in feet;
- K_{v_k} = vertical hydraulic conductivity of model cell in layer K, in feet per day; and
- $K_{v_{k+1}}$ = vertical hydraulic conductivity of model cell in layer k+1, in feet per day.

Stresses

Stresses simulated in the steady-state ground-water flow model include recharge, evapotranspiration, streamflow, stream leakage, and pumpage by

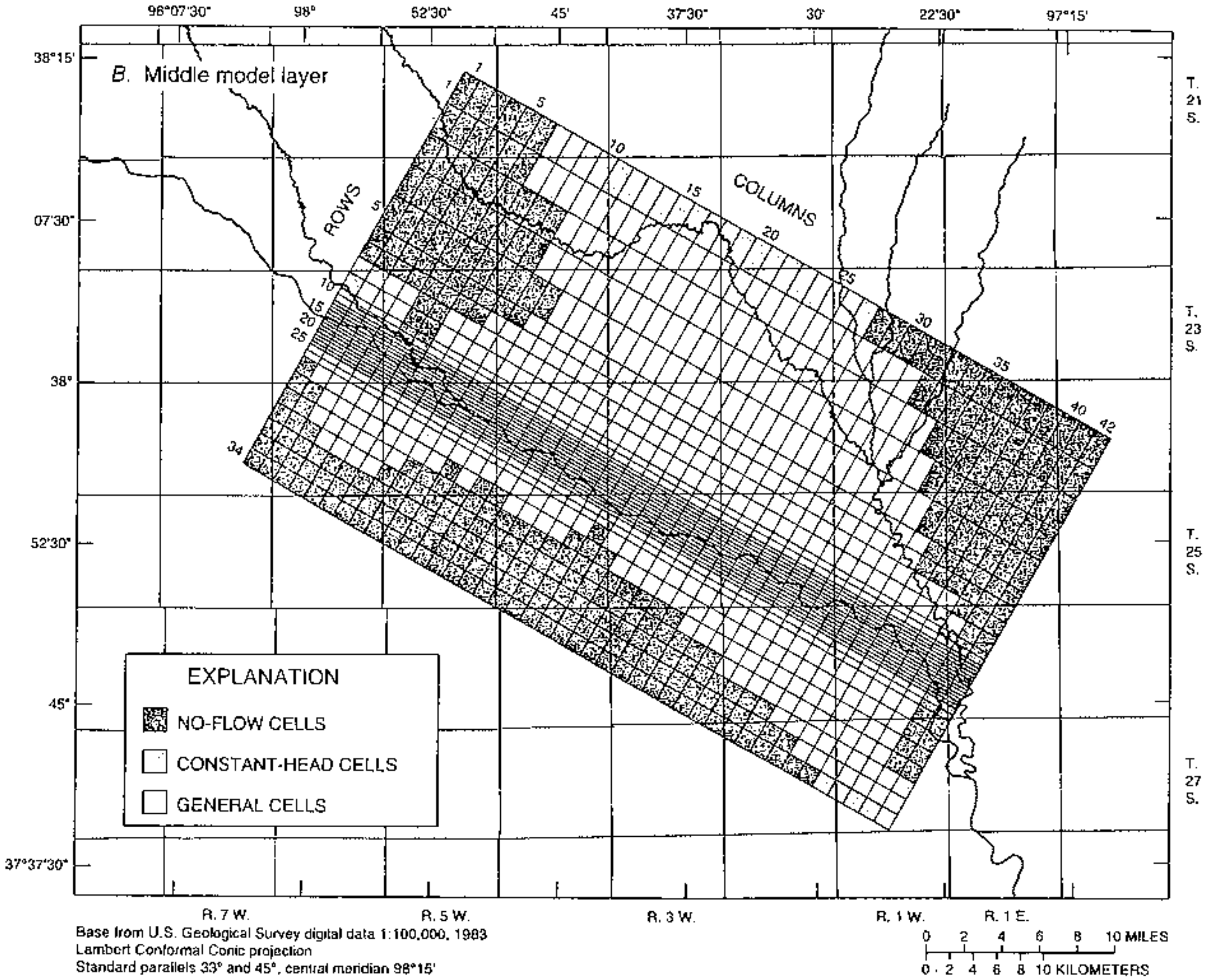


Figure 20. Model grid, row and column numbers, and boundary conditions for (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

wells. Spinazola and others (1985) calculated recharge to their model as a function of 1940 precipitation, the soil type, and thickness of clay in the unsaturated zone. It was assumed that the same recharge values and distribution would reflect 1935–39 conditions and would be appropriate for the steady-state model of this study. Recharge rates used in this model were in the range of 0.1 to 5.5 in/yr (fig. 22).

Evapotranspiration from the ground-water system was simulated in the model. Spinazola and others (1985) arrived at a maximum evapotranspiration rate from the ground-water system of 3.5 in/yr through a trial-aud-error process. A maximum evapotranspiration rate of 3.5 in/yr was used in this model, with a linear decrease in evapotranspiration rate from 3.5 in/yr where the water table is at the land surface to 0 where the water table is 10 ft or more below the land surface.

Streamflow was simulated using a stream-routing module (Prudic, 1988) of the MODFLOW program. An estimated base flow was specified for the starting reach (one reach corresponds to one model cell) of each stream in the model (table 3). The stream-routing module calculated streamflow gain or loss in the remaining reaches on the basis of the difference in hydraulic head between the stream reach and the aquifer and on the basis of a streambed-conductance value specified for each stream reach. The streamflow then was calculated as the algebraic sum of streamflow in the upstream reach and gain or loss in each reach. The streamflow specified for the starting reach of the Arkansas River was based on streamflow records for

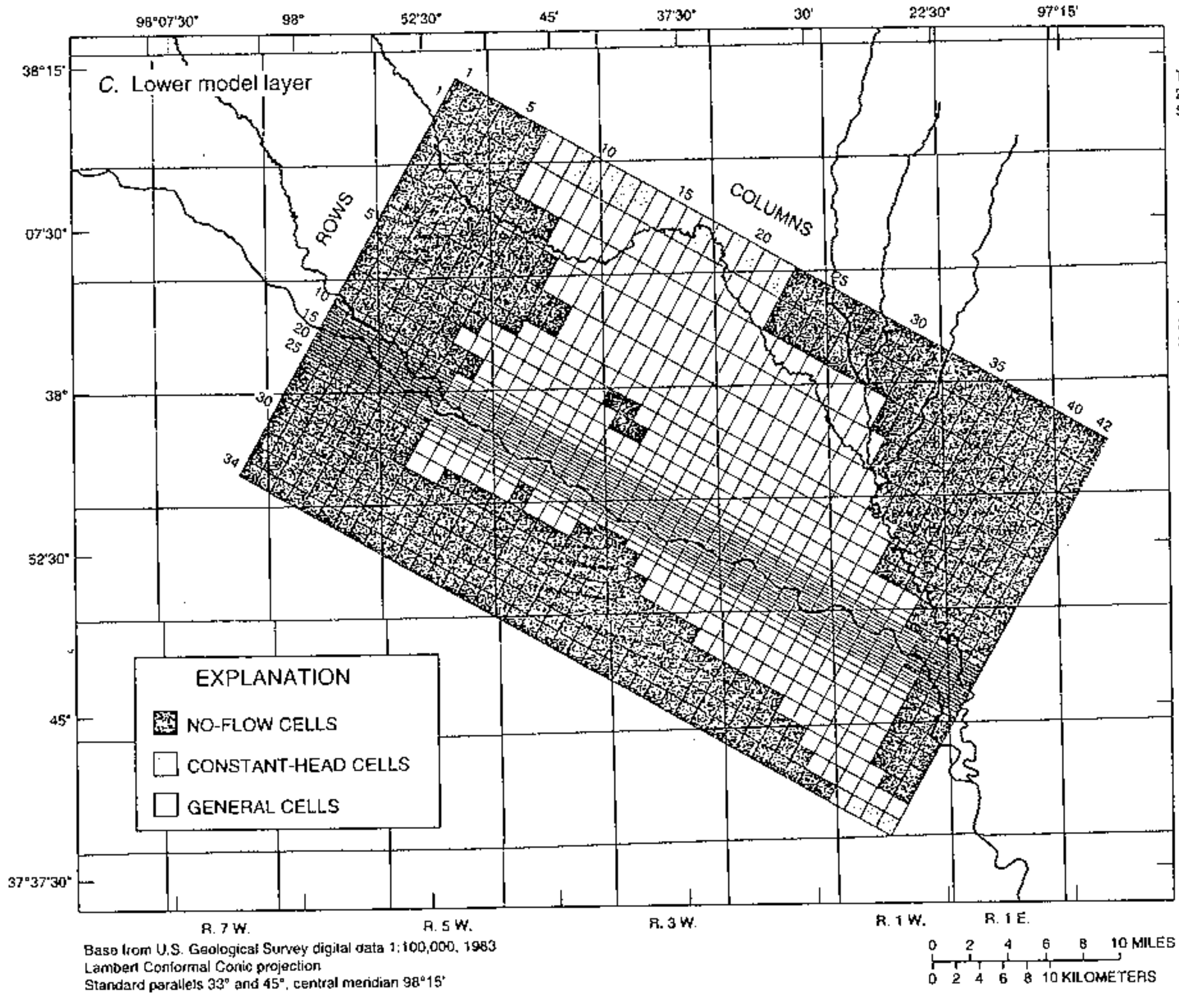


Figure 20. Model grid, row and column numbers, and boundary conditions for (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

the Arkansas River near Hutchinson [period of record, October 1959–present (1992)], the Arkansas River at Wichita (period of record, October 1934–present), and the Little Arkansas River at Valley Center (period of record, June 1922–present).

Streamflow that was exceeded 70 percent of the time, assumed to represent base flow (Hedman and Engel, 1989), was used to simulate streamflow in the Arkansas River. Because of the lack of streamflow data prior to October 1959 for the Arkansas River near Hutchinson, the streamflow in the Arkansas River prior to this time was based on a mathematical relationship between October 1959–89 streamflow data from the Arkansas River near Hutchinson and contemporaneous streamflow data from the Arkansas River at Wichita and the Little Arkansas River at Valley Center. This mathematical relationship and October 1934–39 streamflow data from the Arkansas River at Wichita and the Little Arkansas River at Valley Center were used to estimate streamflow that was exceeded 70 percent of the time in the Arkansas River near Hutchinson for the steady-state model. Streamflow used for the Arkansas River in the steady-state model is smaller than streamflows used in any successive transient- or hypothetical-model simulations (table 3). However, streamflows occurring during the late 1930's reflect the appropriate conditions for the steady-state model of this study. Streamflow for the starting-model reach of the Arkansas River, which was upstream of the gage near Hutchinson, was determined through trial and error so that the simulated streamflow at the location of the Arkansas

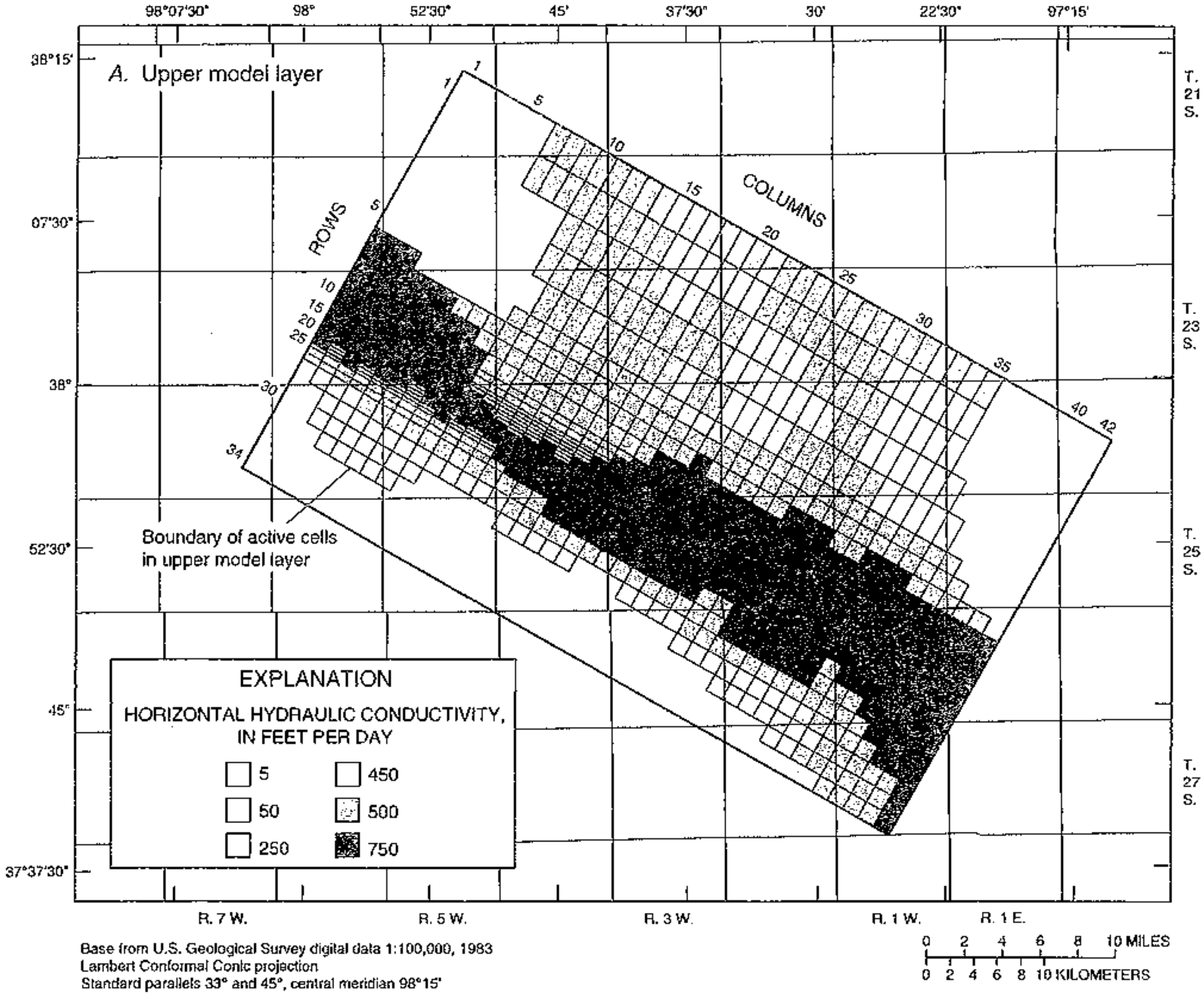


Figure 21. Distribution of horizontal hydraulic conductivity in (*A*) upper, (*B*) middle, and (*C*) lower model layers.

River gage near Hutchinson approximated the estimated streamflow exceeded 70 percent of the time.

Simulated streamflows for the Little Arkansas River were based on seepage-survey data (table 1) and streamflow data for the Little Arkansas River at Valley Center. Seepage-survey data were used to determine an approximate base flow for the starting reach of the river. Then streamflow for the starting-model reach of the Little Arkansas River was determined through trial and error so that the simulated streamflow at the location of the Little Arkansas River at Valley Center gage approximated the measured flow that was exceeded 70 percent of the time. Simulated flows specified for the starting reaches of East Emma, West Emma, and Sand Creeks were approximated on the basis of seepage-survey data. Streamflow in the starting reach of Emma Creek was calculated by the model as the outflow from East and West Emma Creeks.

Stream leakage was simulated by calculating a streambed-conductance term on the basis of the length and width of each stream reach (one stream reach for each model cell), the thickness of the streambed, and the vertical hydraulic conductivity of the streambed, and is expressed by the equation (McDonald and Harbaugh, 1988, p. 6–4):

$$criv = \frac{KLW}{M} \quad , \qquad (2)$$

where

$criv$ = streambed conductance, in feet squared per day;

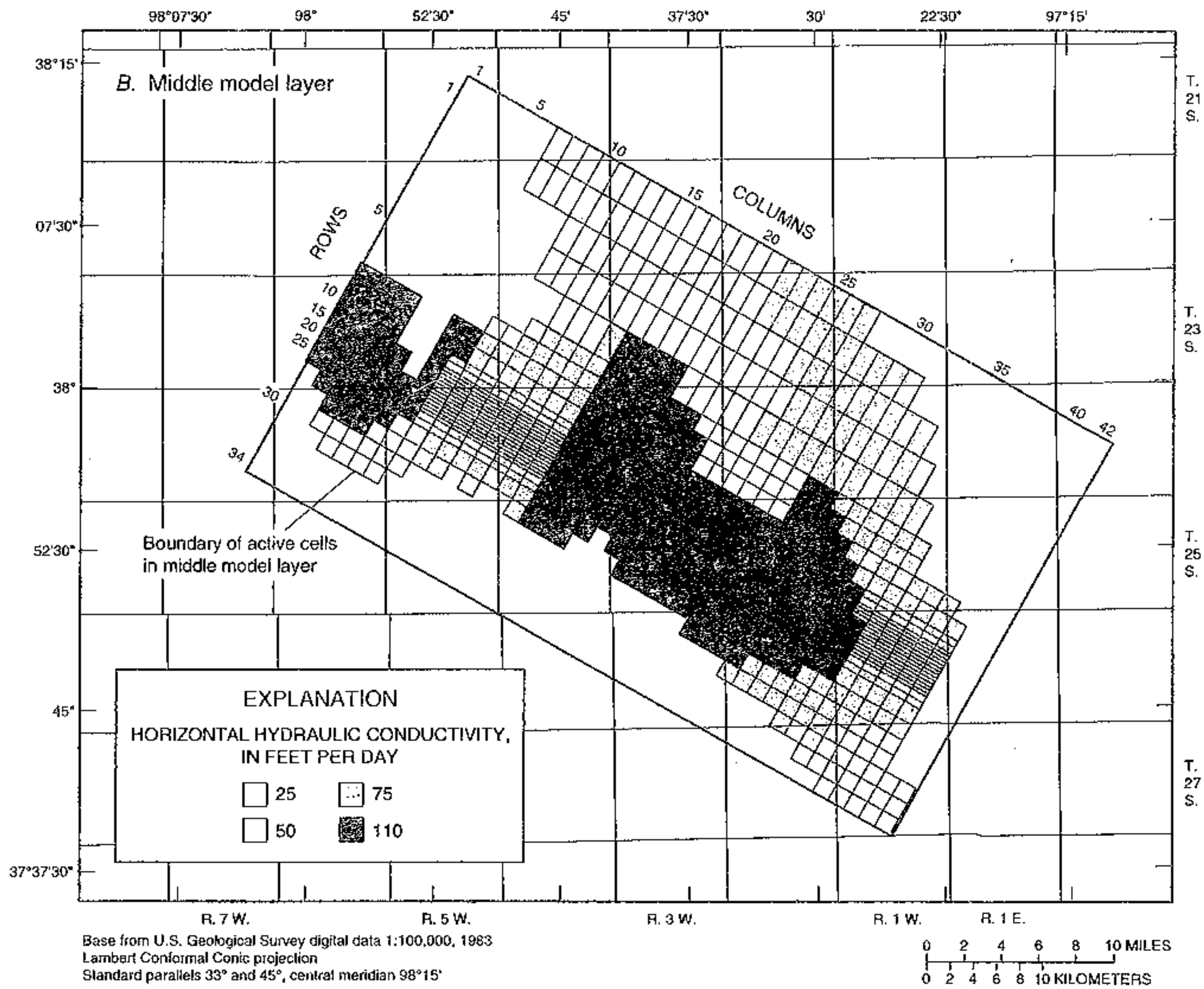


Figure 21. Distribution of horizontal hydraulic conductivity in (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

K = vertical hydraulic conductivity of the streambed, in feet per day;

L = length of stream reach, in feet;

W = width of the stream reach, in feet; and

M = thickness of the streambed, in feet.

The length of each stream reach was set equal to the length of the stream in each model cell. The width of the streams was estimated by onsite observation. Because no discrete streambed could be identified, the thickness of the streambeds was set to one-half of the saturated thickness of the upper model layer for each stream cell (McDonald and Harbaugh, 1988, p. 6–5, 6–6). The initial values of vertical hydraulic conductivity of streambeds were assigned assuming that the Arkansas River would have the largest vertical hydraulic conductivity. The final vertical hydraulic-conductivity values used in the model were 0.5 ft/d for Emma, East Emma, West Emma, and Sand Creeks; 5.0 ft/d for the Little Arkansas River; 1.0 ft/d for Cow Creek; and 50 ft/d for the Arkansas River. In addition to these properties, streambed slope, top-of-streambed altitude, bottom-of-streambed altitude, and a streambed-roughness coefficient were used by the flow-model program to calculate the flow and stream stage in each stream cell. Streambed slope and top-of-streambed altitude were determined from USGS 7 1/2-minute topographic maps. Assuming that there has not been any significant aggradation or degradation of streambed altitude since the topographic maps were made, the streambed altitude used in the model probably is accurate to ±2.5 ft (one-half the contour interval). A streambed-roughness coefficient was

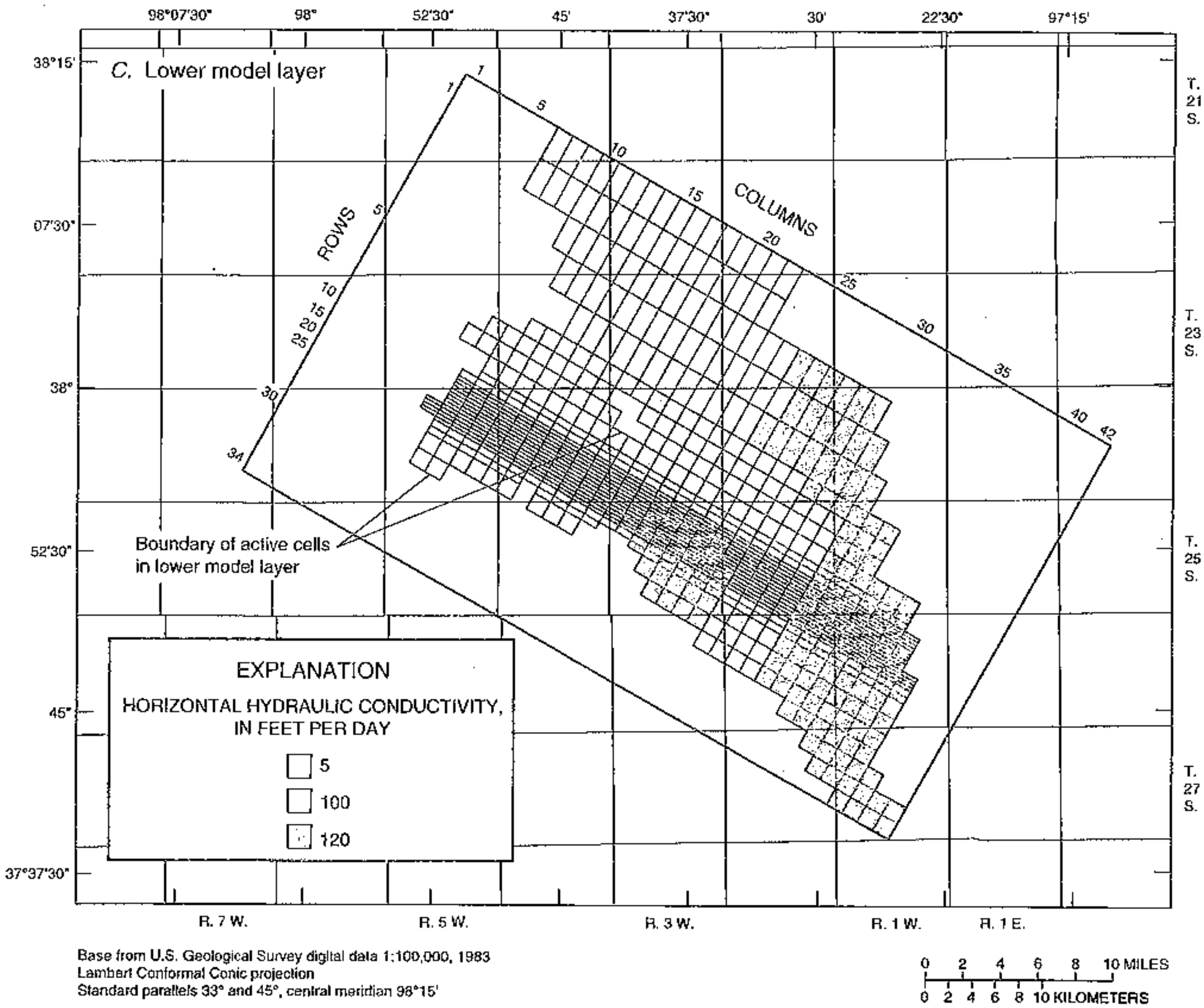


Figure 21. Distribution of horizontal hydraulic conductivity in (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

assigned to streams on the basis of onsite observation of stream channels and a table of Manning's roughness coefficients (Prudic, 1988, p. 10). The roughness coefficient assigned to Cow, Sand, and East and West Emma, and Emma Creeks was 0.03; to the Little Arkansas River, 0.04; and to the Arkansas River, 0.025.

Well pumpage simulated in the steady-state model was relatively small (about 2,066 acre-ft/yr). The location of that pumpage is shown later in the report along with pumpage simulated by the transient model.

Calibration of Steady-State Model

The purpose of calibration is to refine the model so that it is a reasonable representation of the stream-aquifer system. Calibration was done by adjusting the values of recharge, horizontal and vertical hydraulic conductivity, and streambed conductance, within reasonable ranges, to achieve the best fit between simulated and measured hydraulic heads and streamflows. Calibration adjustments later made to the transient model also were applied to the steady-state model. For the steady-state simulation, the simulated potentiometric surface for the middle model layer, assumed to be representative of the potentiometric surface of all three layers, was compared to the 1940 potentiometric surface (fig. 23).

The mean absolute difference between measured hydraulic heads for 235 individual wells and their corresponding layer-2, model-cell simulated hydraulic heads was computed for all and selected

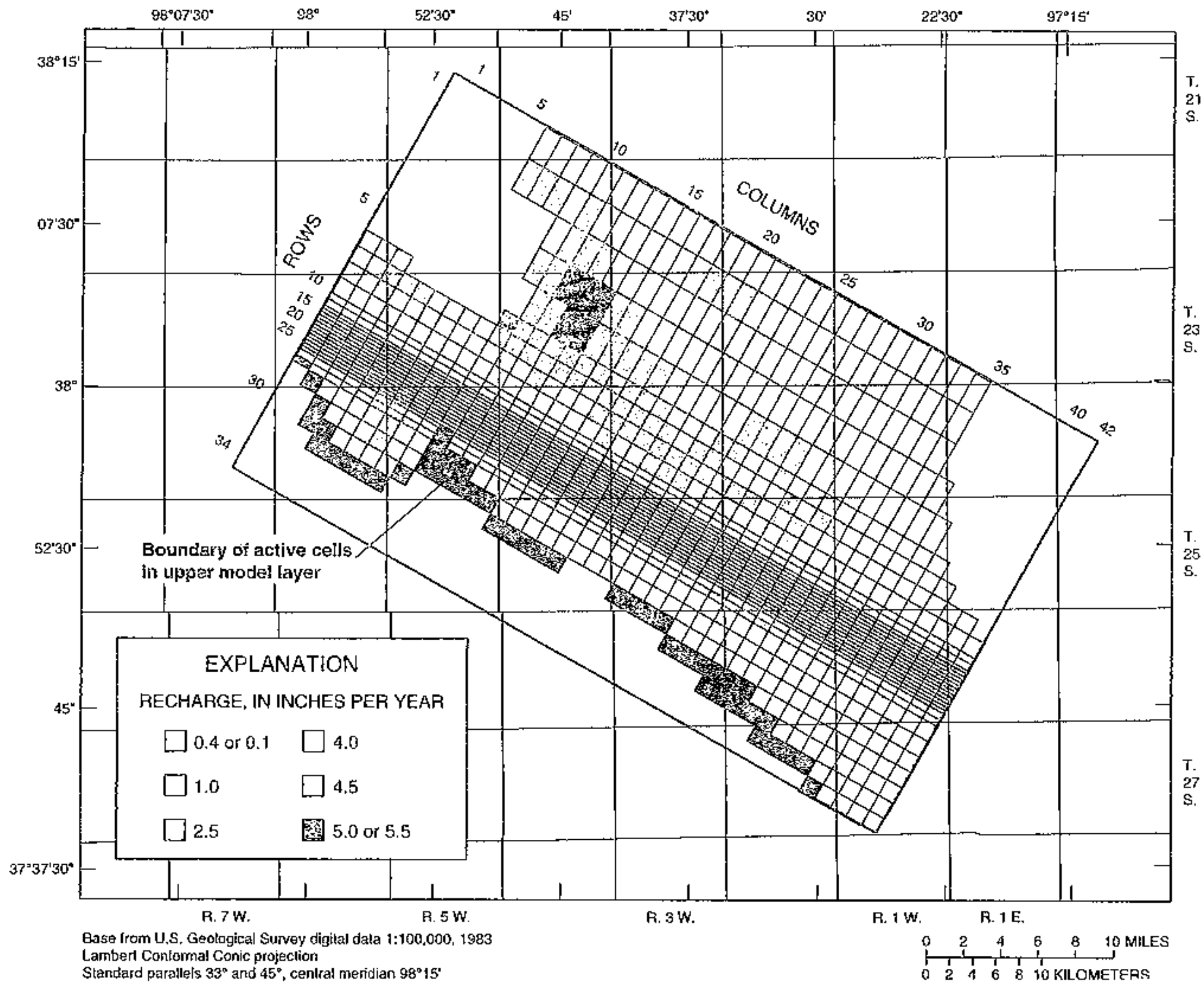


Figure 22. Ground-water recharge rates for steady-state model.

areas of the model (fig. 23). The mean absolute differences were: all of the model, 3.20 ft; area 1, 3.54 ft; area 2, 1.90 ft; area 3, 2.48 ft; area 4, 4.50 ft; and area 5, 2.85 ft.

Prior to 1959, there were no concurrent measurements of streamflow along the streams simulated in the model, so no values of gain or loss in the streams were available for calibration of the steady-state model. Simulated net gains or losses for these streams are shown in table 4.

Sensitivity Analysis

The purpose of sensitivity analysis is to measure how sensitive the model-computed results are to changes in aquifer properties and aquifer stresses. During sensitivity analysis, evapotranspiration, streambed conductance, streamflow, recharge, and hydraulic conductivity were varied from one-half to twice their calibration values. The resulting simulated hydraulic heads were used to calculate the mean absolute deviation from the accepted calibration heads (fig. 24). Changes in the rate of recharge and values of hydraulic conductivity had the most effect on the mean absolute deviation from the accepted calibration hydraulic heads, whereas changes in evapotranspiration, streambed conductance, and streamflow had little effect. Doubling the values of recharge and hydraulic conductivity changed the mean absolute deviation by about 2.9 and 1.4 ft, respectively. These relatively small changes are an indication that water-level changes in the aquifer are constrained by the presence

Table 3. Simulated streamflows, for the starting reach of each stream, used in steady-state and transient models

Name of stream	Layer-1 model cell where streamflow is introduced to model (row,column)	Flows for each stream period, in cubic feet per second: Steady-state model 1935–39	Transient model 1940–52	1953–58	1959–63	1964–70	1971–79	1980–89
Arkansas River	(17,1)	50	365	54	353	186	216	126
Little Arkansas River	(3,10)	2	2	5	1.5	6	4	4
Cow Creek	(11,3)	10	10	10	10	10	10	10
Sand Creek	(1,33)	1	1	1	1	1	1	1
East Emma Creek	(1,28)	1	1	1	1	1	1	1
West Emma Creek	(1,25)	1	1	1	1	1	1	1
Emma Creek[1]								

[1]Water from East Emma and West Emma Creeks join to form Emma Creek.

of the Arkansas River, by the generally shallow depth of the water table below land surface, and by the large hydraulic conductivity of the aquifer material. That is, water levels in the aquifer near the Arkansas River do not decline much below the level of water in the river, and water cannot rise above land surface without running off. Thus, water levels in the aquifer are constrained by natural conditions to a relatively small range. In such a constrained system, a larger range of aquifer properties and stresses will satisfy a given hydraulic-head distribution than in a less-constrained system. The set of data used to represent the stream-aquifer system is not unique but is one of many possible solutions.

Transient Ground-Water Flow Model

Aquifer Properties and Stresses

The transient model had the same aquifer properties as the steady-state model except that specific-yield and storage-coefficient values were included in the simulations. A specific yield of 0.15 was assigned uniformly to the upper model layer. For the middle and lower layers, a specific storage of 0.0001 ft^{-1} was assumed aud multiplied by the layer thickness to get values of storage coefficient for each model cell. Thus, storage coefficients ranged from 0.0004 to 0.014 for the middle layer and 0.0003 to 0.018 for the lower layer.

The transient ground-water flow model also had the same geometry and maximum evapotranspiration rate as the steady-state model, but recharge, streamflow, and pumpage were varied for each stress period, and specific-yield and storage-coefficient values were included in the simulations. Six stress periods were defined for the transient model. The stress periods cover 1940–52, 1953–58, 1959–63, 1964–70, 1971–79, and 1980–89. The first five stress periods were defined by Spinazola and others (1985) on the basis of time periods when there was a relatively uniform trend in well pumpage (fig. 17). During the sixth stress period (1980–89), there were marked fluctuations in the volume of agricultural pumpage (fig. 17), but because the average of each increasing or decreasing trend was about the same, 1980–89 was simulated as one stress period. Stresses were held constant during each stress period but were varied from one stress period to the next.

Stresses in the transient model were defined by using available data. Recharge was based on the mean precipitation at climatic stations at Hutchinson, Mount Hope, and Wichita for each stress period (National Oceanic and Atmospheric Admiuistration, 1935–89). The recharge for each stress period was estimated as follows: (1) The recharge specified for each steady-state model cell was divided by the mean annual precipitation for the pre-1940 period (represented by the steady-state model). (2) The resulting quotient for each model cell then was multiplied by the study-area mean annual precipitation (table 5) for each stress period in the transient model.

Streamflows in the first model reach of the Arkansas River were assigned as the streamflows that were exceeded 70 percent of the time for each stress

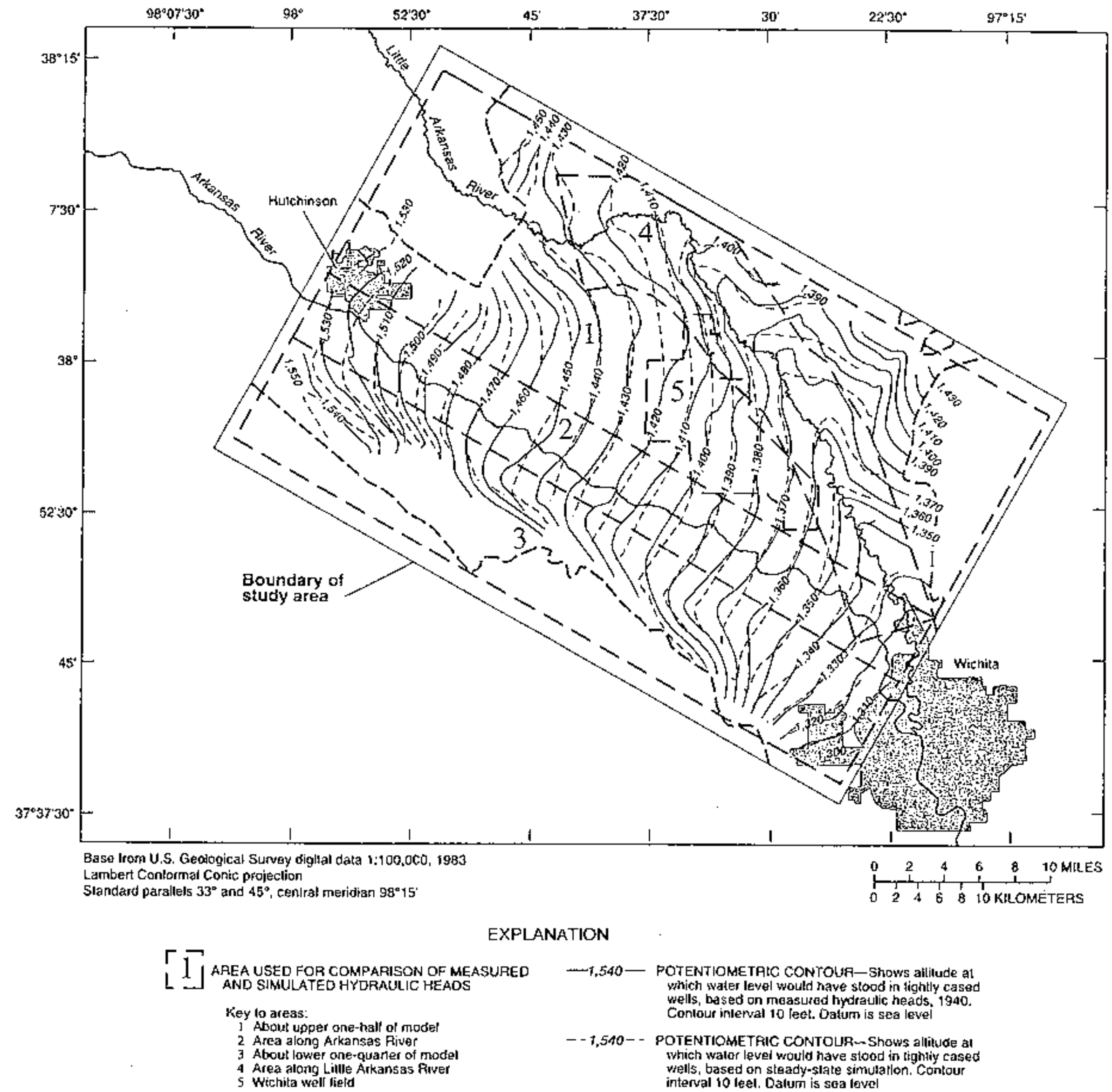


Figure 23. Measured and simulated potentiometric surfaces for middle model layer, 1940 (1940 potentiometric contours modified from Spinazola and others, 1985), and areas used for comparison of measured and simulated hydraulic heads.

period (table 3). For the first two stress periods, streamflow in the Arkansas River near Hutchinson was estimated from streamflow data from gages on the Arkansas River at Wichita and the Little Arkansas River at Valley Center by using the mathematical relation established between streamflow at the gages at Wichita, Valley Center, and the gage near Hutchison. Streamflow in the starting-model reach of the Little Arkansas River was estimated from streamflow data for the Little Arkansas River at Alta Mills and the Little Arkansas River at Valley Center. Streamflows for the starting-model reaches of Cow, Sand, and East and West Emma Creeks for every stress period were assigned the same as the streamflows used in the steady-state model.

Well pumpage for the first five stress periods was taken from model data sets developed by Spinazola and others (1985). Well-pumpage quantity

Table 4. Simulated streamflow at upstream and downstream ends of streams within steady-state model. Except for Emma Creek, upstream-end streamflow is specified in the model-input data sets. The upstream-end streamflow for Emma Creek is calculated by the model

[Flows are in cubic feet per second (ft^3/s)]

Stream	Streamflow at upstream end	Streamflow at downstream end	Net gain (+) or loss (-) of streamflow
Arkansas River	50.0	80.7	+[1]30.7
Little Arkansas River	2.0	73.7	+[2]71.7
Cow Creek	10.0	10.1	+.1
East Emma Creek	1.0	.16	-.84
West Emma Creek	1.0	2.97	+1.97
Emma Creek	3.13	3.95	+.82
Sand Creek	1.0	.67	-.33

[1]Includes tributary inflow from Cow Creek (10.1 ft^3/s).
[2]Includes tributary inflow from Emma and Sand Creeks (4.62 ft^3/s).

and location data for 1989, from the Kansas Department of Agriculture, Division of Water Resources computer files (Topeka), were used to represent pumpage during the sixth stress period. Total well pumpage for 1989 (about 116,265 acre-ft) (fig. 17) was very close to the average for the 1980–89 period (about 116,273 acre-ft). Pumpage was apportioned to the different model layers using the method suggested by McDonald and Harbaugh (1988, p. 8–2). Pumpage for municipal wells in the Wichita well field was apportioned to the three model layers on the basis of the length of well screen in each layer and the assigned hydraulic conductivity of the model layer at each location. Agricultural and industrial pumpage was apportioned to the three model layers on the basis of the assigned thickness and hydraulic conductivity at each pumping location because well-screen length and depth data were not available. Model cells where pumpage was simulated for each stress period are shown in figure 25. A plot of simulated pumpage from each model layer (fig. 26) shows that most of the simulated pumpage was from the upper model layer, and the least was from the middle model layer.

Calibration of Transient Model and Sensitivity Analysis

As a basis for calibration of the transient model, simulated hydraulic heads were compared to measured heads, and simulated streamflow was compared to

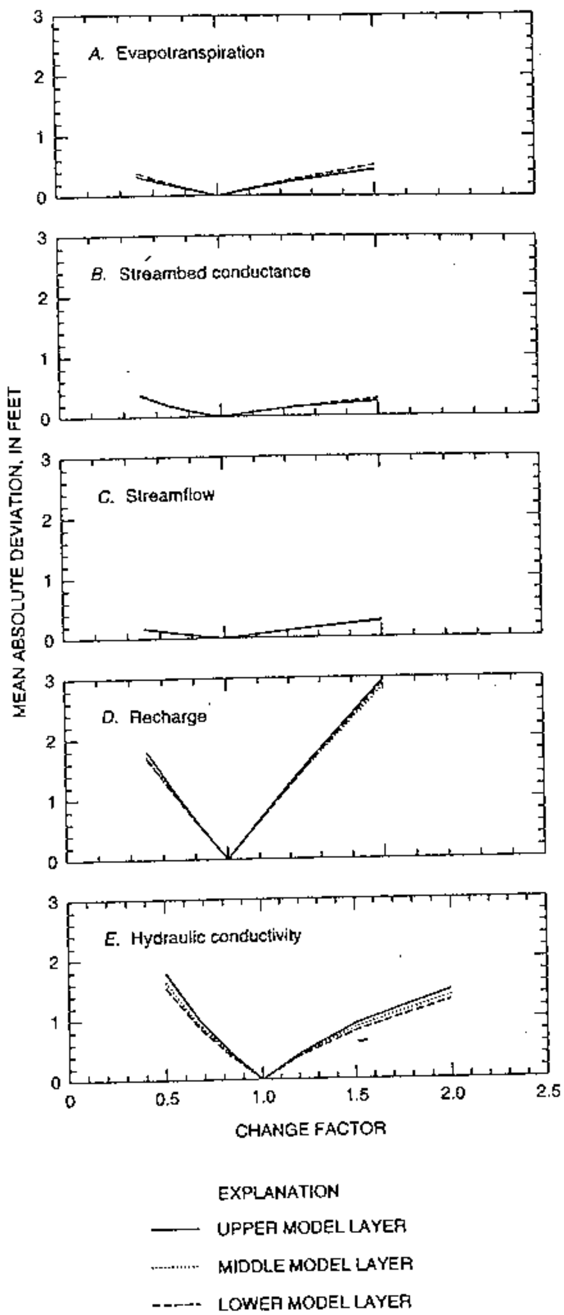


Figure 24. Mean absolute deviations of simulated hydraulic heads from accepted model-calibration heads for changes in (*A*) evapotranspiration, (*B*) streambed conductance, (*C*) streamflow, (*D*) recharge, and (*E*) hydraulic conductivity.

Table 5. Mean annual precipitation at Hutchinson, Mount Hope, and Wichita climatic stations and for the study area for steady-state and transient model stress periods

[Data from National Oceanic and Atmospheric Administration, 1935–89]

Climatic station	Mean precipitation for stress periods, in inches per year						
	Steady-state model	Transient model					
	1935–39	1940–52	1953–58	1959–63	1964–70	1971–79	1980–89
Hutchinson	27.83	[1]34.02	24.22	28.00	26.09	31.53	30.02
Mount Hope	27.87	[2]34.81	26.30	33.73	29.10	32.51	30.73
Wichita	33.42	34.19	23.16	34.06	28.81	29.26	30.52
Mean for the study area	**29.71**	**34.34**	**24.56**	**31.93**	**28.00**	**31.10**	**30.42**

[1]Mean annual precipitation for 1950 estimated on the basis of mean annual precipitation at Mount Hope and Wichita climatic stations.
[2]Mean annual precipitation for 1941–44 estimated on the basis of mean annual precipitation at Hutchinson and Wichita climatic stations.

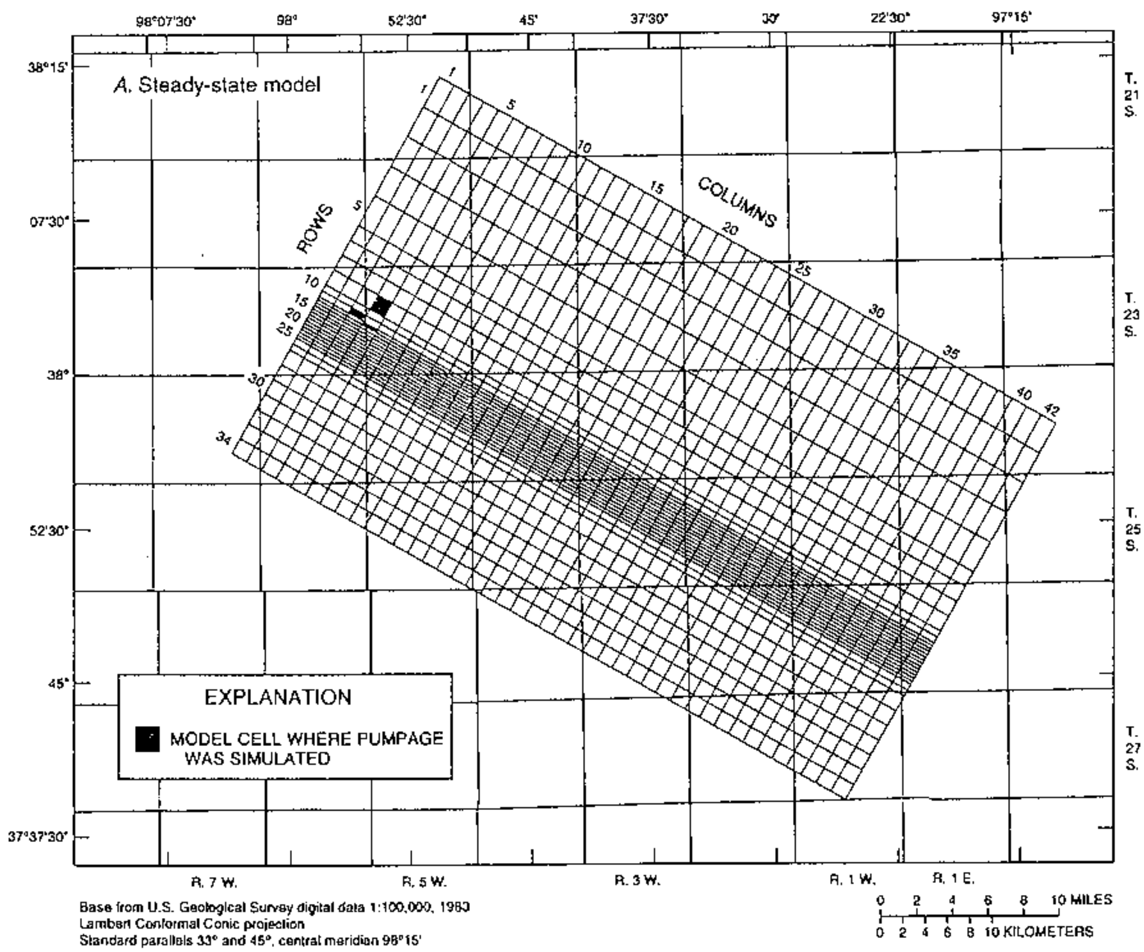


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89.

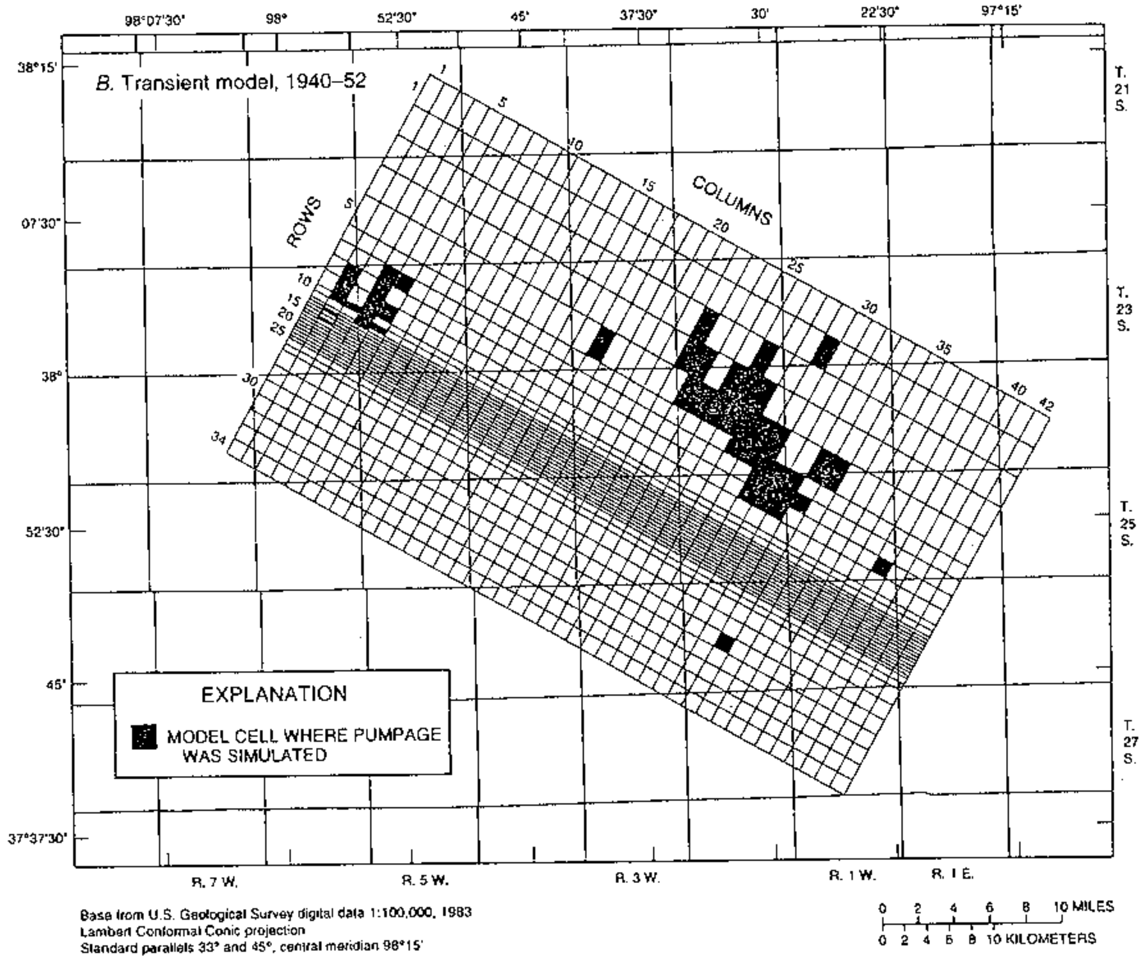


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

historic streamflow data. Simulated middle-layer hydraulic-head distributions were compared to measured head distributions for the end of 1989 (fig. 27). Minor adjustments to model-input hydraulic conductivity and recharge values were necessary before a satisfactory fit was achieved. These changes also were applied to the steady-state model. Simulated hydraulic heads also were compared to water-level hydrographs at 10 well locations (fig. 28). Simulated hydraulic heads follow the long-term trend of measured heads (fig. 28) but do not reflect seasonal or short-term variations in water levels because recharge, pumpage, and streamflows were held constant during each stress period.

The mean absolute difference between hydraulic heads for 232 individual wells and their corresponding middle-layer model cell for the end of 1989 were computed for all of and selected areas of the model (fig. 27). The mean absolute differences were: all of the model area, 4.67 ft; area 1, 5.76 ft; area 2, 2.47 ft; area 3, 2.15 ft; area 4, 4.56 ft; and area 5, 6.76 ft.

Streamflow that was exceeded 70 percent of the time at each gaging station, assumed to represent base flow, was compared to model-simulated flow in the model-stream reach where the gaging station was located (fig. 29). Because streamflows specified for the starting stream reach in the model were held constant for each stress period, the model did not simulate the

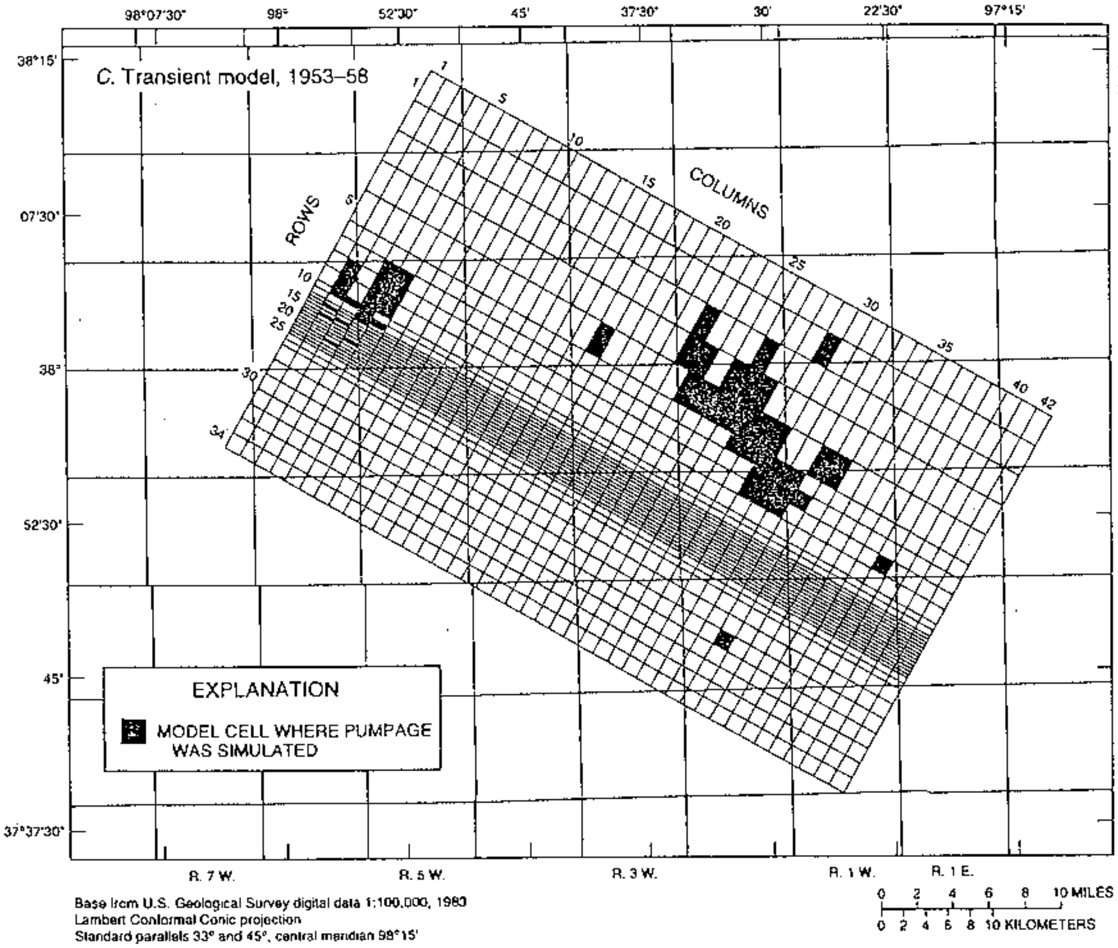


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

annual seasonal or short-term variation of measured streamflow. For most stress periods, the model approximately simulated the annual average base flow for that stress period (fig. 29). Increases or decreases in the rate of streamflow specified for each stress period result in large changes in simulated flow for the Arkansas River at the beginning of each stress period. The gradual changes in simulated streamflow within each stress period are caused by stream-aquifer interaction in the model (fig. 29).

Sensitivity analysis indicated that the transient model was most sensitive to changes in hydraulic conductivity and recharge and least sensitive to changes in streamflow, storage coefficient, and streambed conductance. Sensitivity analysis was done by comparing the mean absolute deviation of simulated hydraulic heads to the accepted transient-model calibration heads (fig. 30). A separate, complete, transient simulation (1940–89) was made for each property changed. A change factor of 0.83 or 1.2 times for hydraulic conductivity and recharge changed the mean absolute deviation from the calibration heads by about 0.45 and 0.18 ft, respectively (fig. 30).

Water Budgets

Water budgets for the ground-water flow model (table 6) show the inflow to and outflow from the model at the end of steady-state and transient stress

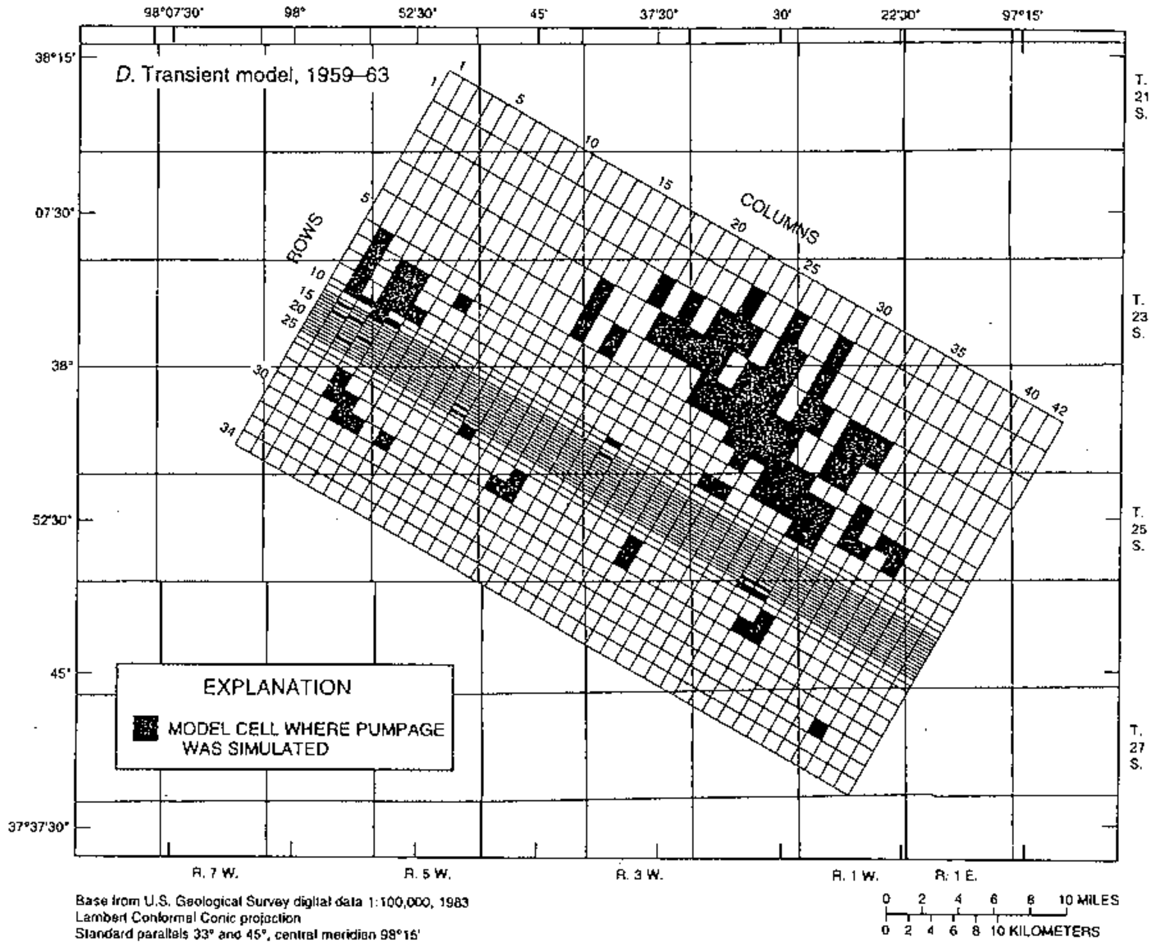


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

periods. In general, from 1940 to 1989 there were appreciable increases of boundary inflow in the upper layer, streamflow loss in the upper layer, and well pumpage in all layers; decreases of aquifer storage in the upper layer, boundary outflow in the upper layer, streamflow gain in the upper layer, and evapotranspiration in the upper layer; and increases of leakage from the upper layer to the middle layer, and from the middle layer to the lower layer. The increases of boundary inflow and streamflow loss, the decreases in boundary outflow, evapotranspiration, and streamflow gain, and the net change of storuge resulted in a net increase of water available to the model of about 160 ft^3/s. Almost all of this net increase of water is accounted for by the 1940–89 increase in well withdrawals of about 158 ft^3/s. The increasing leakage to the middle and lower layers is due to increasing well pumpage from these layers.

Parameters for Simulations of Hypothetical Conditions

A series of transient simulations were used to estimate the possible effect of changing recharge, streamflow, and pumpage on ground-water levels, streamflow, the volume of water exchanged between the Arkansas River and the *Equus* beds aquifer, the quantity of chloride lost from the river to the aquifer and the distribution in the aquifer of chloride originating from the river. All boundary conditions

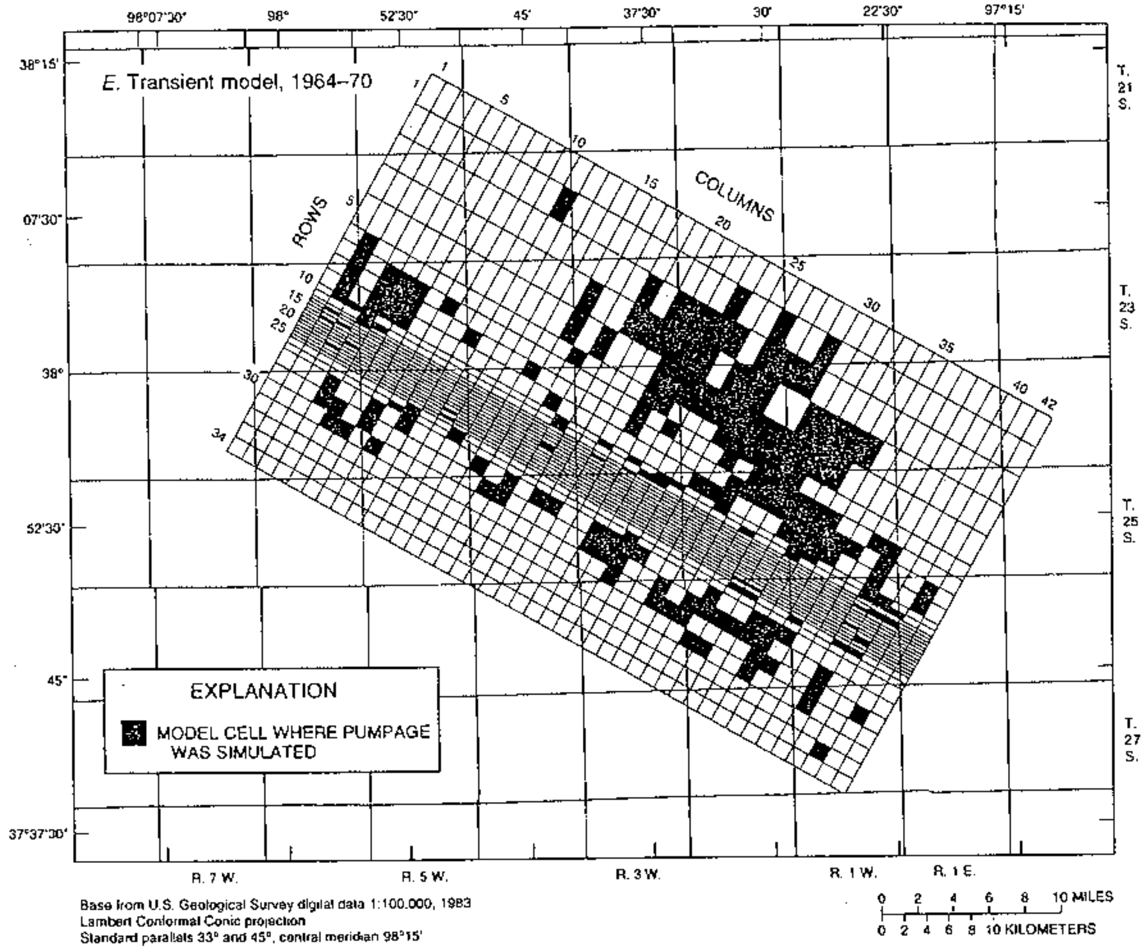


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

and aquifer properties were the same as those used in the transient model (1940–89). Recharge, streamflow, and pumpage were varied in these simulations of hypothetical conditions.

Recharge was varied to simulate dry, normal, and wet climatic conditions. Small and large values of recharge were based on the driest and wettest 10-year periods between 1940 and 1989 that were recorded at the Hutchinson, Mount Hope, and Wichita climatological stations. The average annual precipitation for the driest 10-year period, 1952–61, was 25.53 in. The average annual precipitation for the wettest 10-years, 1942–51, was 35.68 in. The average precipitation, based on the precipitation for the 1940–89 period, was 30.67 in. (National Oceanic and Atmospheric Administration, 1935–89). Thus, precipitation during the driest 10-year period was about 83 percent of the average precipitation. Precipitation during the wettest 10-year period was about 116 percent of the average precipitation. Recharge during dry and wet periods was assumed to differ from the average recharge by these same percentages.

Streamflow was varied to simulate periods of sustained low, average, and high streamflow. Small and large values of streamflow were based on the 10-year period with the smallest and largest flows exceeded 70 percent of the time in the Arkansas River near Hutchinson. For the period of no record at this site (prior to 1959), streamflows were extrapolated from records for the Arkansas River at Wichita and the

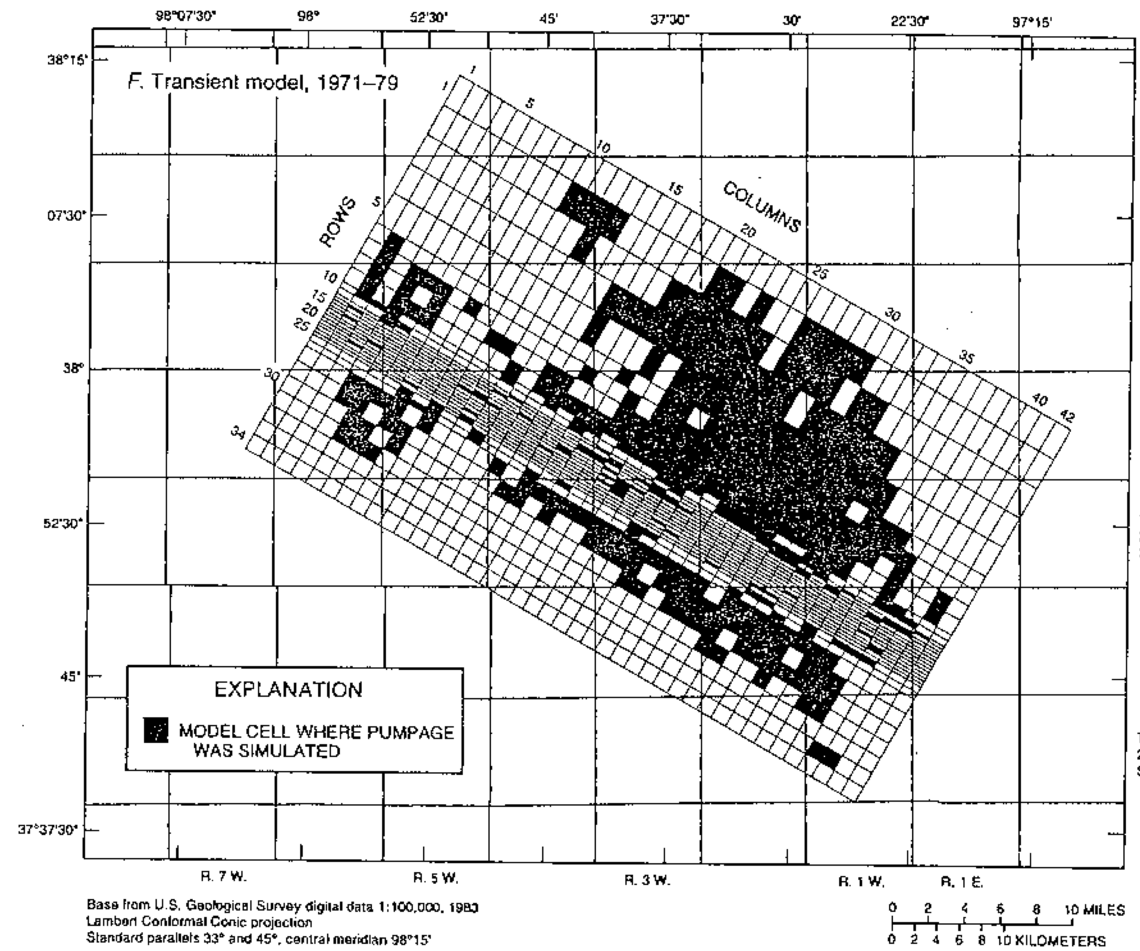


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

Little Arkansas River at Valley Center. The smallest 10-year-period streamflow that was exceeded 70 percent of the time was 75 ft^3/s, for 1934–43. The largest 10-year-period streamflow that was exceeded 70 percent of the time was 390 ft^3/s, for 1942–51. The average streamflow that was exceeded 70 percent of the time was taken as the average of the smallest 10-year period and largest 10-year-period streamflows, about 230 ft^3/s.

Pumpage in the simulations of hypothetical conditions was varied from no change, to a 1-percent per year increase of 1989 pumpage, a 2-percent per year increase of 1989 pumpage, and a 3-percent per year increase of 1989 pumpage. Hereinafter, the three percentage options will be referenced as the 1-percent per year, 2-percent per year, and 3-percent per yea increases.

Least-squares regression analysis of pumpa data indicated that from 1960–89 pumpage volum increased by about 2,800 acre-ft or 2.4 percent pe year of 1989 pumpage. Thus, the 1-, 2-, and 3-per per year increases used in the hypothetical simulati represent relatively small, average, and large pump increases. The pumpage increases were assigned t model cells where ground-water withdrawals alrea were occurring in 1989. Therefore, the increases o pumpage may be unrealistic in some areas that alre have reached the maximum pumpage allocation allowed by current (1992) law, such as parts of the Wichita well field.

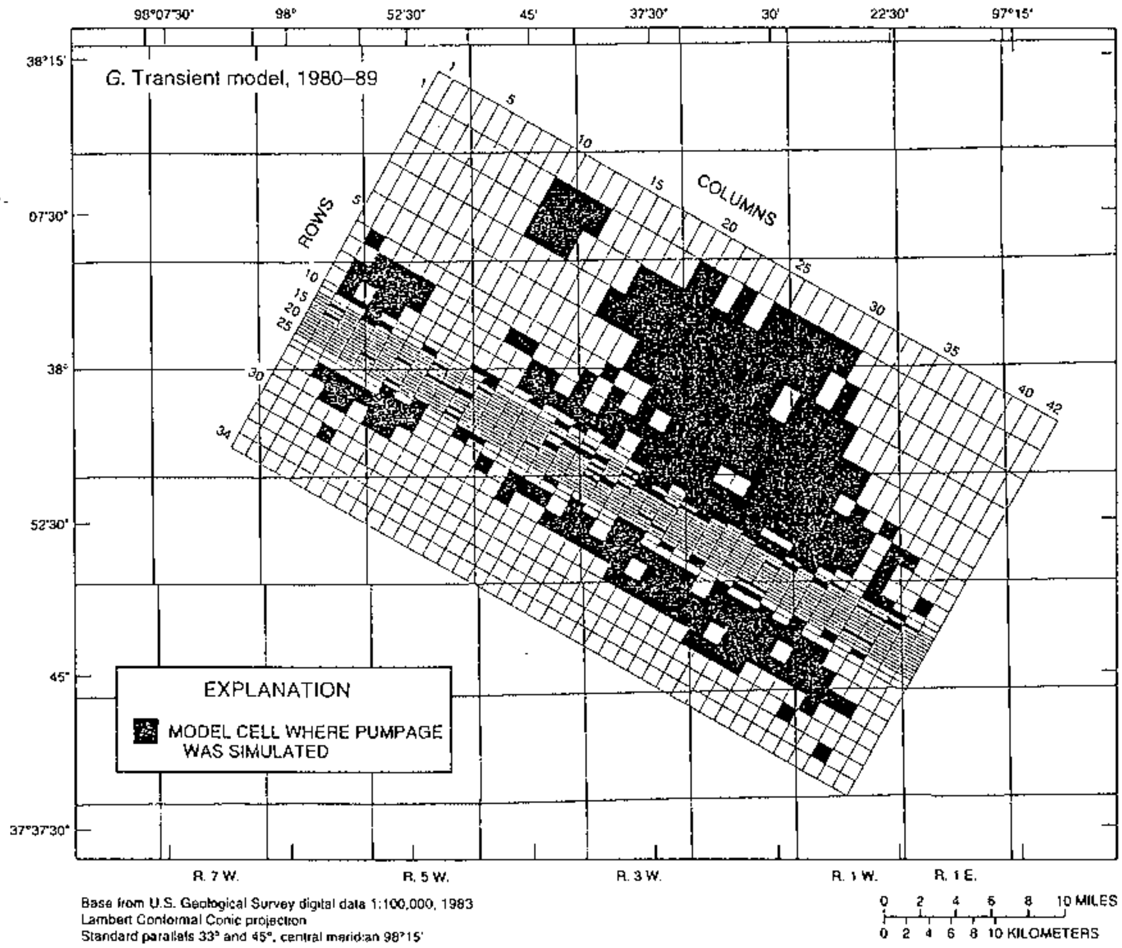


Figure 25. Model cells where pumpage from any of the three model layers was simulated in (*A*) steady-state model and in transient model for (*B*) 1940–52, (*C*) 1953–58, (*D*) 1959–63, (*E*) 1964–70, (*F*) 1971–79, and (*G*) 1980–89—Continued.

A total of 36 simulations were used for the combinations of three different recharge conditions, three different streamflow conditions, and four different pumpage conditions. The simulations were divided into three 10-year stress periods. Each stress period was divided into 10 time steps. Recharge and streamflow were held constant for each simulation, and except for the simulations of no change from 1989 conditions, pumpage was increased by 1, or 2, or 3 percent per year of 1989 pumpage. Thus, at the end of a 30-year simulation, a 1-percent per year increase in pumpage represented a 30-percent increase in pumpage over 1989 pumpage, a 2-percent per year increase in pumpage represents a 60-percent increase over 1989 pumpage; and a 3-percent per year increase in pumpage represents a 90-percent increase over 1989 pumpage.

The actual amount of pumpage simulated was less than 1-, 2-, or 3-percent per year for some of the simulations (fig. 31) when upper layer model cells were simulated as going dry. Model cells go dry when the simulated water level declines below the bottom of the cell. When this happens the stresses for that model cell, including pumpage, no longer are included in model calculations, so the total simulated pumpage is less than the amount in the input data sets. The decrease of pumpage in the model is similar to decline in the productivity of wells if water levels in the aquifer decline.

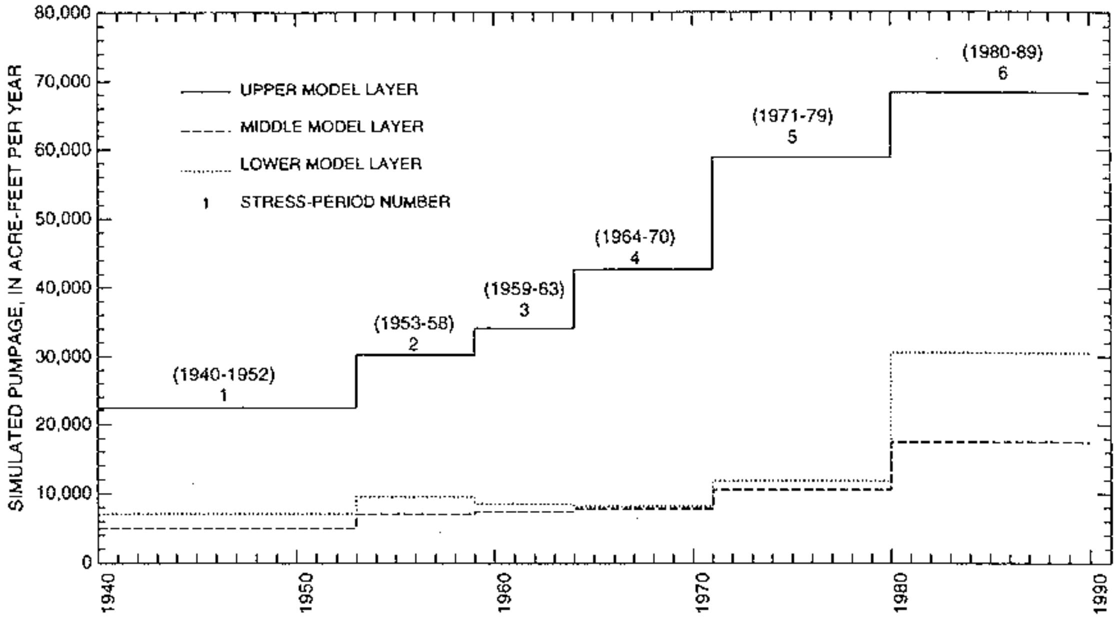


Figure 26. Simulated pumpage used in transient model for each model layer.

Discussion of Ground-Water Flow-Model Results

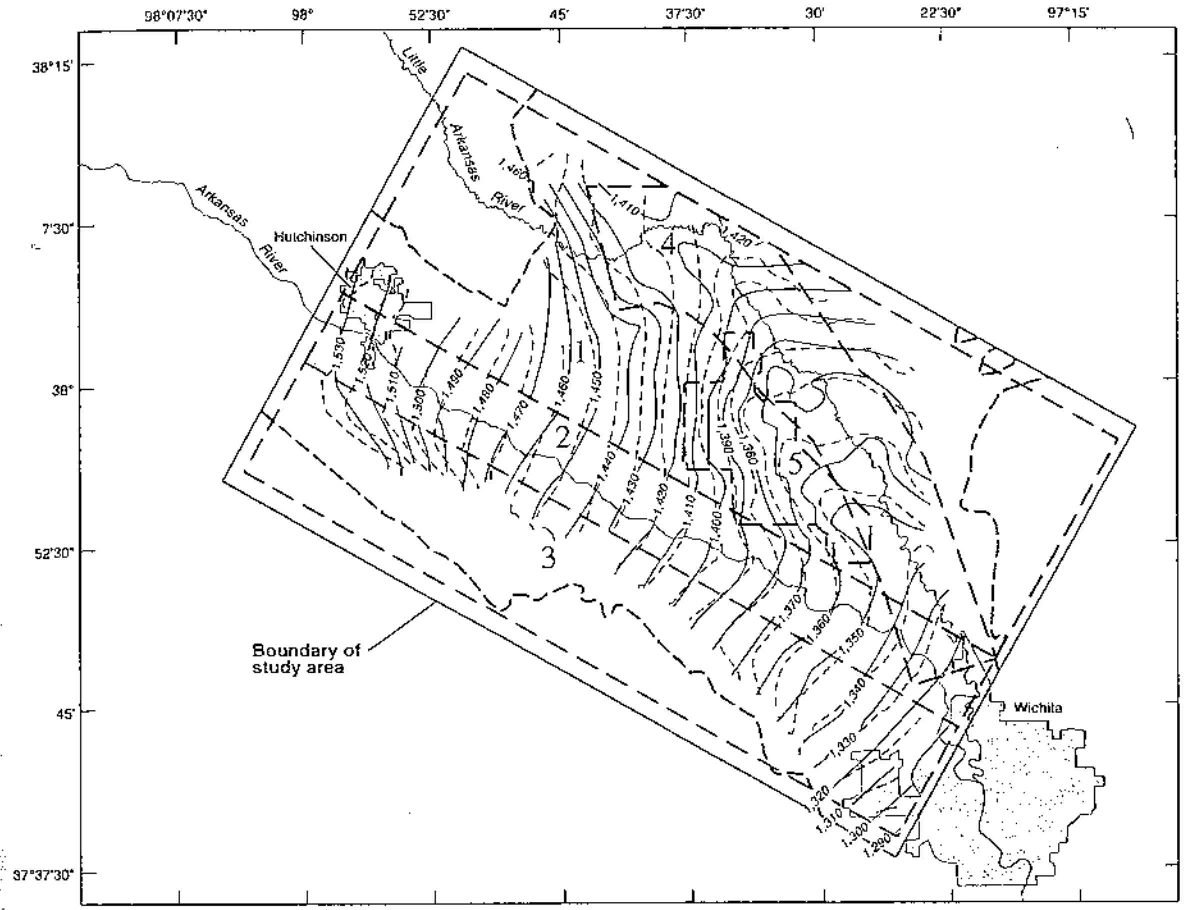
Discussion of flow-model results will focus on two transient-simulation periods—the simulation of historical conditions (1940–89) and the simulation of hypothetical conditions for 1990–2019.

Steady-State and Transient Simulations, 1940–89

During 1940–89, the Arkansas River, acting as a relatively constant-head source, has maintained aquifer water levels near the river at close to their 1940 levels. Measured water-level data and model results indicate that water levels in most of the *Equus* beds aquifer away from the river have experienced long-term declines since about 1940 (fig. 28) because of increasing ground-water pumpage from the aquifer. In part of the Wichita well field area, water levels have declined more than 30 ft (fig. 28, well 24S02W08DBB01). Near the Arkansas River, however, water levels have declined only about 1 ft (fig. 28, well 25S03W09CCC01) since the 1940's. Water-level declines in the aquifer near the Arkansas River were moderated by increasing streamflow losses from the river.

Declining water levels in the *Equus* beds aquifer during 1940–89 have resulted in decreased base flow in the Arkansas and Little Arkansas Rivers. In 19 the Arkansas River had a simulated cumulative lo about 12 ft^3/s and a simulated cumulative gaiu of about 33 ft^3/s for the stream reach within the moc area, for a net cumulative gain of about 21 ft^3/s (fig. 32*A*). Since 1940, loss from the river has increased. At the end of 1989, the simulated cumı tive loss was about 59 ft^3/s and the gain about 7 f for a net cumulative loss of about 52 ft^3/s (fig. 32 stress period 6). In 1940, the Little Arkansas Rivei a simulated net cumulative gain of about 67 ft^3/s the river reach within the model area. By the end 1989, simulated net cumulative gain in the Little Arkansas River had decreased to about 27 ft^3/s (fig. 32*B*, stress period 6).

During 1940–89, the quantity of chloride d charged from the Arkansas River to the *Equus* be aquifer increased (fig. 33) in direct proportion to ı volume of water loss from the river to the aquifer Calculation of chloride discharge was based on simulated water loss from the river to the aquifer on the median chloride concentration of 630 mg/l measured in Arkansas River water samples collec during this study. The median chloride concentrat may have been larger or smaller in the past. The calculated cumulative load of chloride discharged from the river to the aquifer in 1940 was about 21 ton/d within the model area (fig. 33). At the en of 1989, the calculated load of chloride discharge



EXPLANATION

[1] AREA USED FOR COMPARISON OF MEASURED AND SIMULATED HYDRAULIC HEADS

Key to areas:
1 About upper one-half of model
2 Area along Arkansas River
3 About lower one-quarter of model
4 Area along Little Arkansas River
5 Wichita well field

Some of the numbered areas overlap

—1,520— POTENTIOMETRIC CONTOUR—Shows altitude at which water level would have stood in tightly cased wells, based on measured hydraulic heads, 1989. Contour interval 10 feet. Datum is sea level

- -1,520- - POTENTIOMETRIC CONTOUR—Shows altitude at which water level would have stood in tightly cased wells, based on end of transient simulation. Contour interval 10 feet. Datum is sea level

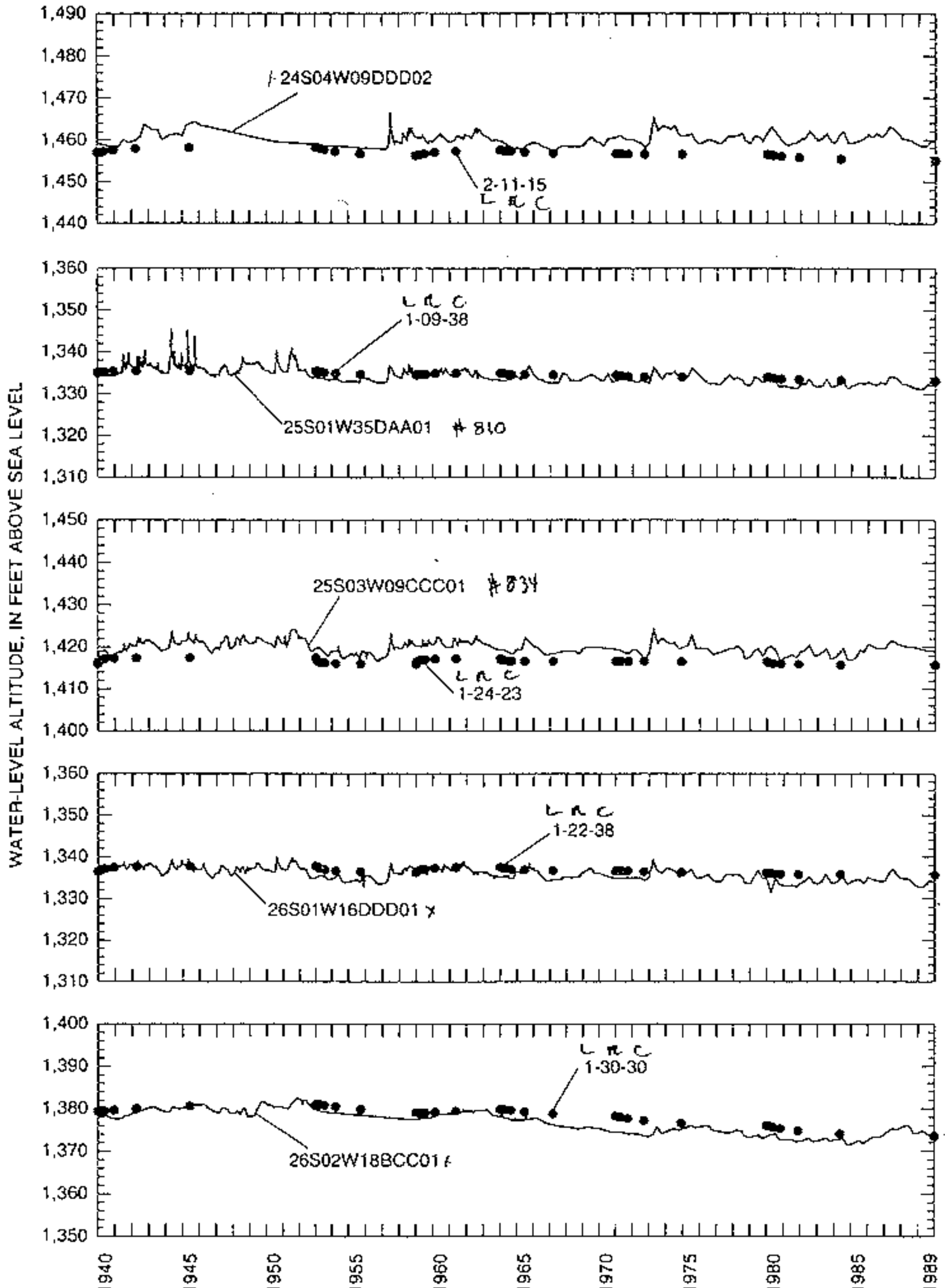
— — — — BOUNDARY OF *EQUUS* BEDS AQUIFER

Figure 27. Measured and simulated potentiometric surface for 1989 in the middle model layer (1989 potentiometric contours based on data available in U.S. Geological Survey WATSTORE data base) and areas used for comparison of measured and simulated hydraulic heads.

increased to about 100 ton/d within the model area (fig. 33). Some of the chloride discharged from the river probably was recaptured by the river throughout its gaining reaches. Because the Arkansas River has changed from a net gaining stream to a net losing stream, the proportion of chloride discharged to the aquifer to chloride recaptured from the aquifer increased during 1940–89.

Transient Simulations, 1990–2019

Transient simulations of hypothetical conditious were used to estimate the possible effect of changing recharge, streamflow, and pumpage on ground-water levels. Simulated water levels for a model cell (layer 1, row 5, column 24) located in the central part of the Wichita well field declined in all 36 simulations

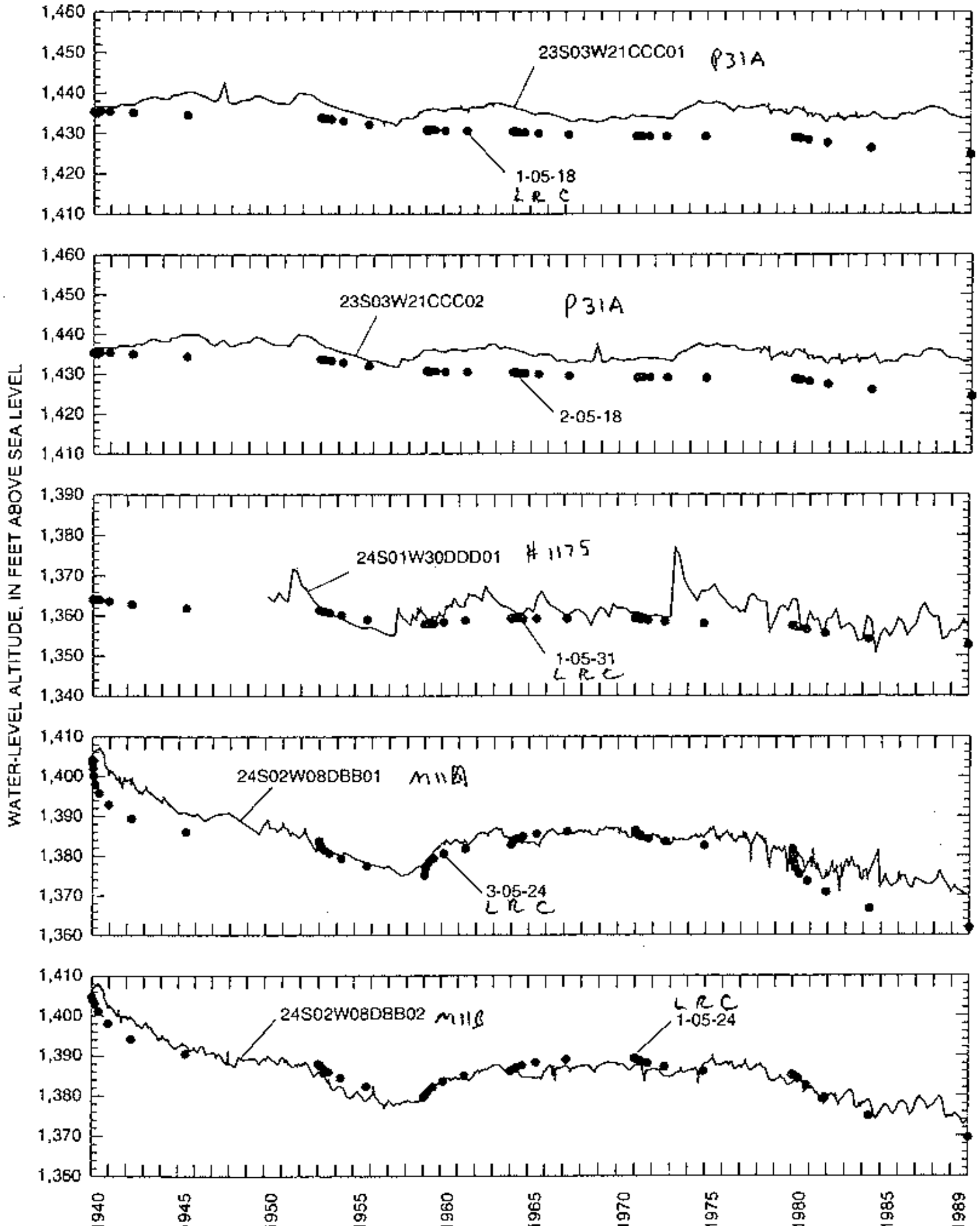


EXPLANATION

• 2-11-15—Simulated water-level altitude and model layer-row-column designation

24S04W09DDD02—Measured water-level altitude and well number

Figure 28. Measured and simulated water-level altitudes for transient simulation, 1940–89 (measured water levels on file with the U.S. Geological Survey, Lawrence, Kans.). Well locations shown in figure 6.



EXPLANATION

• 1-05-18—Simulated water-level altitude and model layer-row-column designation

23S03W21CCC02—Measured water-level altitude and well number

Figure 28. Measured and simulated watar-level altitudes for transient simulation, 1940–89 (measured water levels on file with the U.S. Geological Survey, Lawrence, Kans.). Well locations shown in figure 6—Continued.

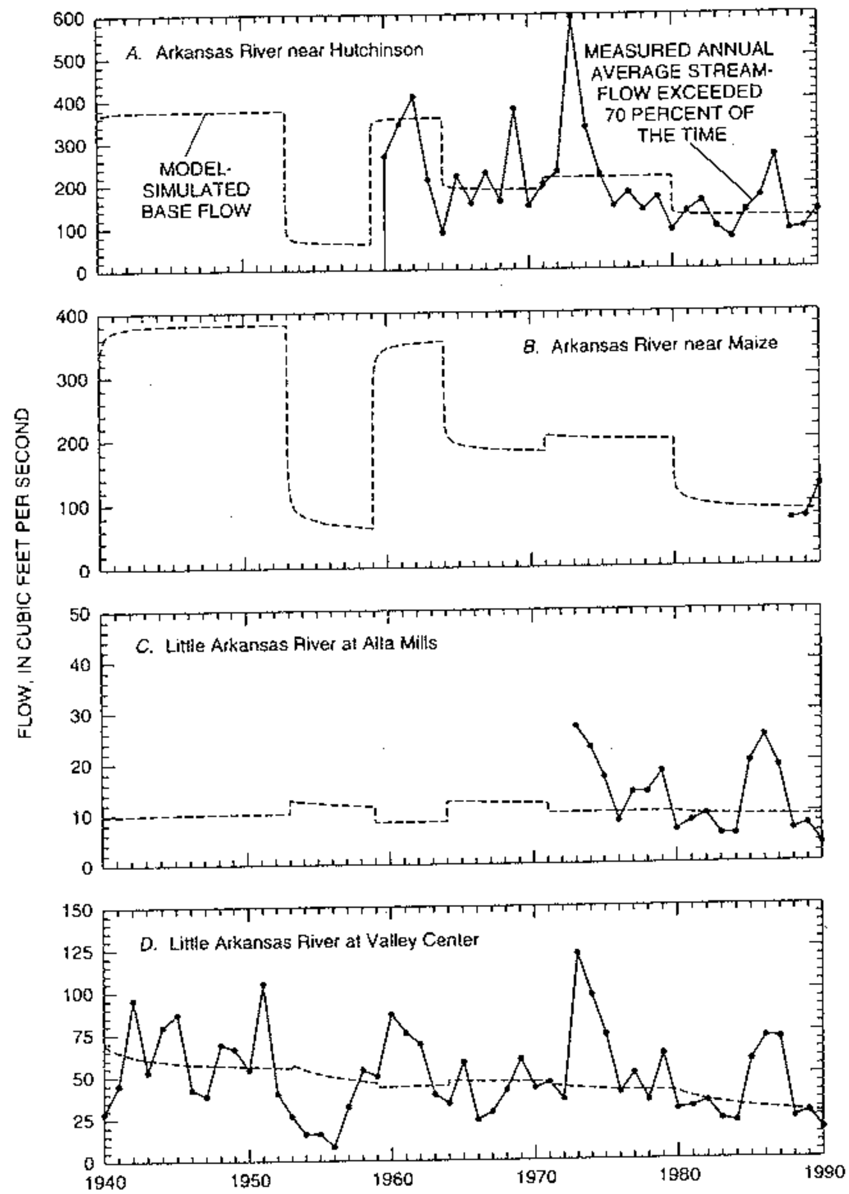


Figure 29. Measured annual average streamflow exceeded 70 percent of time and model-simulated base flow at gaging stations on the (*A*) Arkansas River near Hutchinson, (*B*) Arkansas River near Maize, (*C*) Little Arkansas River at Alta Mills, and (*D*) Little Arkansas River at Valley Center, 1940–90 (measured annual average flow data on file with the U.S. Geological Survey, Lawrence, Kans.).

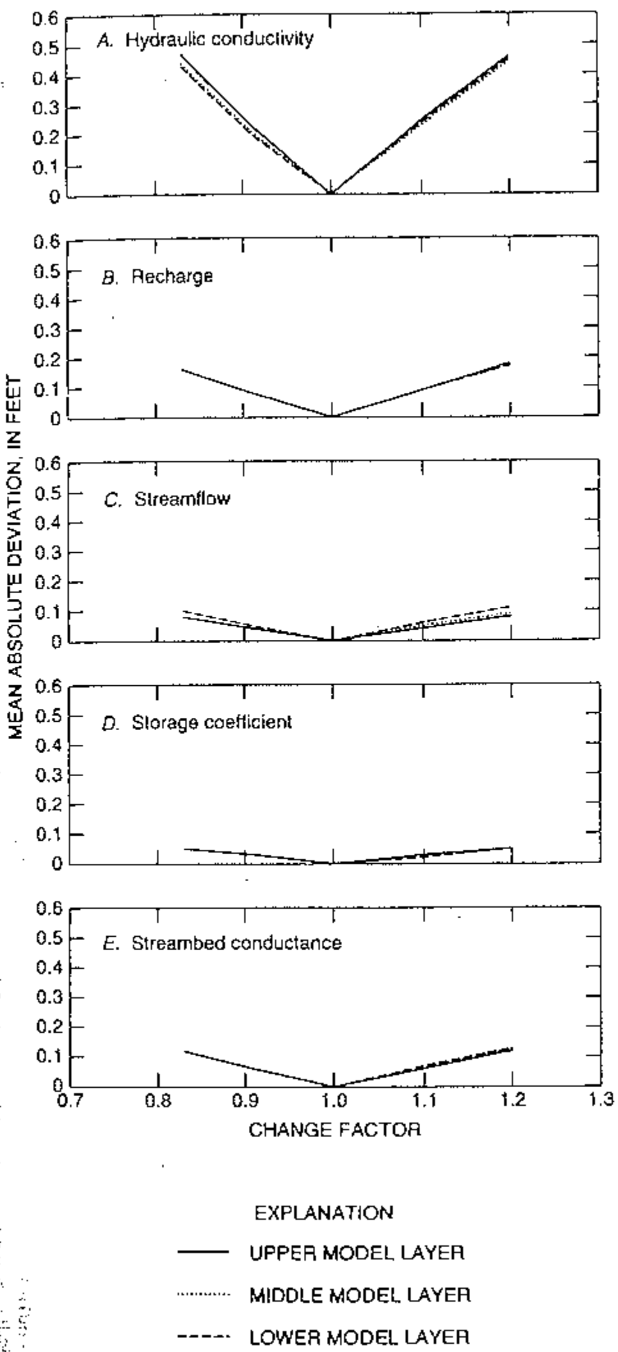


Figure 30. Mean absolute deviation of simulated hydraulic heads from accepted transient-model calibration heads for changes in (*A*) hydraulic conductivity, (*B*) recharge, (*C*) streamflow, (*D*) storage coefficient, and (*E*) streambed conductance.

(fig. 34). Changes in pumpage had the most effect on water levels. Changes in recharge had a moderate effect, and changes in streamflow had the least effect on water levels. Simulations with no increase in pumpage showed water-level declines from 1989 levels that ranged from 0.2 ft (large recharge, large streamflow) to about 10 ft (small recharge, small streamflow) by the end of 2019. These simulations show that, with no increase in pumpage in the Wichita well field area, water levels probably will remain within 10 ft of 1989 water levels, depending on long-term climatic conditions that affect recharge and streamflow. Simulations with a 1-percent per year increase in pumpage showed water-level declines from 1989 levels that ranged from about 27 to 42 ft by the end of 2019. Simulations with a 2-percent per year increase in pumpage showed water-level declines from 1989 levels that ranged from about 54 to 69 ft by the end of 2019. Simulations with a 3-percent per year increase in pumpage showed water-level declines from 1989 levels that ranged from about 75 to 78 ft by the end of 2019.

Simulated water levels for a model cell (layer 1, row 23, column 24) adjacent to the Arkansas River remained relatively stable compared to water levels in the Wichita well field because of the nearby presence of the Arkansas River (fig. 35). Changes in streamflow had the most effect on water levels, whereas pumpage and recharge showed the least effect. Simulations with no increase in pumpage since 1989 showed water-level changes from 1989 levels that ranged from about +1.3 ft (large recharge, large streamflow) to about -0.5 ft (small recharge, small streamflow). Simulations with a 1-percent per year increase in pumpage showed water-level changes from 1989 levels that ranged from about +1.0 to -0.7 ft. Simulations with a 2-percent per year increase in pumpage showed water-level changes from 1989 levels that ranged from about +0.9 to -1.0 ft. Simulations with a 3-percent per year increase in pumpage showed water-level changes from 1989 levels that ranged from about +0.7 to -1.2 ft.

Simulations using average recharge and streamflow were used to estimate the possible effect of changes in pumpage on the volume of water lost from the Arkansas River to the *Equus* beds aquifer and on base flow in the Arkansas River (fig. 36). Within the model area, the estimated cumulative loss from the Arkansas River to the aquifer by the end of 2019, assuming no increase in pumpage since 1989, was

Table 6. Simulated steady-state and transient ground-water budgets

[Values are given in cubic feet per second; E, exponential]

	Budget term	Steady state pre-1940	Transient stress periods 1 1940–52	2 1953–58	3 1959–63	4 1964–70	5 1971–79	6 1980–89
			Upper model layer					
Inflow	Change in storage	0	2.35	16.42	1.26	5.39	2.92	9.07
	Boundary flow	26.57	27.85	32.16	29.48	31.94	32.01	34.68
	Recharge	131.76	142.55	104.11	135.17	118.45	131.81	128.87
	Streamflow loss	13.32	26.15	27.66	34.38	37.23	46.13	63.00
	Leakage from middle layer	39.63	29.66	26.99	25.75	24.96	22.30	17.75
	Total inflow	**211.28**	**228.56**	**207.34**	**226.04**	**217.97**	**235.17**	**253.37**
Outflow	Change in storage	0	1.26	0	6.01	.26	.30	.06
	Boundary flow	32.62	35.55	31.52	33.85	31.63	30.60	29.23
	Well pumpage	2.79	30.88	41.70	47.06	58.92	81.22	94.36
	Evapotranspiration	31.62	28.49	18.79	22.91	19.51	17.48	12.01
	Streamflow gain	102.68	84.46	65.80	66.86	60.30	52.07	34.41
	Leakage to middle layer	41.48	47.94	49.69	49.24	47.41	53.50	83.32
	Total outflow	**211.19**	**228.58**	**207.50**	**225.93**	**218.03**	**235.17**	**253.39**
			Middle model layer					
Inflow	Change in storage	0	.09	.56	.06	.16	.11	.35
	Boundary flow	2.34	2.41	2.72	2.56	2.72	2.73	2.94
	Leakage from upper layer	41.48	47.94	49.69	49.24	47.41	53.50	83.32
	Leakage from lower layer	21.30	14.45	13.06	12.90	12.52	11.18	9.87
	Total inflow	**65.12**	**64.89**	**66.03**	**64.76**	**62.81**	**67.52**	**96.48**
Outflow	Change in storage	0	.03	0	.18	8.94E-3	5.17E-3	6.39E-4
	Boundary flow	1.60	1.66	1.49	1.49	1.38	1.32	1.28
	Well pumpage	.06	6.85	9.68	10.20	10.76	14.56	24.12
	Leakage to upper layer	39.63	29.66	26.99	25.75	24.96	22.30	17.75
	Leakage from lower layer	23.80	26.70	27.89	27.08	25.71	29.35	53.34
	Total outflow	**65.09**	**64.90**	**66.05**	**64.70**	**62.83**	**67.53**	**96.49**

Table 6. Simulated steady-state and transient ground-water budgets—Continued

	Budget term	Steady state pre-1940	Transient stress periods 1 1940–52	2 1953–58	3 1959–63	4 1964–70	5 1971–79	6 1980–89
			Lower model layer					
Inflow	Change in storage	0	0.11	0.59	0.07	0.15	0.11	0.40
	Boundary flow	.47	.43	.49	.50	.53	.52	.54
	Leakage from middle layer	23.80	26.70	27.89	27.08	25.71	29.35	53.34
	Total inflow	**24.27**	**27.24**	**28.97**	**27.65**	**26.39**	**29.98**	**54.28**
Outflow	Change in storage	0	.02	0	.19	.01	2.55E-3	9.03E-5
	Boundary flow	2.95	3.07	2.74	2.75	2.55	2.47	2.31
	Well pumpage	0	9.71	13.18	11.77	11.31	16.34	42.11
	Leakage to middle layer	21.30	14.45	13.06	12.90	12.52	11.18	9.87
	Total outflow	**24.25**	**27.25**	**28.98**	**27.61**	**26.39**	**29.99**	**54.29**
			All model layers					
Inflow	Change in storage	0	2.55	17.57	1.39	5.70	3.14	9.82
	Boundary flow	29.38	30.69	35.37	32.54	35.19	35.26	38.16
	Recharge	131.76	142.55	104.11	135.17	118.45	131.81	128.87
	Streamflow loss	13.32	26.15	27.66	34.38	37.23	46.13	63.00
	Total inflow	**174.46**	**201.94**	**184.71**	**203.48**	**196.57**	**216.34**	**239.85**
Outflow	Change in storage	0	1.31	0	6.38	.28	.31	.06
	Boundary flow	37.17	40.28	35.75	38.09	35.56	34.39	32.82
	Well pumpage	2.85	47.44	64.56	69.03	80.99	112.12	160.59
	Evapotranspiration	31.62	28.49	18.79	22.91	19.51	17.48	12.01
	Streamflow gain	102.68	84.46	65.80	66.86	60.30	52.07	34.41
	Total outflow	**174.32**	**201.98**	**184.90**	**203.27**	**196.64**	**216.37**	**239.89**

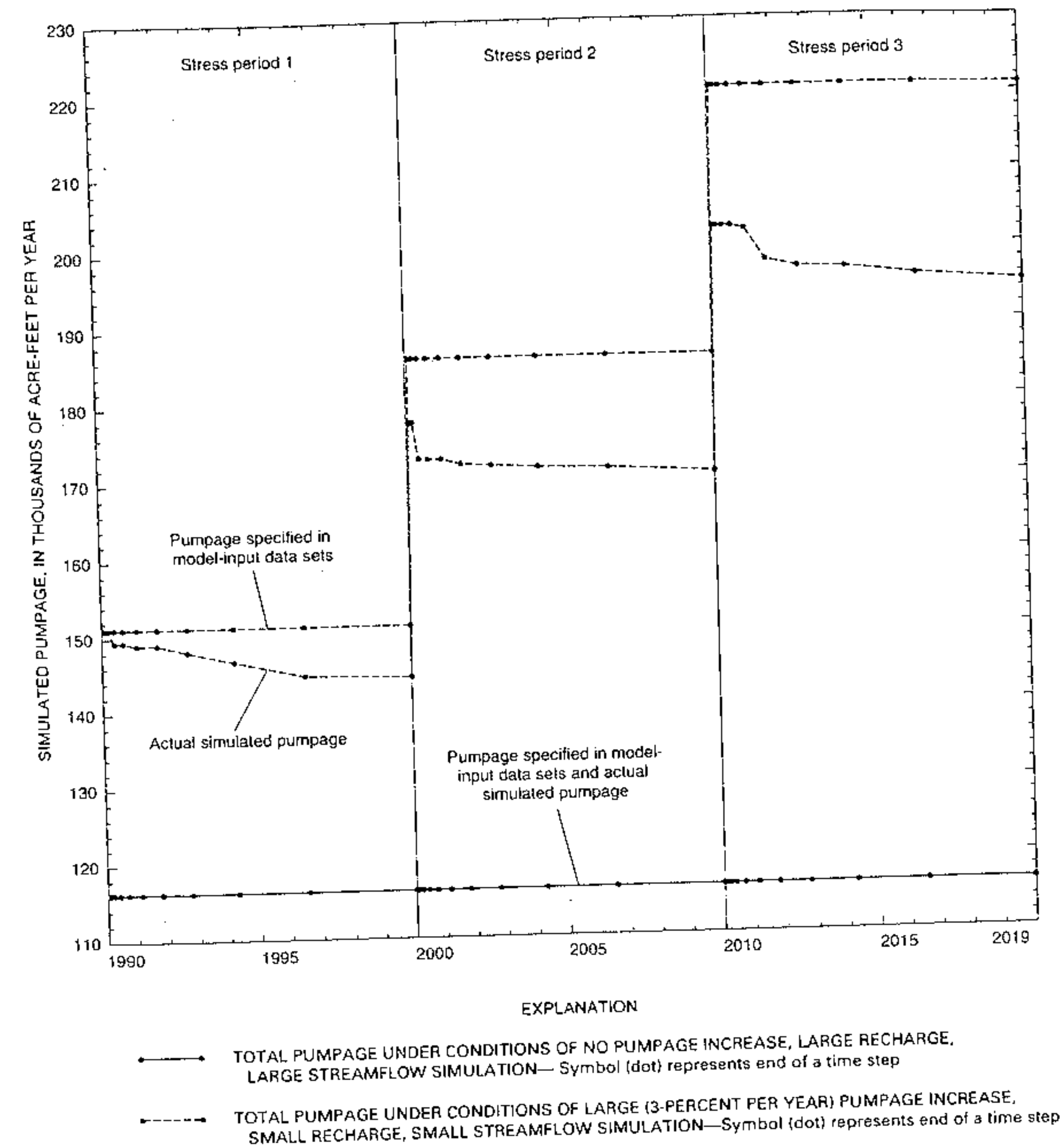


Figure 31. Pumpage specified in model-input data sets and actual simulated pumpage for two simulations of hypothetical conditions: (1) no pumpage increase, large recharge, large streamflow, and (2) large pumpage increase, small recharge, small streamflow. These two simulations represent the extremes (smallest and largest, respectively) of stress on the aquifer.

about 65 ft^3/s, and the cumulative gain was about 6 ft^3/s, giving a net base-flow loss from the river of 59 ft^3/s (fig. 36*A*). Assuming a 1-percent per year increase in pumpage, the estimated cumulative loss from the river by the end of 2019 was about 84 ft^3/s, and the cumulative gain was about 4 ft^3/s, giving a net base-flow loss from the river of 80 ft^3/s (fig. 36*B*). Assuming a 2-percent per year increase in pumpage, the estimated cumulative loss from the river by the end of 2019 was about 99 ft^3/s, and the cumulative gain

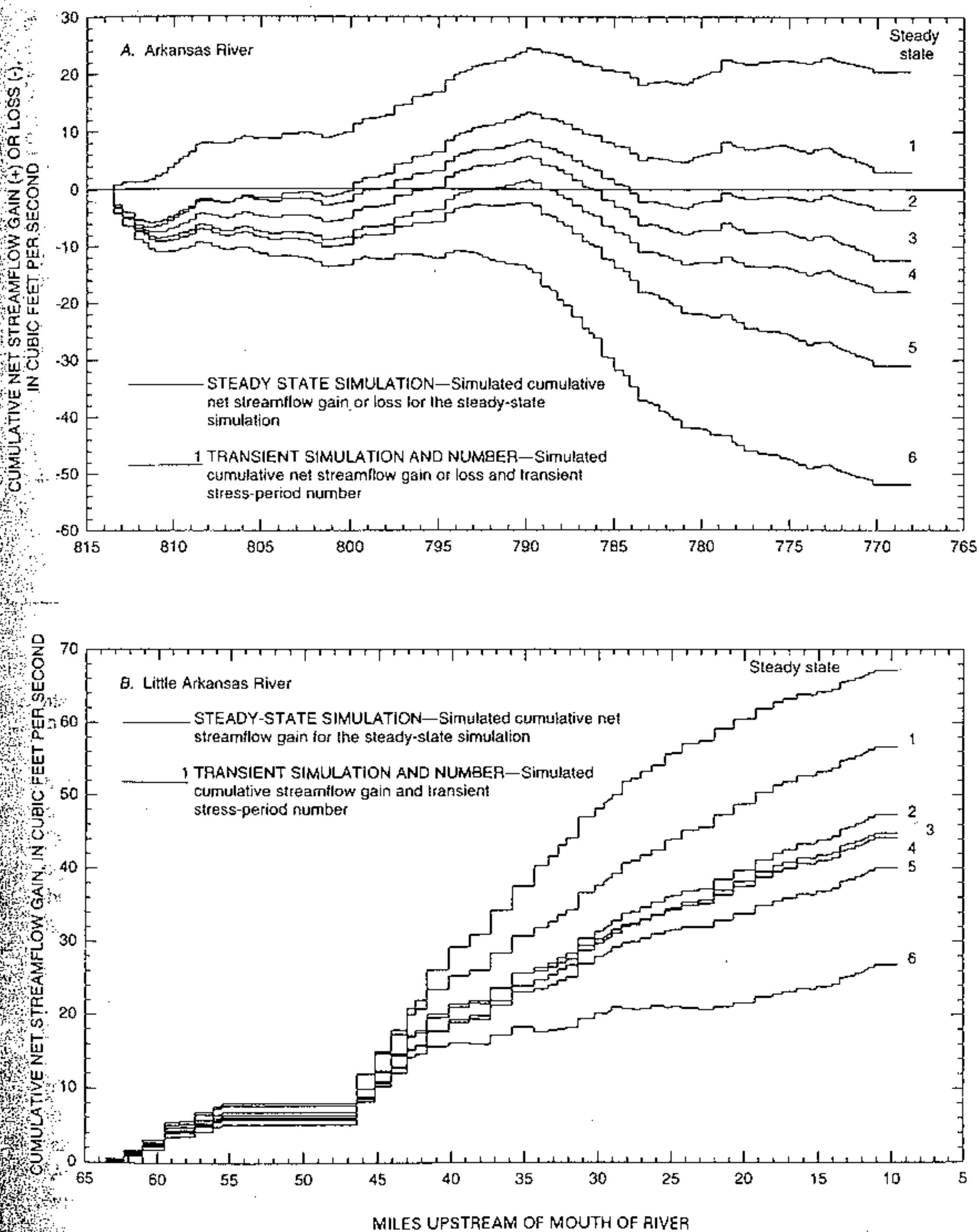


Figure 32. Simulated cumulative net streamflow gain from or loss to the *Equus* beds aquifer at end of each stress period for the (*A*) Arkansas River and (*B*) Little Arkansas River.

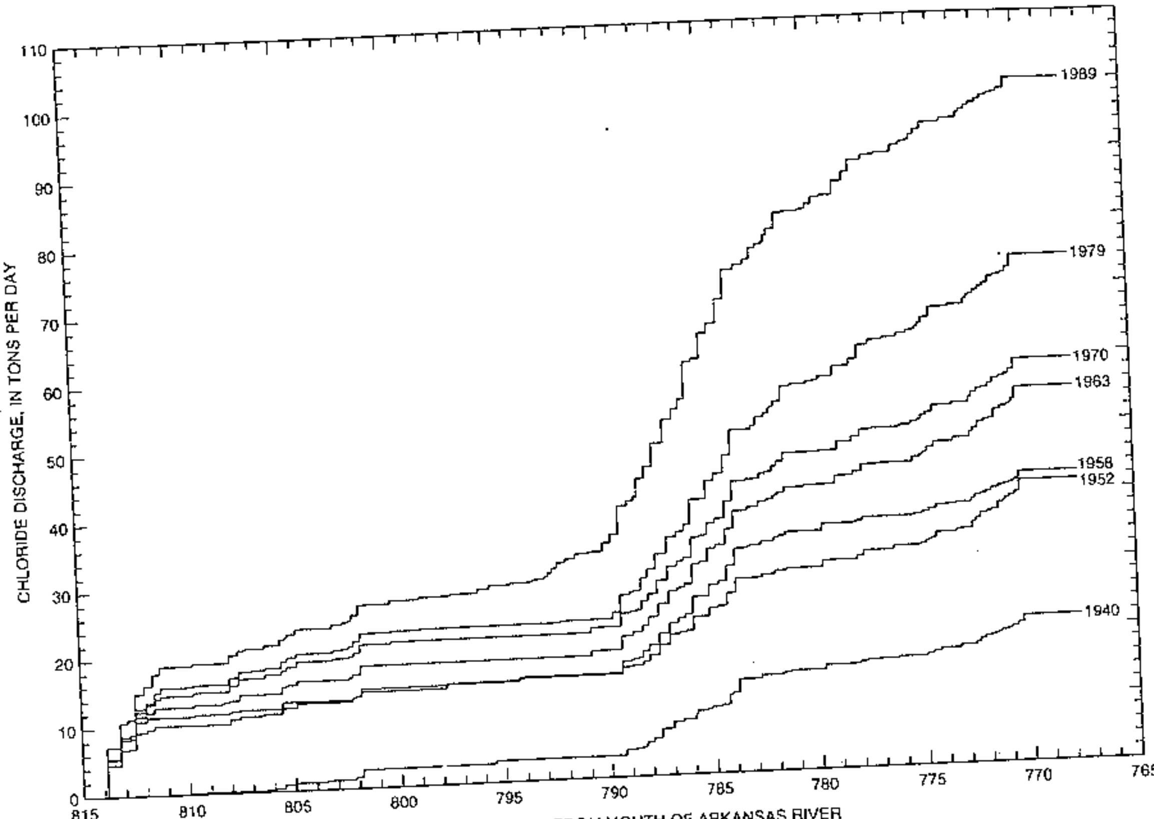


Figure 33. Estimated cumulative chloride discharge from Arkansas River to *Equus* beds aquifer at end of each stress period based on simulated streamflow loss from the Arkansas River and assuming a chloride concentration of 630 milligrams per liter.

was about 2 ft^3/s, giving a net base-flow loss from the river of 97 ft^3/s (fig. 36*C*). Assuming a 3-percent increase in pumpage, the estimated cumulative loss from the river by the end of 2019 was about 118 ft^3/s, and the cumulative gain was about 1 ft^3/s, giving a net base-flow loss from the river of 117 ft^3/s (fig. 36*D*).

During the simulated 1990–2019 period, estimated chloride discharge from the Arkansas River to the aquifer increased over 1989 estimated quantities (fig. 37) in proportion to increases in loss of river water. Assuming no increase in pumpage since 1989 and a 630-mg/L chloride concentration in Arkansas River water, the estimated cumulative chloride-load discharge within the model area was about 110 ton/d by the end of 2019 (fig. 37*A*). Assuming a 1-percent per year increase in pumpage since 1989, the estimated cumulative chloride-load discharge within the model area was about 142 ton/d by the end of 2019 (fig. 37*B*). Assuming a 2-percent per year increase in pumpage since 1989, the estimated cumulative chloride-load discharge within the model area was about 169 ton/d by the end of 2019 (fig. 37*C*). Assuming a 3-percent per year increase in pumpage since 1989, the estimated cumulative chloride-load discharge within the model area was about 200 ton/d by the end of 2019 (fig. 37*D*).

Particle Tracking

MODPATH is a program developed by Pollock (1989) that computes and displays the location through time of water particles using results from the MODFLOW flow-model program. MODPATH was used to display the zones of interaction between the Arkansas River and the *Equus* beds aquifer and to simulate the paths that particles of water from the Arkansas River follow as they move through the aquifer. Assuming that dissolved constituents, such as

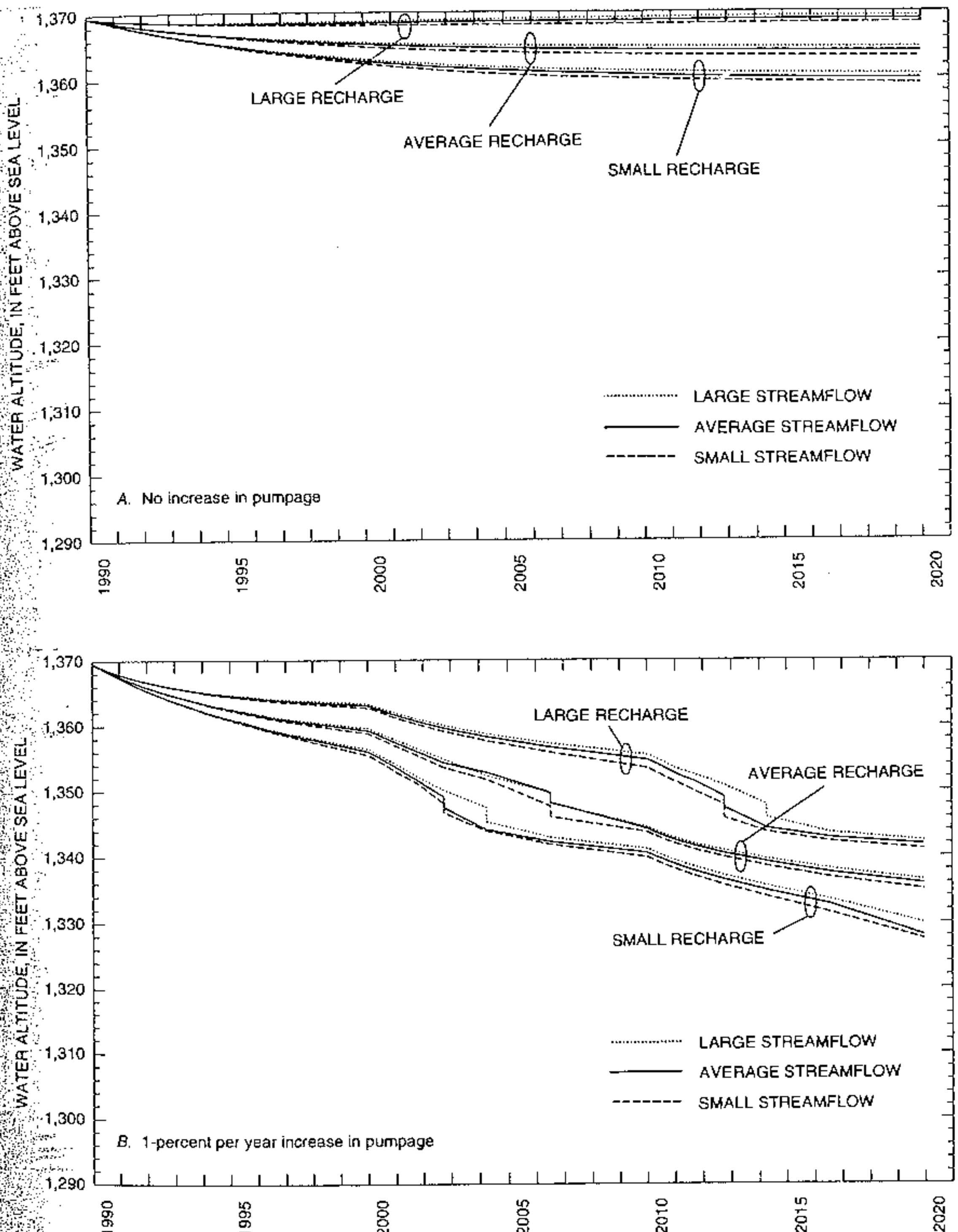
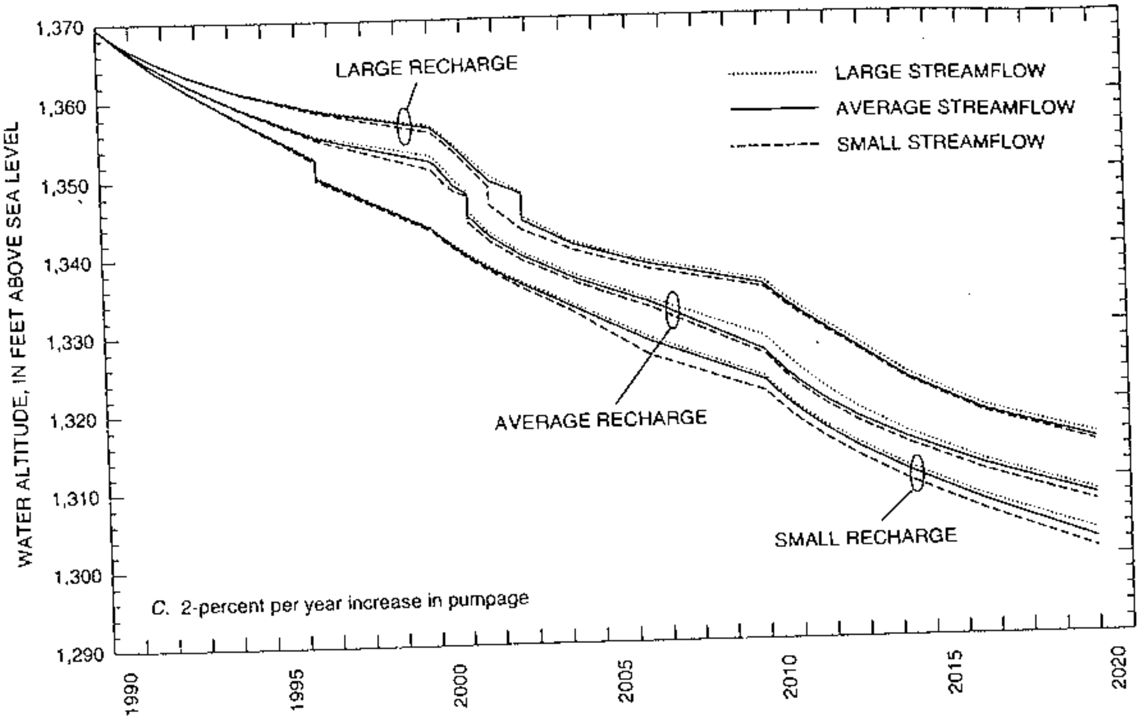


Figure 34. Estimated water levels at model row-5, column-24 in Wichita well field for simulations assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989. The abrupt declines of water-level altitude in (*B*), (*C*), and (*D*) result from switching from upper layer to middle layer water-level altitudes when the water-level altitude declined below the bottom of the upper layer model cell.



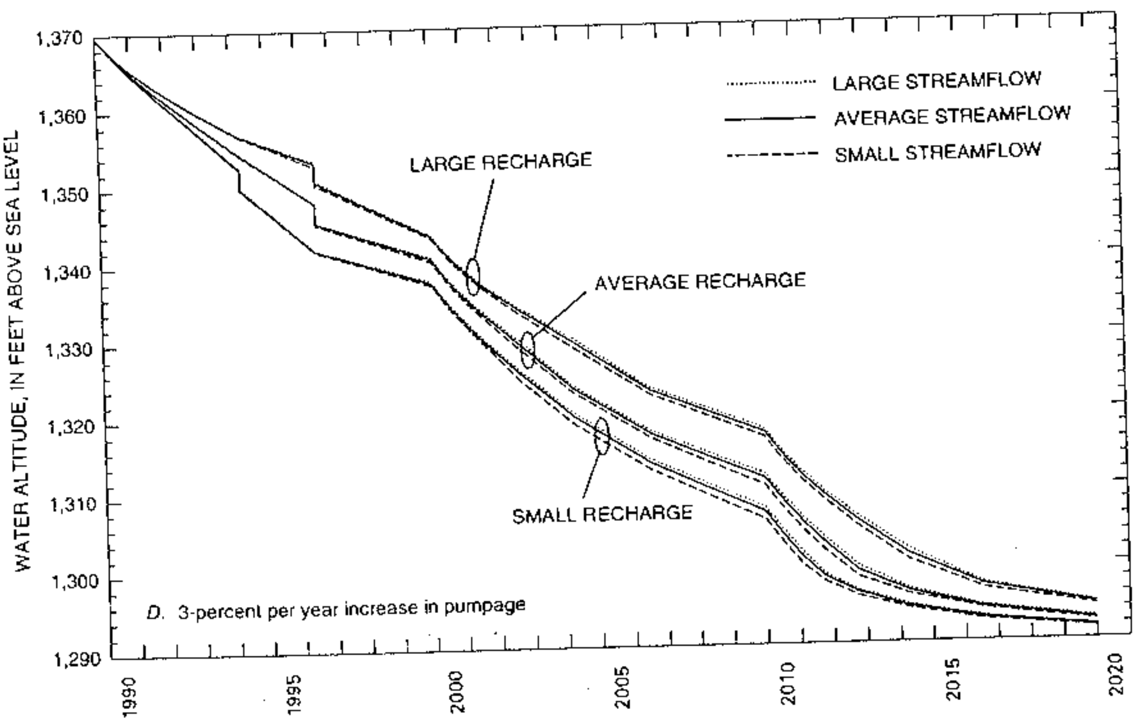
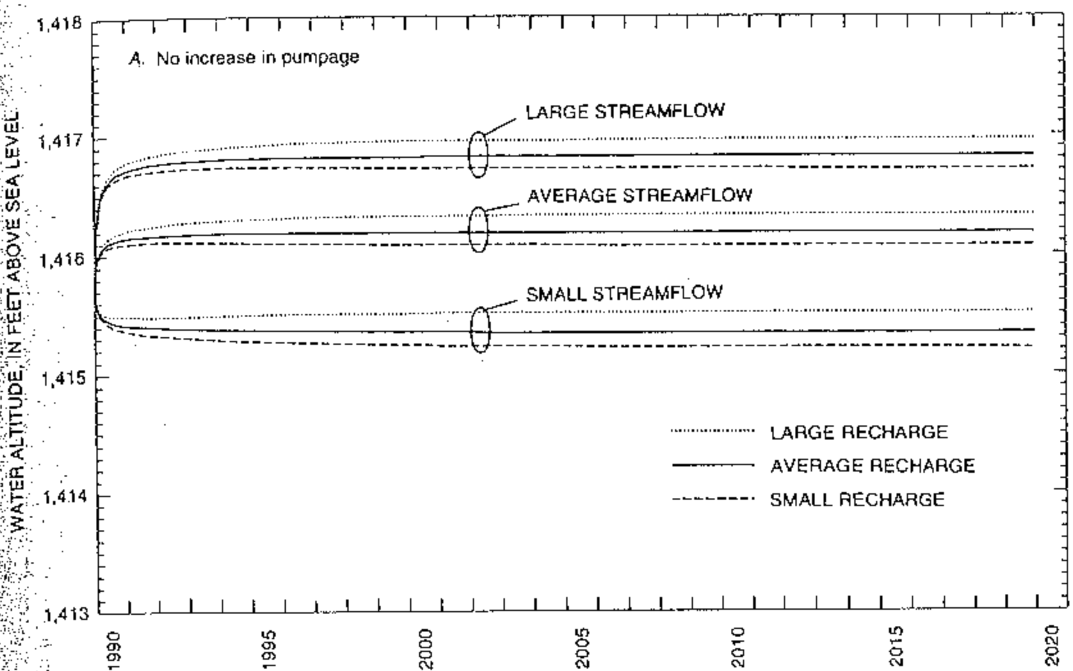


Figure 34. Estimated water levels at model row-5, column-24 in Wichita well field for simulations assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989. The abrupt declines of water-level altitude in (*B*), (*C*), and (*D*) result from switching from upper layer to middle layer water-level altitudes when the water-level altitude declined below the bottom of the upper layer model cell—Continued.



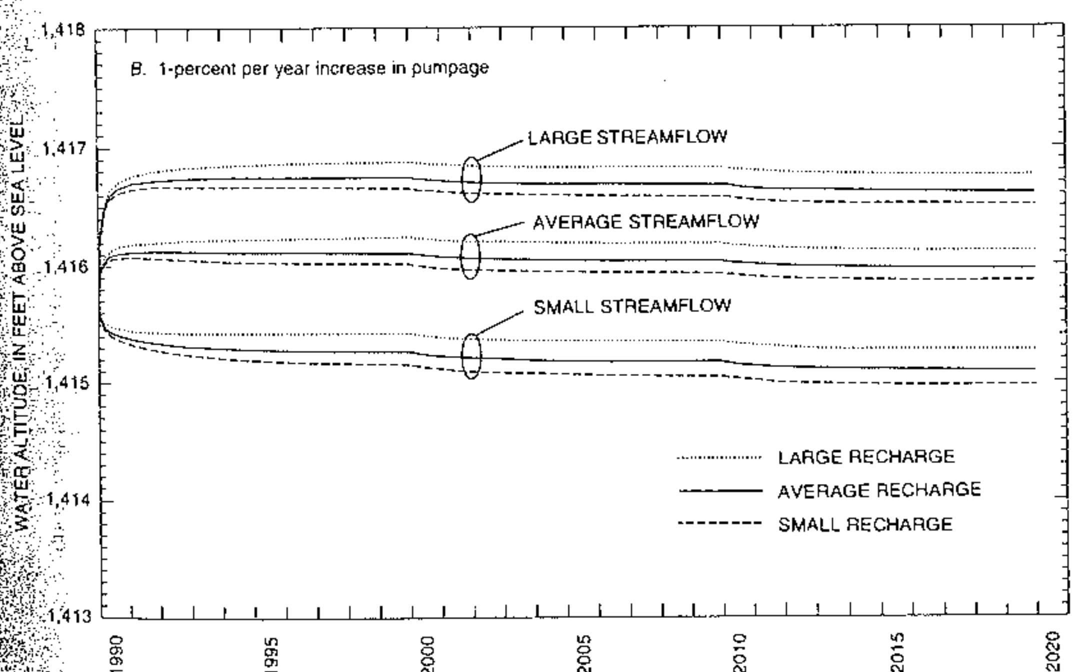


Figure 35. Estimated water levels at model row-24, column-23 near Arkansas River for simulations assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989.

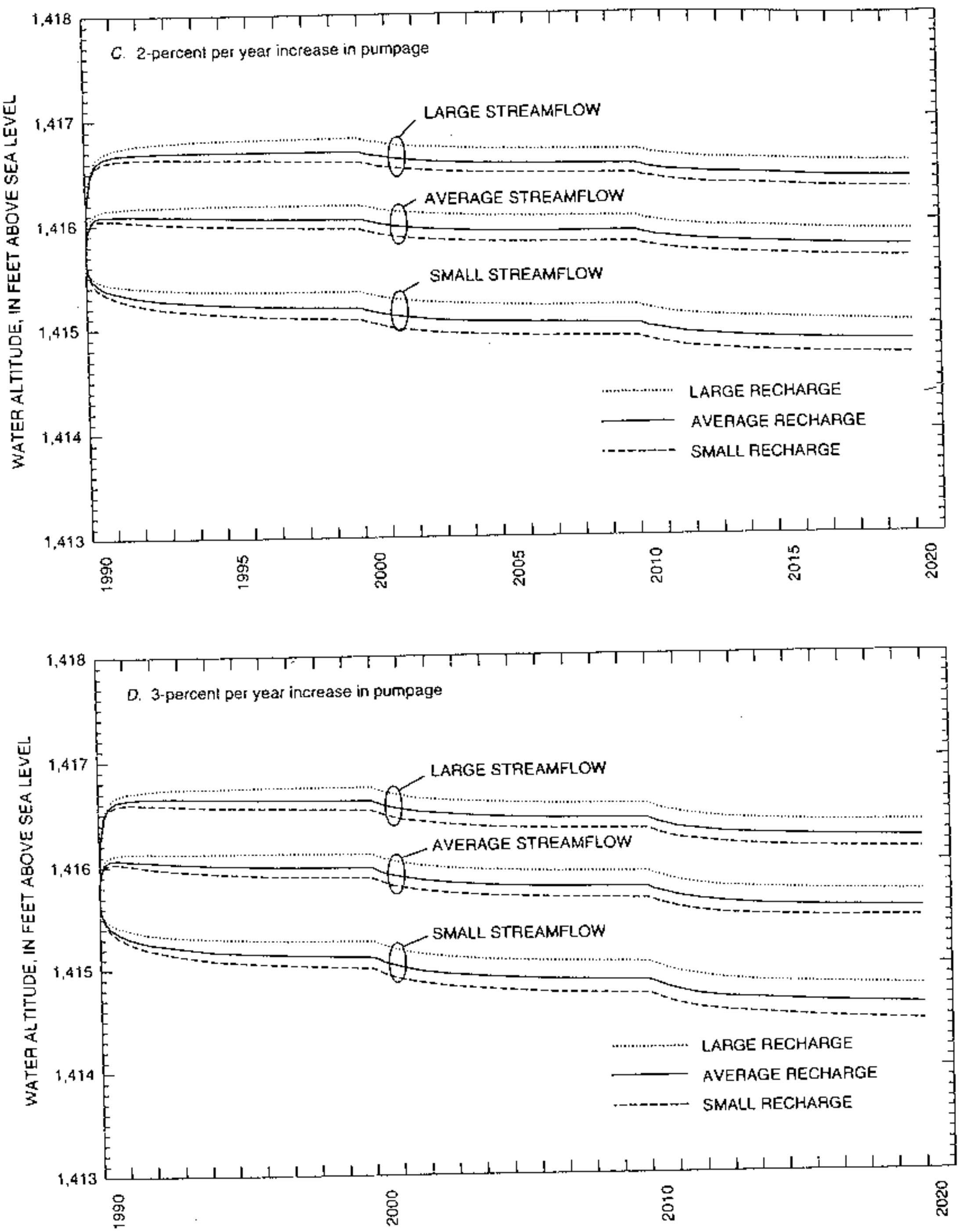


Figure 35. Estimated water levels at model row-24, column-23 near Arkansas River for simulations assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

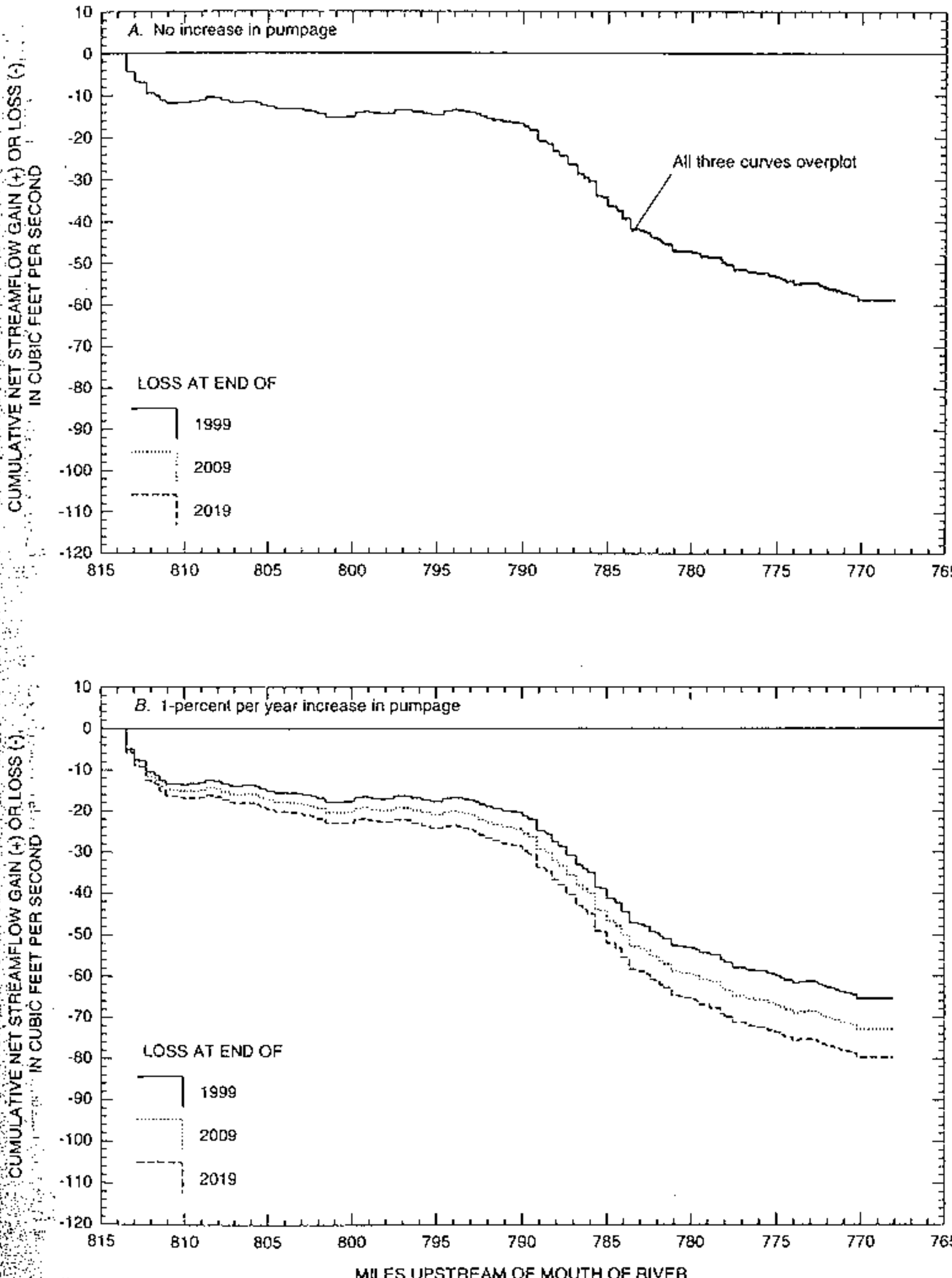
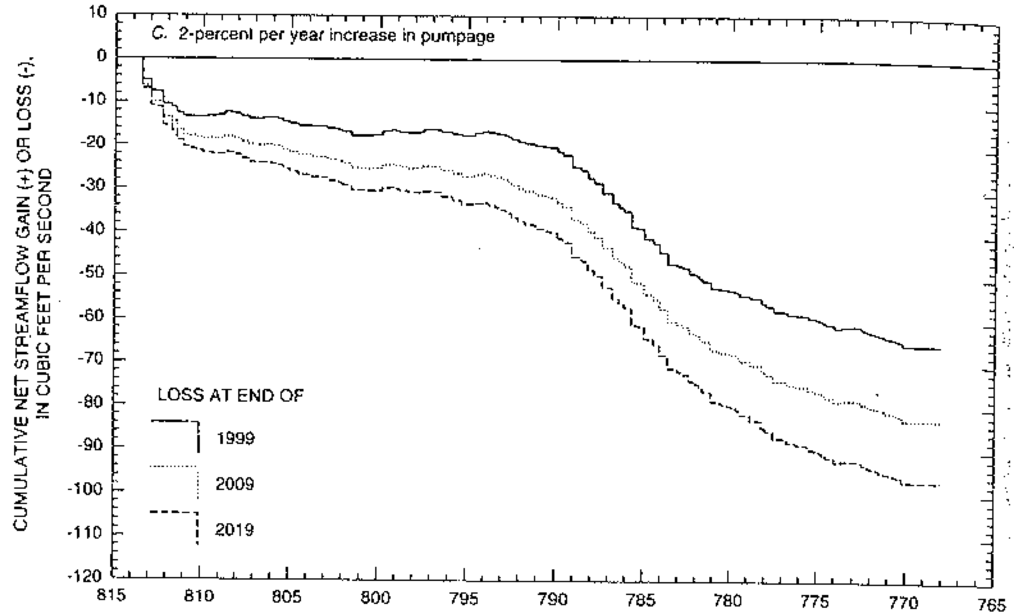


Figure 36. Estimated cumulative net streamflow loss of water from Arkansas River to *Equus* beds aquifer at end of each stress period assuming (*A*) no incraase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989.



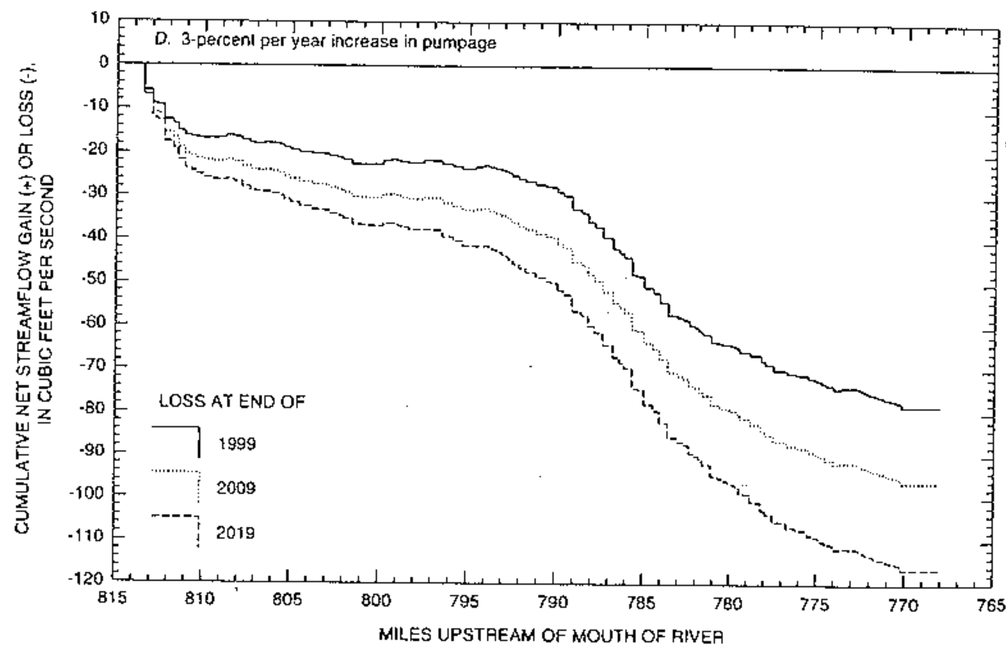


Figure 36. Estimated cumulative net streamflow loss of water from Arkansas River to *Equus* beds aquifer at end of each stress period assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

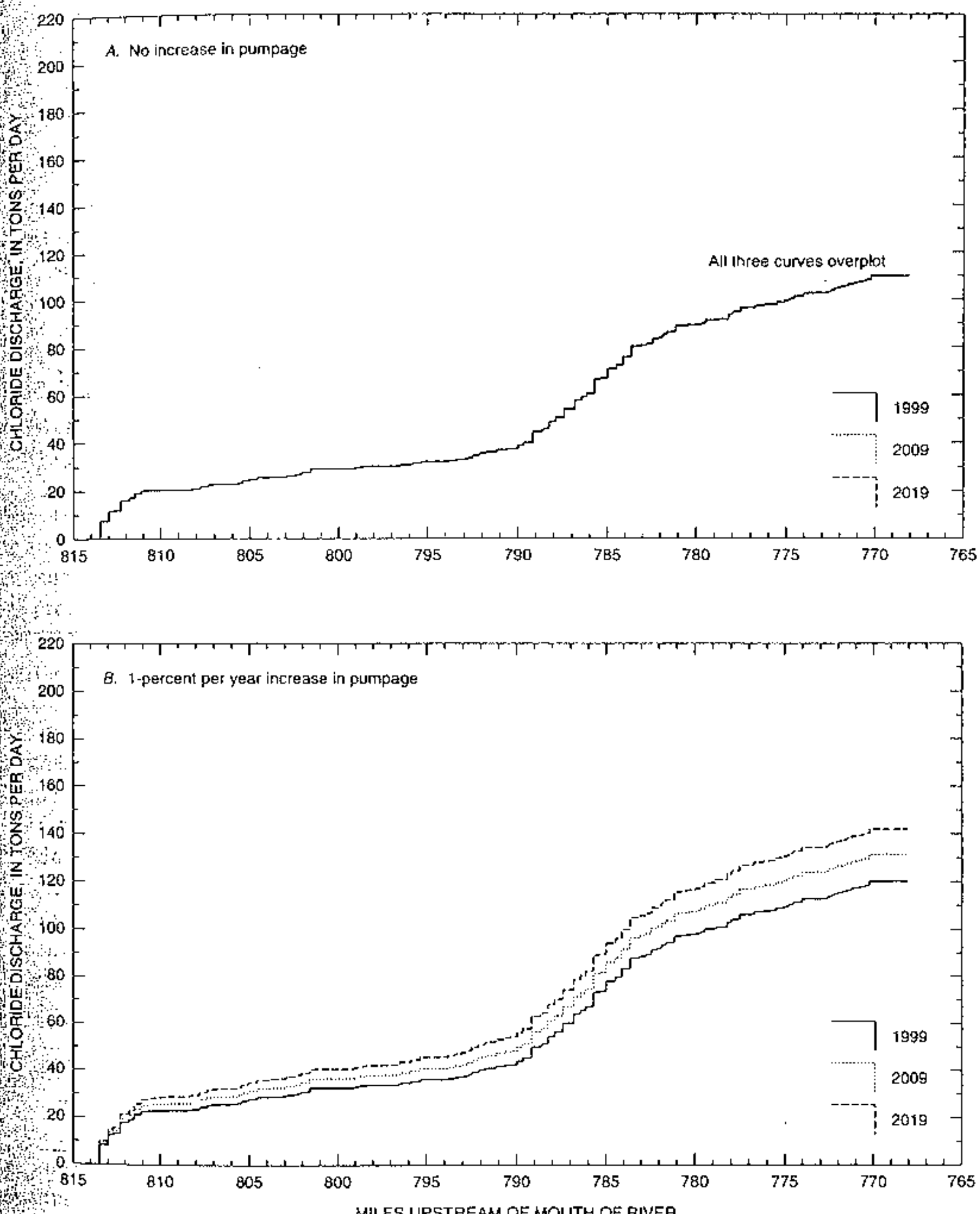
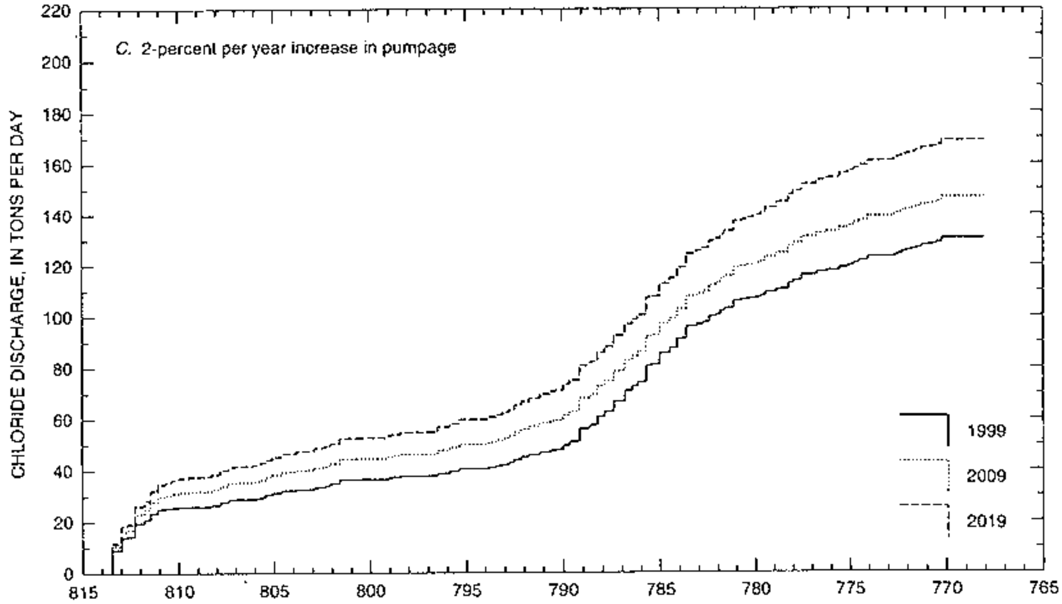


Figure 37. Estimated cumulative chloride discharge from Arkansas River to *Equus* beds aquifer at end of each stress period assuming a chloride concentration of 630 milligrams per liter and (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989.



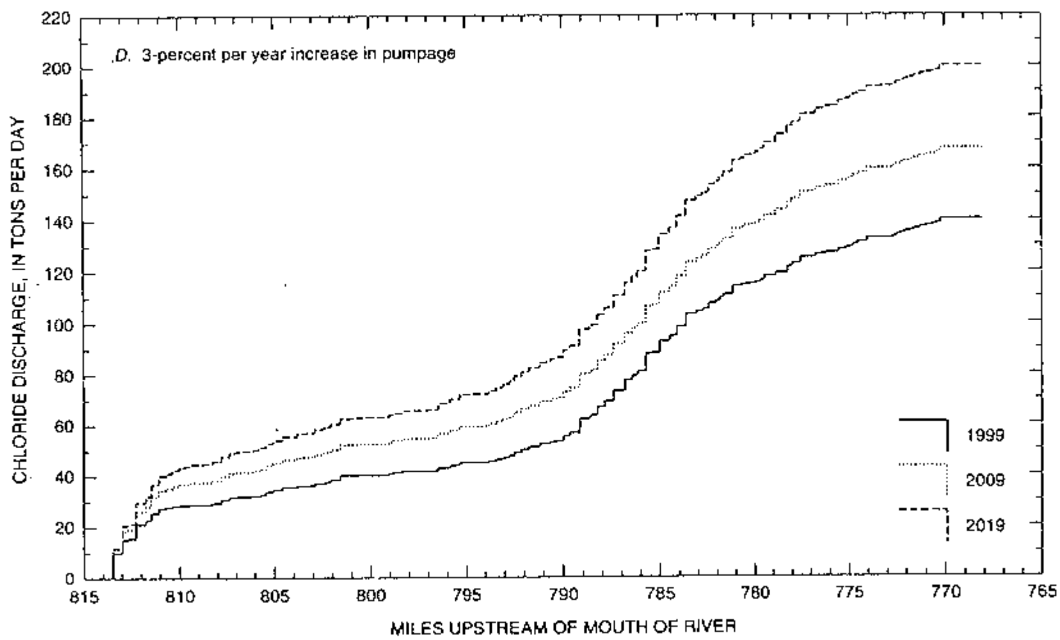


Figure 37. Estimated cumulative chloride discharge from Arkansas River to *Equus* beds aquifer at end of each stress assuming a chloride concentration of 630 milligrams per liter and (*A*) no increase in pumpage since 1989, (*B*) a 1-perce year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent pe increase in pumpage since 1989—Continued.

chloride, move with ground water, the particle tracker also was used to simulate the location through time of chloride particles in the aquifer.

Although MODPATH simulates the advective transport of constituents such as chloride, it does not simulate the concentration of chloride in the aquifer. MODPATH does not account for dispersion, mixing, or retardation that may occur during the transport of the chemical constituent and does not correctly simulate the flow paths of particles in a solution that is much denser than the surrounding solution, such as concentrated oil-field brine in freshwater.

In addition to the aquifer properties, stresses, and boundary conditions required by MODFLOW, one additional property, aqnifer porosity, was required for the particle-tracking program. A porosity of 0.20 was uniformly assigned for all three layers of the model and was chosen as an average of the 0.15 specific yield nsed in the flow model and the 0.25 porosity used by Spinazola and others (1985) in their solute-transport model.

The assumptions and limitations of the MODPATH program are:

(1) MODFLOW output represents steady-state conditions (the version of MODPATH used did not simulate transient conditions).
(2) Particles moving horizontally from oue model cell to another cell in the same model layer move to the same location coordinates within the new cell as in the previons cell.
(3) For cells in which the volume of discharge to a well or river or other hydraulic sink is less than the volume of ground-water flow into the cell (weak sinks), it cannot be determined whether any particular particle discharges to the hydraulic sink or flows through the cell.
(4) Particle paths through weak-sink cells may not be accurate if discharge from the cell cannot be represented as being uniformly distributed across a cell.

In addition to these assumptions and limitations, all of the assumptions and limitations inherent in the MODFLOW program applied. Path-line accuracies due to weak-sink cells and large cell sizes (discretization error) may be improved by finer discretization in space and time.

Particle-tracking simulations were done using ground-water flow-model results from the steady-state simulation, from the end of each stress period in the transient simulation, and from the end of each stress period in the transient hypothetical simulations. Ending conditions from each transient stress period were used in the steady-state particle-tracking simulations to represent each transient stress period. This approach is generally valid if it is assumed that most of the change in storage occurs early in the transient stress periods and that the later part of each transient stress period approaches steady-state conditions. These assumptions are generally satisfied for stress periods 1–6 (1940–89, fig. 38) and for stress periods for all variations of the hypothetical simulations except for the large pumpage, small recharge, small streamflow simulation (fig. 38). The error in simulated particle location will be larger for those stress periods where most of the storage change does not occur early in the stress period. This error cannot be quantified without a transient particle-tracking simulation. Generally, however, the error probably would be to overestimate the distance of particle movement.

To track the flow of water from the Arkansas River into the aquifer, one particle was placed near the upper surface of each Arkansas River model cell. This distribution of particles in river model cells will be called the "source distribution." The particle tracker then was used to simulate particle flow paths through the aquifer, for specified periods of time, under the steady-state conditions used to represent ground-water flow-model stress periods.For the steady-state ground-water flow-model simulation, particles representing chloride from the river were tracked to represent the steady-state distribution of particles in the aquifer in the late 1930s. This steady-state distribution of particles and the source distribution of particles were combined and used as the starting particle distribution for particle-tracking simulations during the first transient stress period. The resulting particle distribution from each stress period and the source distribution of particles were combined and used as the starting particle distributiou for each following stress period. For the 1940–89 and 1990–2019 stress periods, the particle-tracking simulations were made for a period of time corresponding to the length of each transient stress period. Thus, the MODPATH program was used to simulate the distribution of Arkansas River chloride in the aquifer at the end of each stress period.

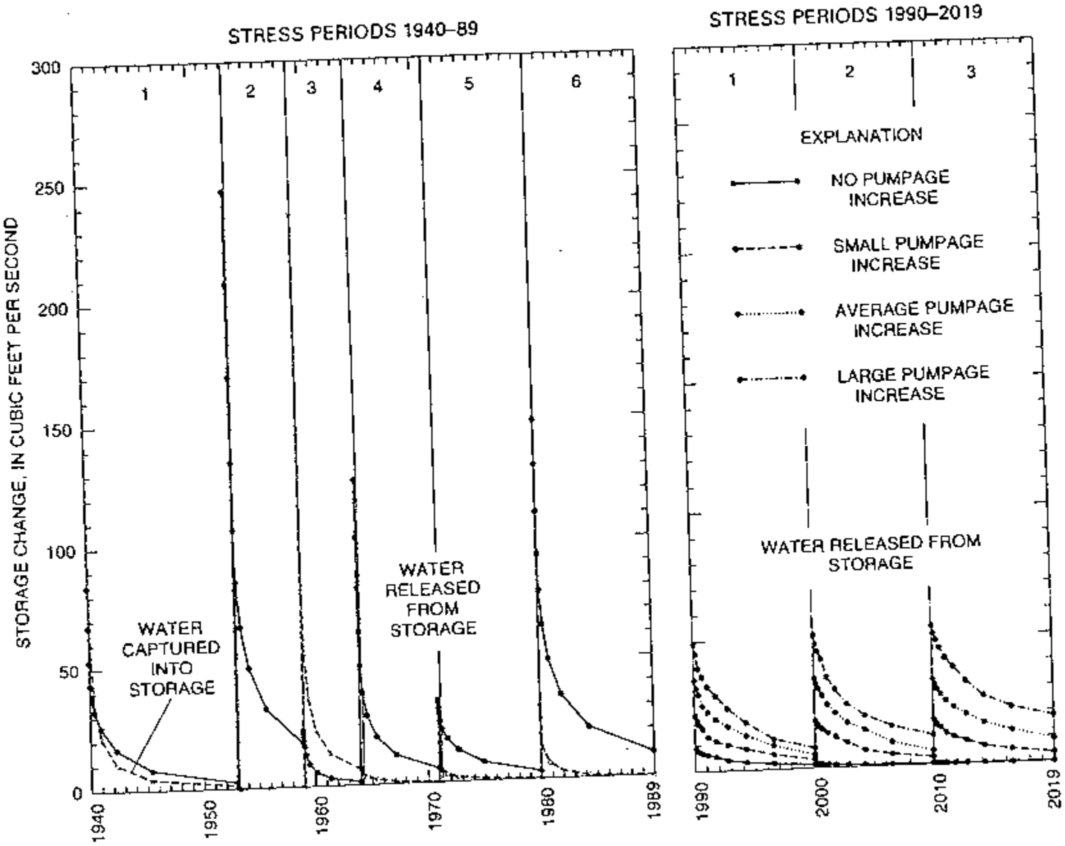


Figure 38. Simulated rate of storage change for stress periods 1–6 (1940–89) and hypothetical stress periods 1–3 (1990–2019). Hypothetical stress-period rates of storage change are for no, small, average, and large pumpage increases and small recharge and small streamflow simulations.

Discussion of Particle-Tracking Results, 1940–89

The MODPATH program was used to simulate the distribution in the aquifer of particles representing chloride from the Arkansas River (fig. 39). The lines in figure 39 show the maximum extent of chloride from the river at the end of each stress period; the lines do not indicate chloride concentration. In the upper model layer, the steady-state (pre-1940) distribution of chloride particles was limited to a small area near Hutchinson, a narrow band along part of the upstream reach of the river, and a wider band along the downstream reach of the river (fig. 39*A*). In the middle and lower model layers, particles are distributed in narrow bands near part of the downstream reach of the river (figs. 39*B* and 39*C*). The distribution of particles in the upper layer indicates areas where the Arkansas River was naturally losing water to the aquifer prior to 1940. For 1940–89, the particle-distribution areas in all three layers are larger with each successive stress period (fig. 39), and new distributions of particles emerge as longer reaches of the river begin to lose water to the aquifer. The particle-tracking simulations indicate that most of the chloride from the river stayed in the upper model layer, but some moved into the lower two layers. The shape of the 1989 distribution of particles in the upper layer generally is similar to the shape of the 100-mg/L line of equal chloride concentration in the upper unit (fig. 19*A*). Particle distributions indicate that chloride in the upper and middle aquifer units may

A. Upper model layer, 1940–89

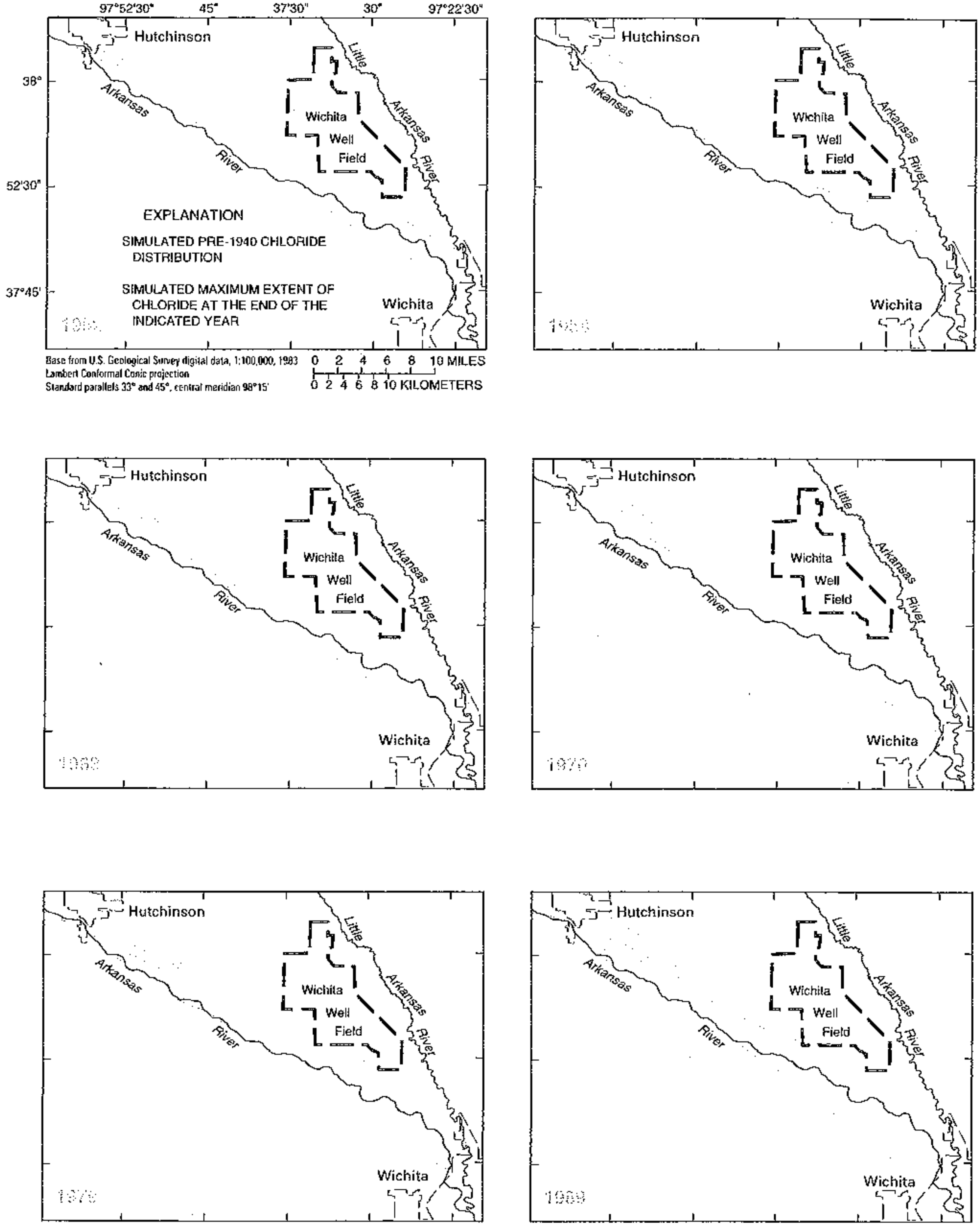


Figure 39. Simulated distribution of chloride from Arkansas River at end of each stress period in (*A*) upper, (*B*) middle, and (*C*) lower model layers.

B. Middle model layer, 1940–89

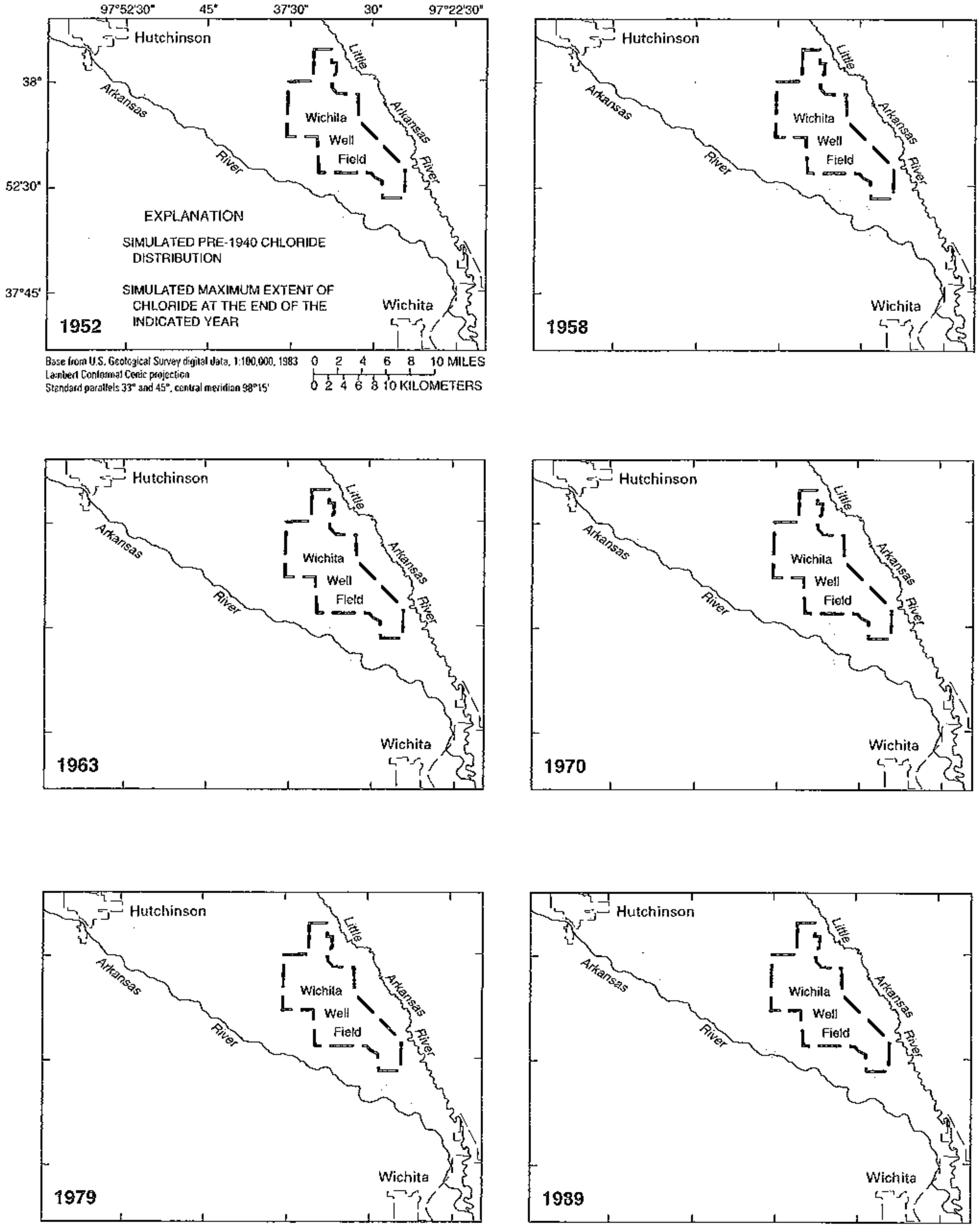


Figure 39. Simulated distribution of chloride from Arkansas River at end of each stress period in (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

C. Lower model layer, 1940–89

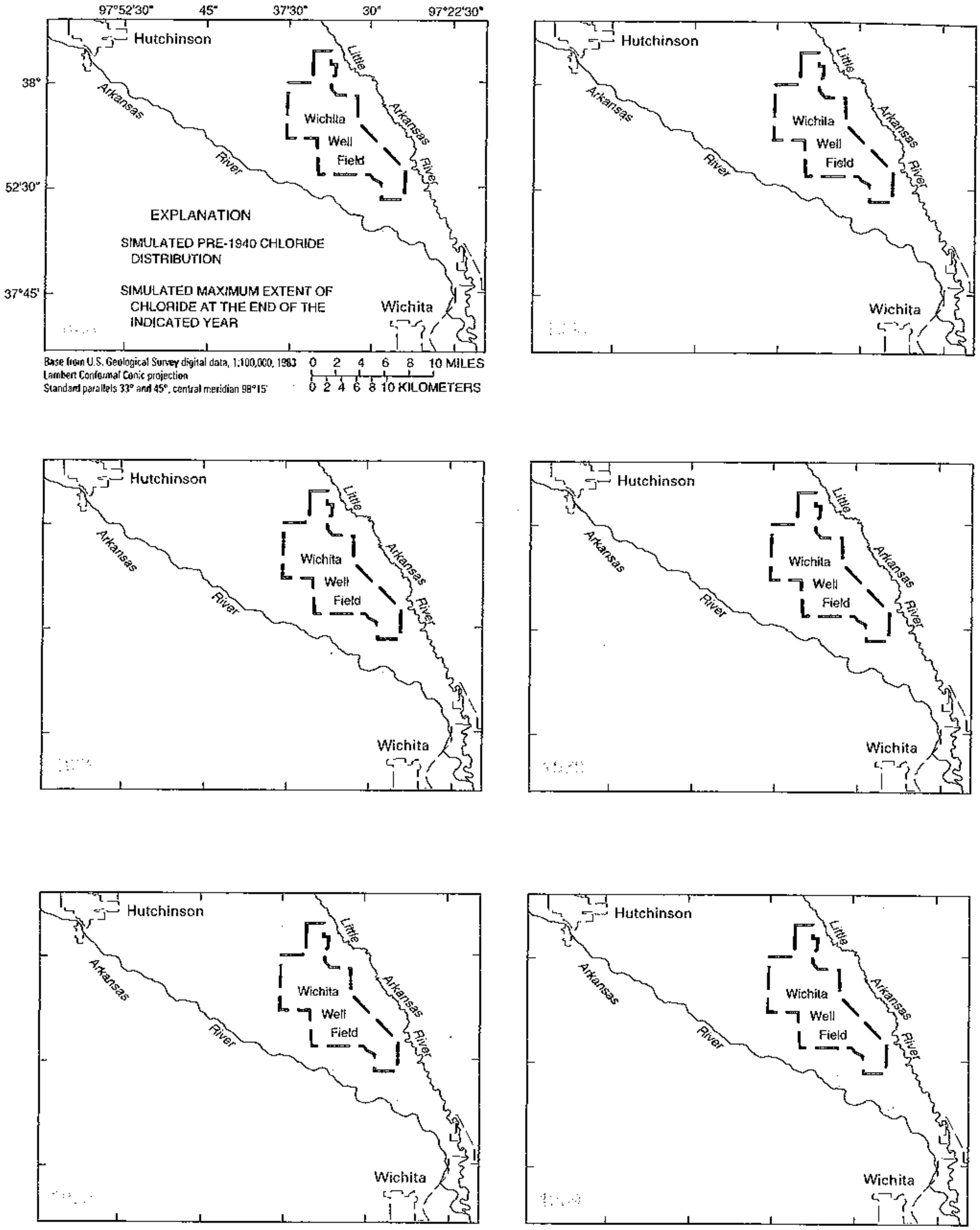


Figure 39. Simulated distribution of chloride from Arkansas River at end of each stress period in (*A*) upper, (*B*) middle, and (*C*) lower model layers—Continued.

have reached the edge of the Wichita well field as early as 1963 (fig. 39). Measured concentrations near the edge of the Wichita well field (fig. 19*A*), however, only are about 50 mg/L in the upper unit, probably because of dilution and dispersion of the chloride from the river as it moved through the aquifer. Measured natural chloride concentrations generally are the same or larger in the middle and lower units than in the upper unit (fig. 19). Because the particle-tracking simulations indicate that most of the chloride from the river stays in the upper unit, the measured chloride in the lower two units apparently is derived primarily from another source, such as the Hutchinson Salt Member of the Wellington Formation.

Discussion of Particle-Tracking Results, 1990–2019

Particle-tracking simulations for 1990–2019 show the distribution of chloride in the upper model layer north of the river expanding towards the Wichita well field and, south of the downstream reach of the river, expanding to the southwest (fig. 40). Particle distributions in the middle and lower layers also show expanded or modified distributions (fig. 40). The simulations with larger pumpage rates showed more extensive particle movement than simulations with smaller pumpage rates. In the middle and lower model layers, particles are simulated as reaching the edge of or entering the Wichita well field by the end of 1999.

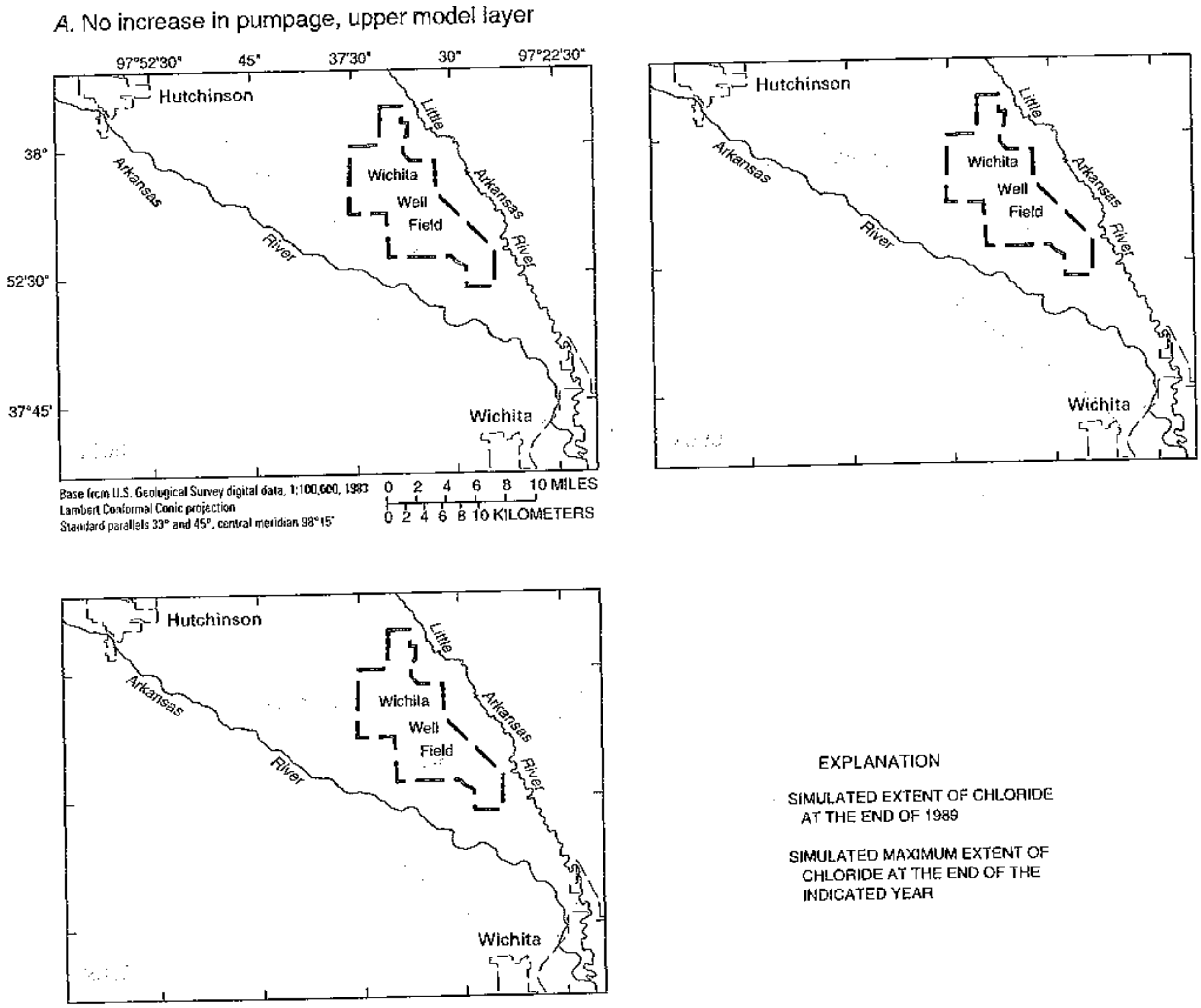


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989.

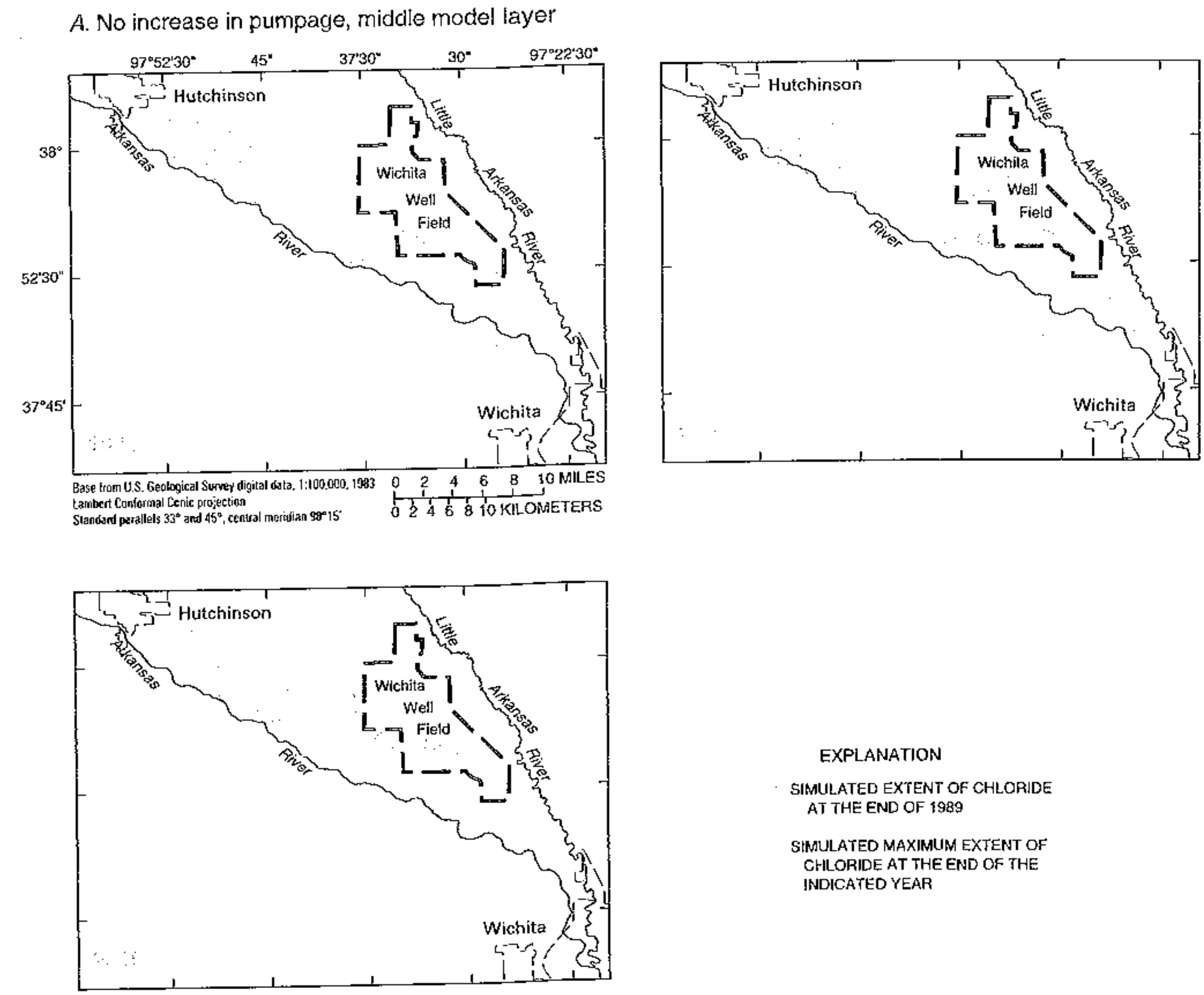


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

Because most of the municipal wells in the Wichita well field are screened in the middle and lower units of the aquifer, the chloride in the middle and lower layers would move toward the Wichita well field and eventually be found in well water.

SUMMARY

In 1988, a 4-year study was undertaken to improve understanding of the hydrologic and chemical interaction between the Arkansas River and the *Equus* beds aquifer in south-central Kansas so that water-management agencies can develop strategies to minimize the effects of poor river-water quality on the aquifer and of increasing ground-water withdrawals on consequent streamflow decreases. A network of 155 clustered observation wells at 55 sites was established along lines perpendicular and parallel to the Arkansas River between Hutchinson and Wichita. Water levels in these wells were measured monthly during most of the study period, and water samples for chemical analysis were obtained annually. On the basis of gamma logs and lithologic descriptions, the *Equus* beds sediment was divided into lower, middle, and upper units.

Analysis of water-level data from circa 1940 and 1989 shows that ground water near the Arkansas River flows parallel to the general direction of river flow, whereas ground water near the Little Arkansas River flows at an angle towards the river. Very little

A. No increase in pumpage, lower model layer

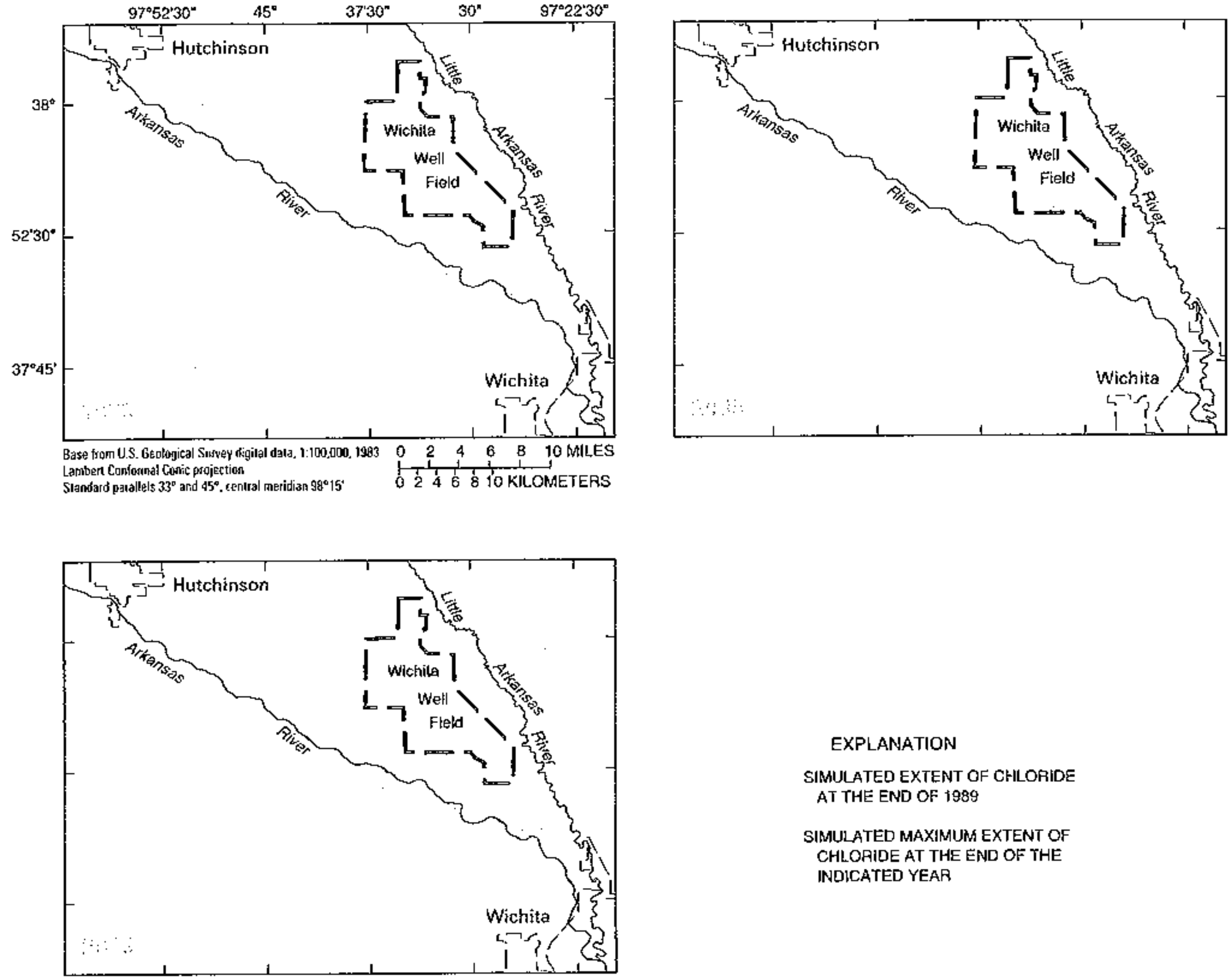


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

vertical flow was indicated by the data except near well sites EB-203 and EB-204.

Withdrawal of ground water by wells from the *Equus* beds aquifer has been increasing since 1940 when the Wichita well field was first developed. In the late 1950's and early 1960's, agricultural and industrial water use began increasing. Since the late 1970's, agricultural water use has been the single largest use of water. Continuous pumping of municipal wells and irrigation wells in the Wichita well field has caused ground-water levels to decline as much as 30 ft or more.

Two natural and three human-induced sources of chloride affect water quality in the study area. The natural sources are Arkansas River water and water from the Permian-age Wellington Formation. The human-induced sources are brine from oil-field, salt-mining, and salt-refining activities. Chloride concentrations measured in Arkansas River water at two sites during the winter of 1934–35 ranged from 392 to 1,895 mg/L. The median chloride concentration in Arkansas River water samples collected from five sites during 1988–91 ranged from 620 to 640 mg/L. The natural chloride concentration in the upper unit of the *Equus* beds aquifer is progressively smaller farther from the river. During this study the largest concentrations of natural chloride were found in the lower unit of the aquifer. Most of the natural chloride in the upper

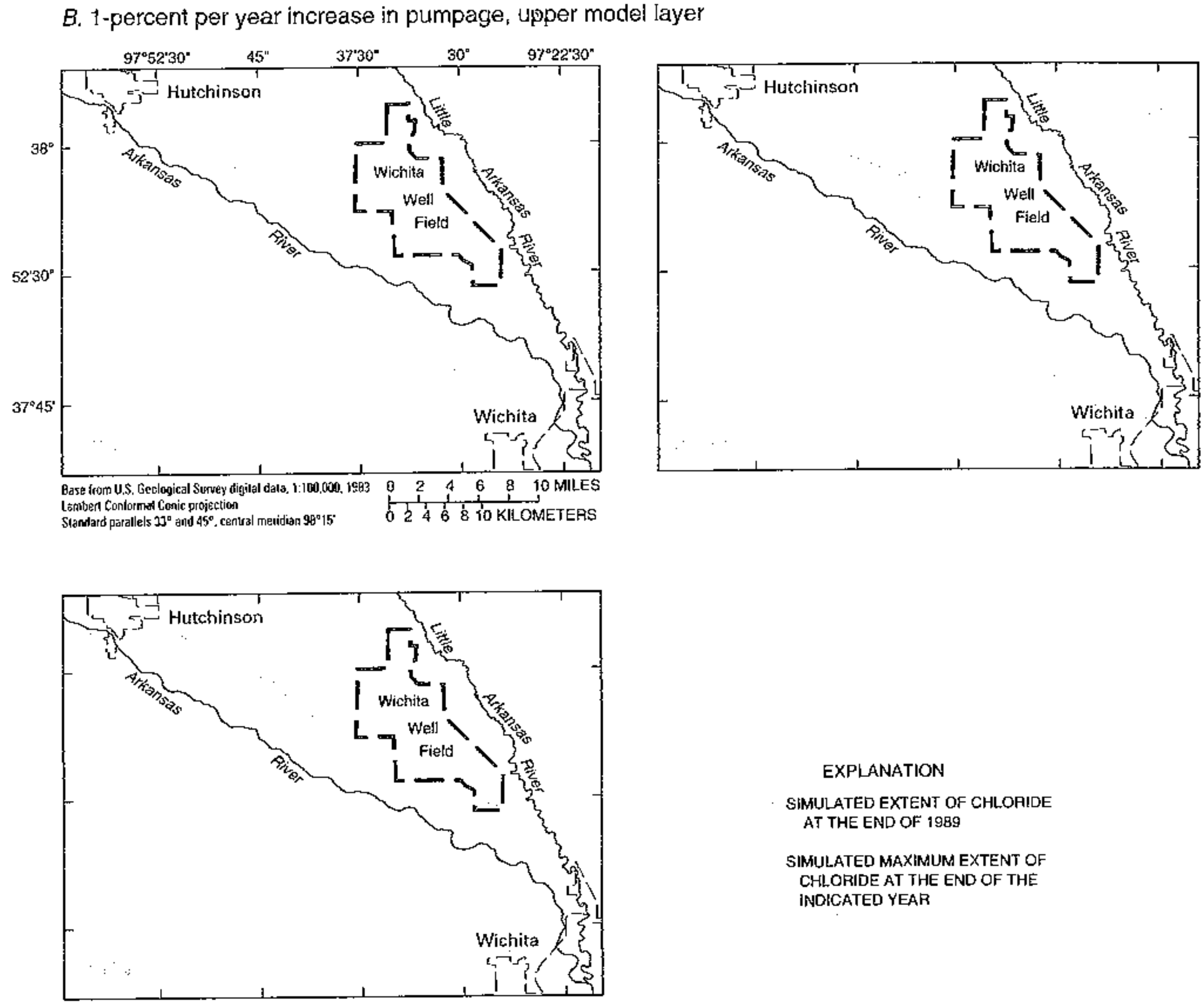


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

unit of the aquifer is probably from the Arkansas River, and most of the natural chloride in the lower unit of the aquifer is probably from saline water in the Wellington Formation.

The hydraulic interaction of the stream-aquifer system was simulated using the three-dimensional, finite-difference, flow-model program, MODFLOW. A steady-state model was developed and calibrated to simulate pre-1940 aquifer and stream conditions, and a transient model was developed and calibrated to simulate 1940–89 aquifer and stream conditions. The transient model then was used to estimate possible aquifer and stream conditions during 1990–2019.

Ground-water levels declined in the study area during 1940–89. Data and the results of model simulations indicate that water levels in the Wichita well field area declined as much as 30 ft or more because of increasing ground-water withdrawals from the aquifer. Near the Arkansas River, however, ground-water-level declines have been moderated by streamflow losses from the river.

In response to the declining ground-water levels, streamflow gains decreased in the Arkansas and Little Arkansas Rivers during 1940–89. In 1940, the Arkansas River had a simulated base-flow gain of about 21 ft^3/s within the model area, but by the end of 1989 had a simulated base-flow loss of about 52 ft^3/s.

B. 1-percent per year increase in pumpage, middle model layer

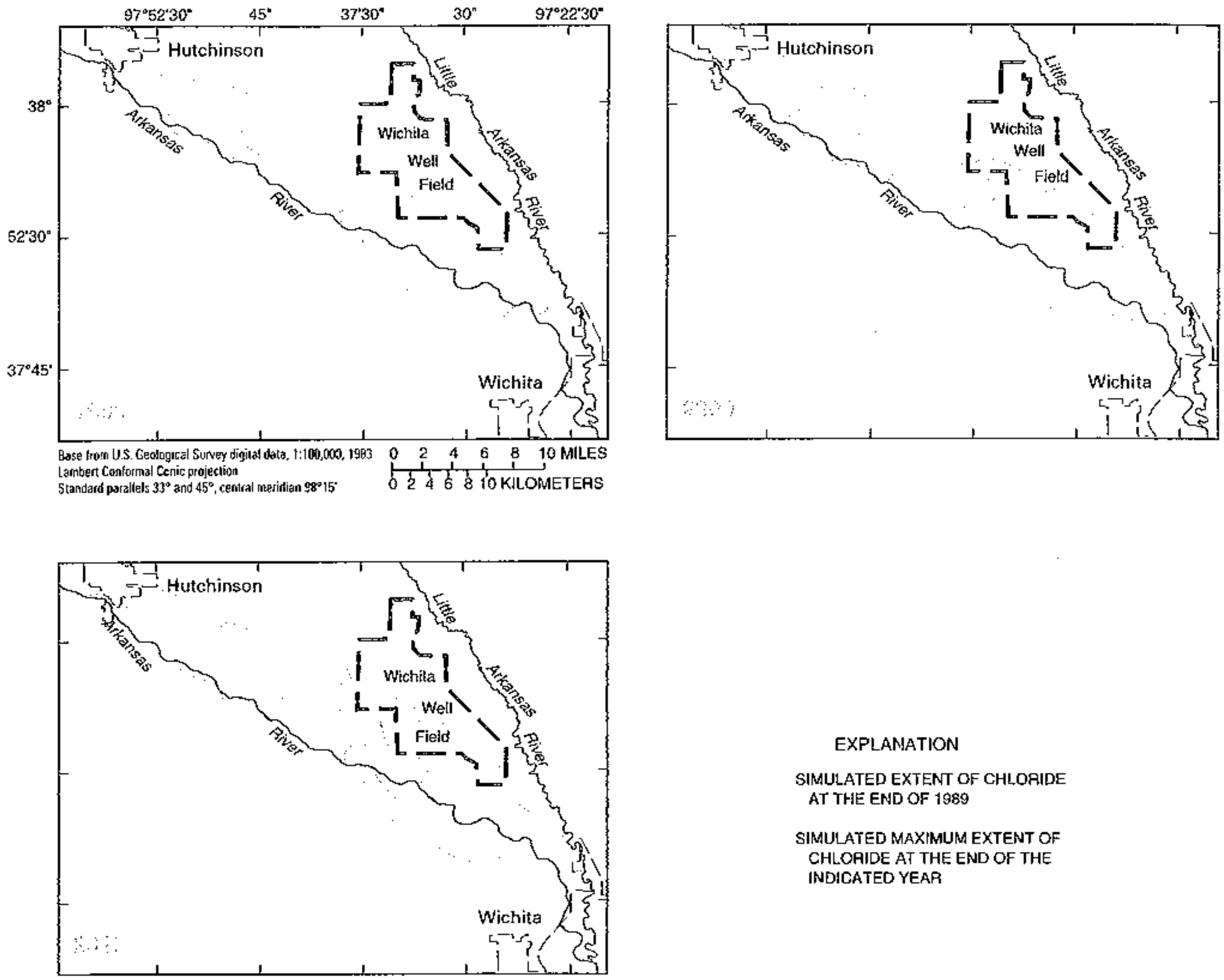


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

In 1940, the Little Arkansas River had a simulated base-flow gain of about 67 ft^3/s within the model area, but by the end of 1989 had a simulated base-flow gain of about 27 ft^3/s.

During 1940–89, the quantity of chloride discharged from the Arkansas River to the *Equus* beds aquifer increased in direct proportion to the volume of water loss from the river. On the basis of simulated streamflow and assuming that the chloride concentration in river water that moves into the aquifer is 630 mg/L, the chloride-load discharge from the river to the aquifer was estimated to be about 21 ton/d in 1940 and about 100 ton/d by 1989.

Results of simulations of hypothetical conditions during 1990–2019, using projected ranges of recharge, streamflow, and pumpage, indicate that water levels could decline from 1989 water levels by 0.2 to 78 ft in the central part of the Wichita well field and increase as much as 1.3 ft or decline as much as 1.2 ft near the Arkansas River by 2019 depending on the values of recharge, streamflow, and pumpage. With no increase in pumpage over 1989 quantities, simulations show that water levels in the Wichita well field probably will remain within 10 ft of 1989 water levels, depending on long-term climatic conditions that affect recharge and streamflow. The largest simulated water level decline from 1989 water levels in the Wichita

B. 1-percent per year increase in pumpage, lower model layer

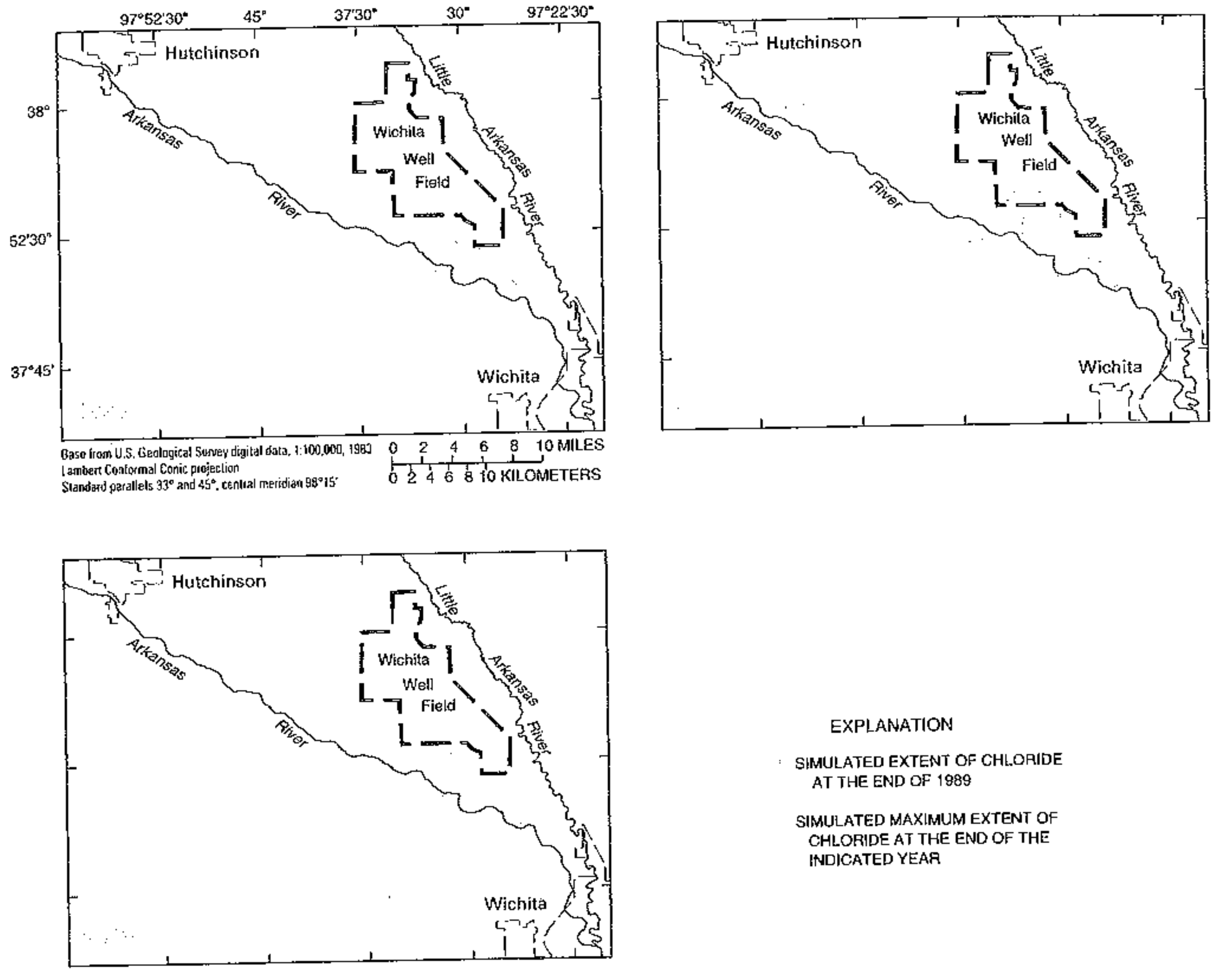


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

well field (78 ft) could occur under conditions of small recharge and streamflow and a 3-percent per year increase in pumpage. The largest simulated water-level decline from 1989 water levels near the Arkansas River (1.2 ft) could occur under conditions of small recharge and streamflow and a 3-percent per year increase in pumpage.

Results of simulations of hypothetical conditions during 1990–2019 indicate that streamflow losses from the Arkansas River could increase as pumpage increases because more river water would be lost to the aquifer. For hypothetical conditions of average recharge and streamflow, base-flow loss within the model area ranged from 59 to 117 ft^3/s, for no increase and a 3-percent per year increase in pumpage since 1989, respectively.

During 1990–2019, estimated chloride discharge from the Arkansas River to the *Equus* beds aquifer increased over 1989 estimated quantities in proportion to increases in loss of river water. Assuming hypothetical conditions of average recharge and streamflow and a 630-mg/L concentration of chloride in river water, the chloride-load discharge from the river in the model area could range from 110 to 200 ton/d by 2019.

The distribution in the aquifer of chloride from the river was simulated using the particle-tracking program MODPATH. Although MODPATH cannot

C. 2-percent per year increase in pumpage, upper model layer

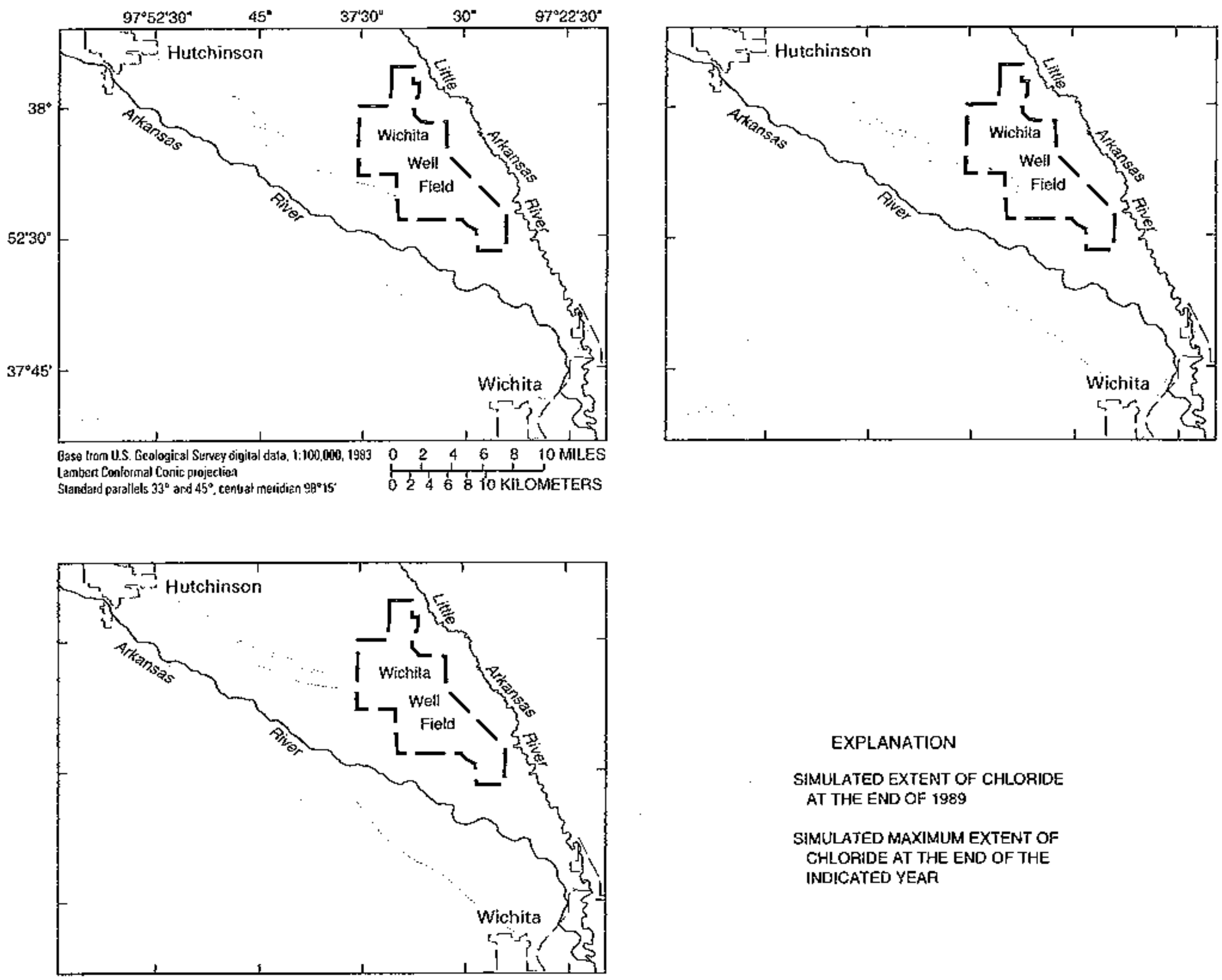


Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

calculate chloride concentration, it can show the path that dissolved constituents would follow assuming that there is no dispersion, mixing, or retardation of those constituents. Results from the steady-state and transient flow-model simulations were used to simulate the flow paths of particles representing chloride from the river to the aquifer and the distribution of those particles at the end of each stress period. During 1940–89, the simulated distribution of particles representing Arkansas River chloride in the aquifer expanded from relatively narrow bands near the river to a wider distribution within the aquifer and may have reached the edge of the Wichita well field as early as 1963. Most of the chloride stayed in the upper aquifer unit, but some moved into the lower two units. Particle-tracking simulations of 1990–2019 hypothetical conditions show the distribution of chloride expanding north towards the Wichita well field and southwest from the downstream reach of the Arkansas River. Simulations with larger pumpage rates show farther movement of chloride than simulations with smaller pumpage rates.

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C. 2-percent per year increase in pumpage, middle model layer

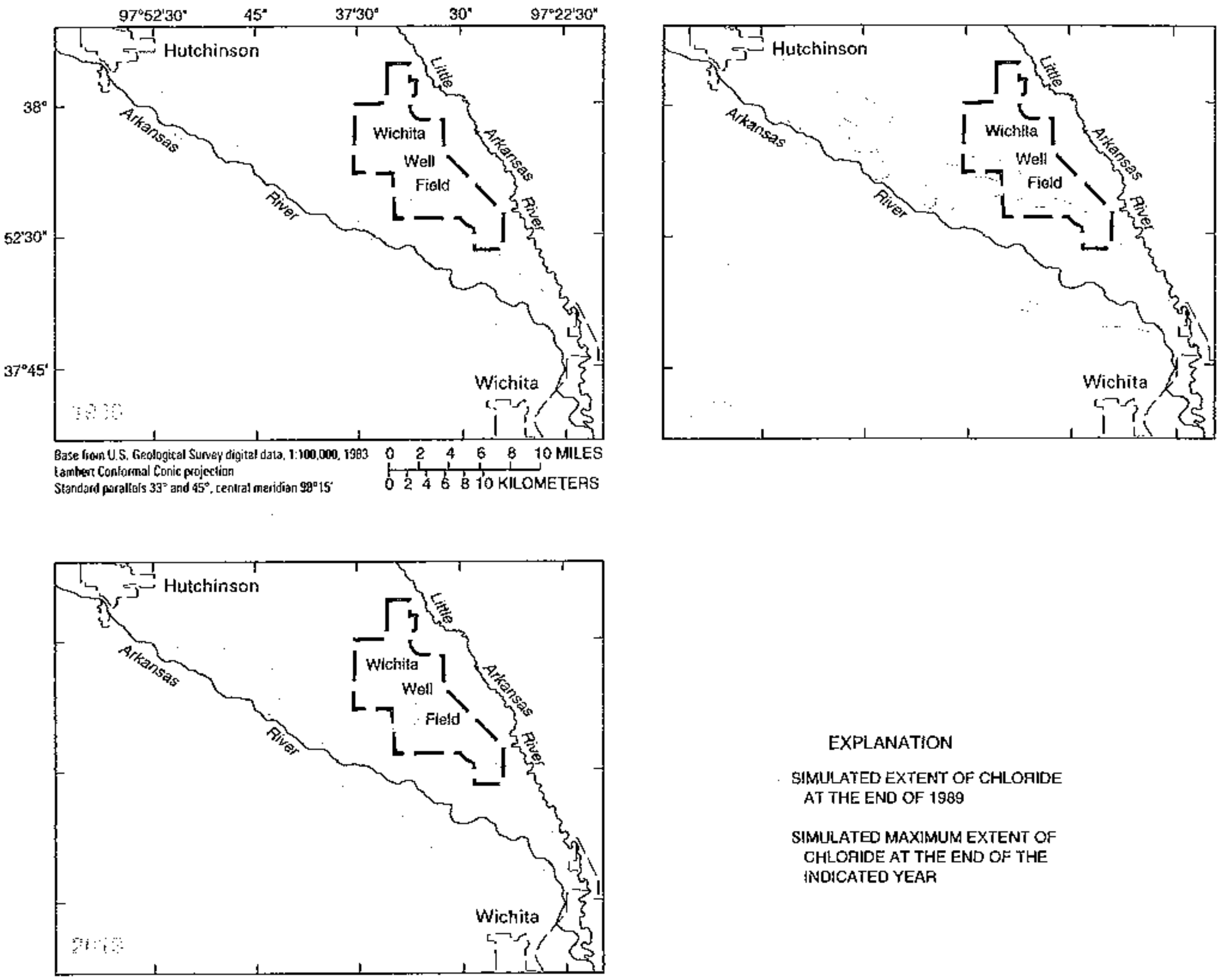

Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

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C. 2-percent per year increase in pumpage, lower model layer

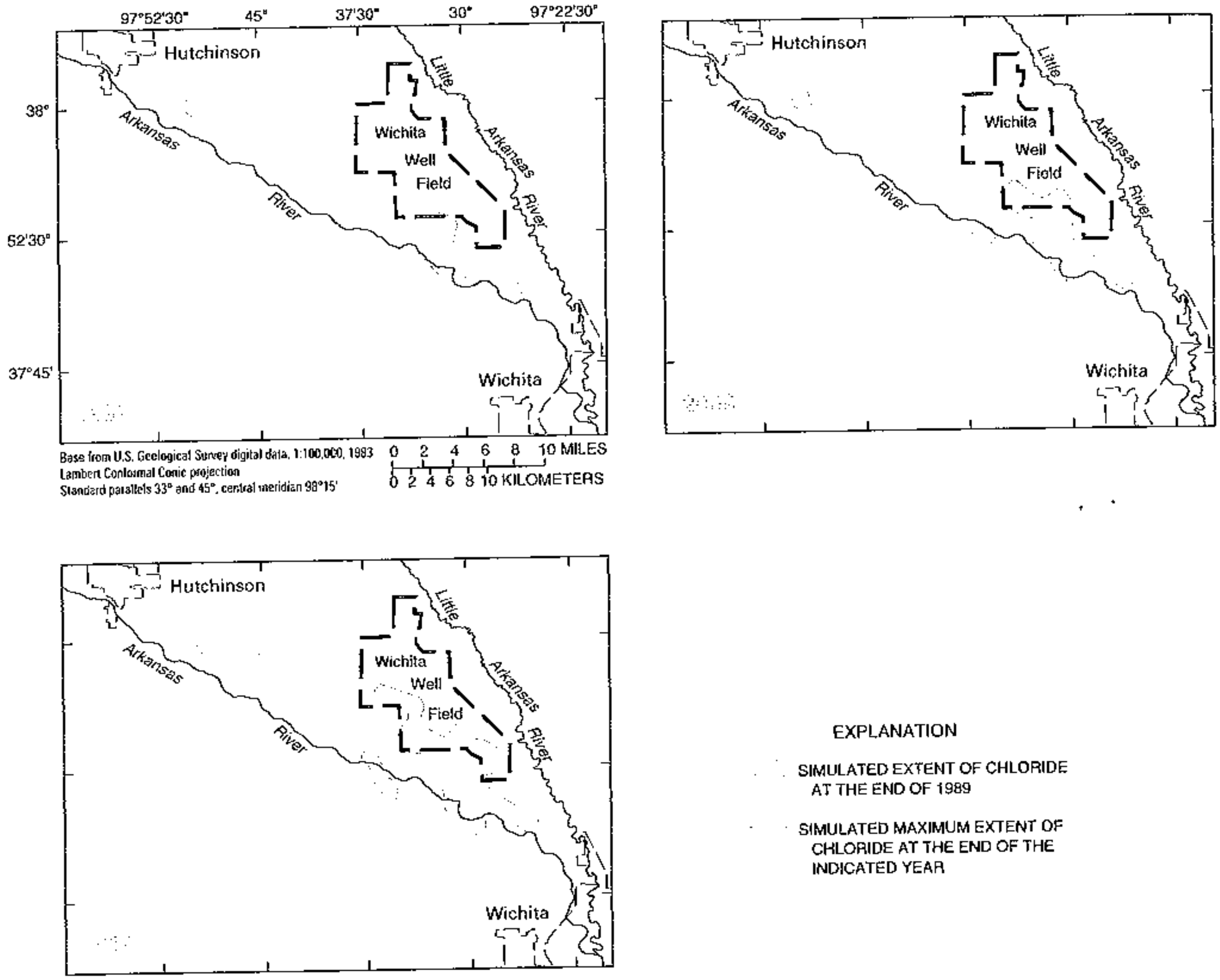

Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

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D. 3-percent per year increase in pumpage, upper model layer

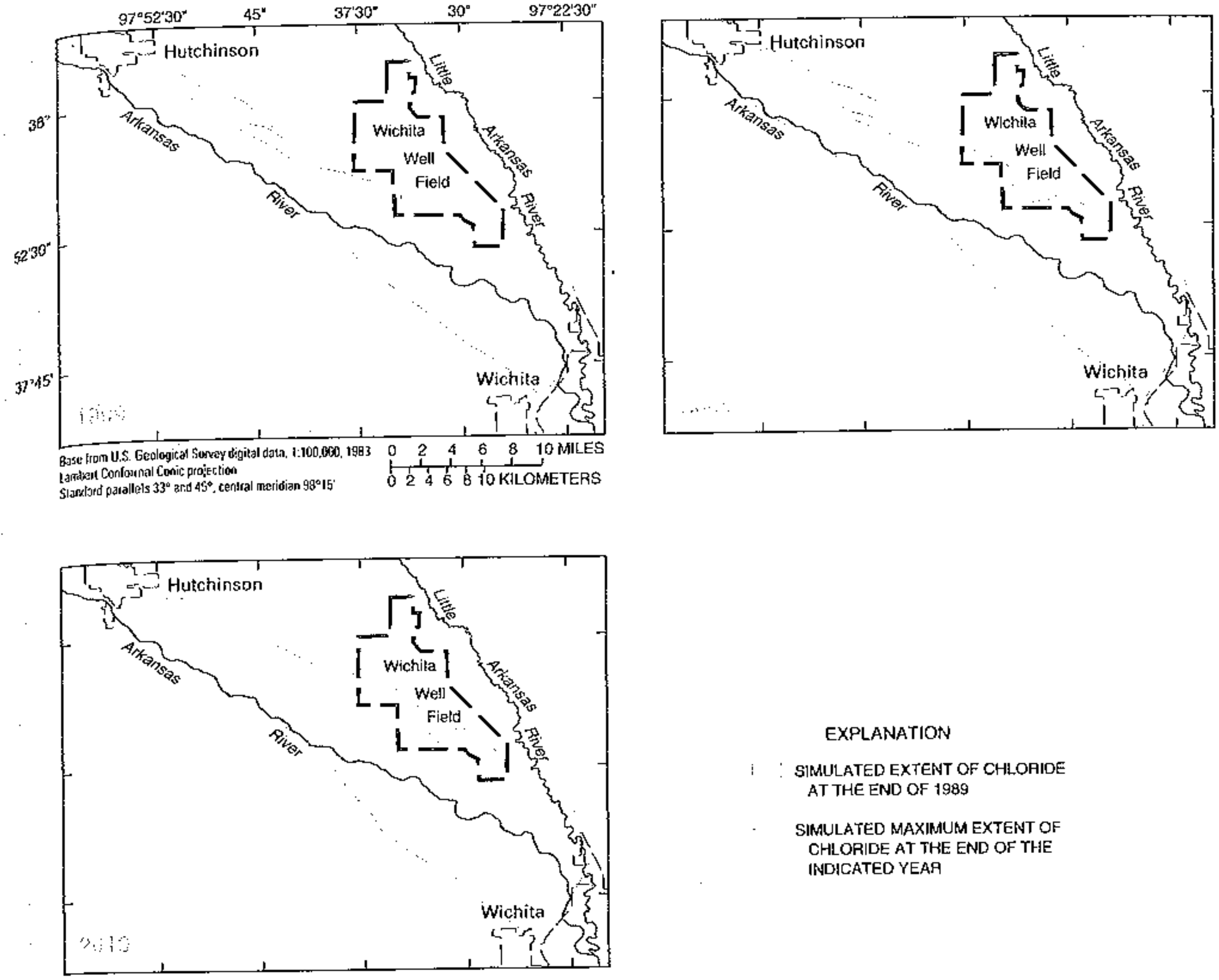

Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

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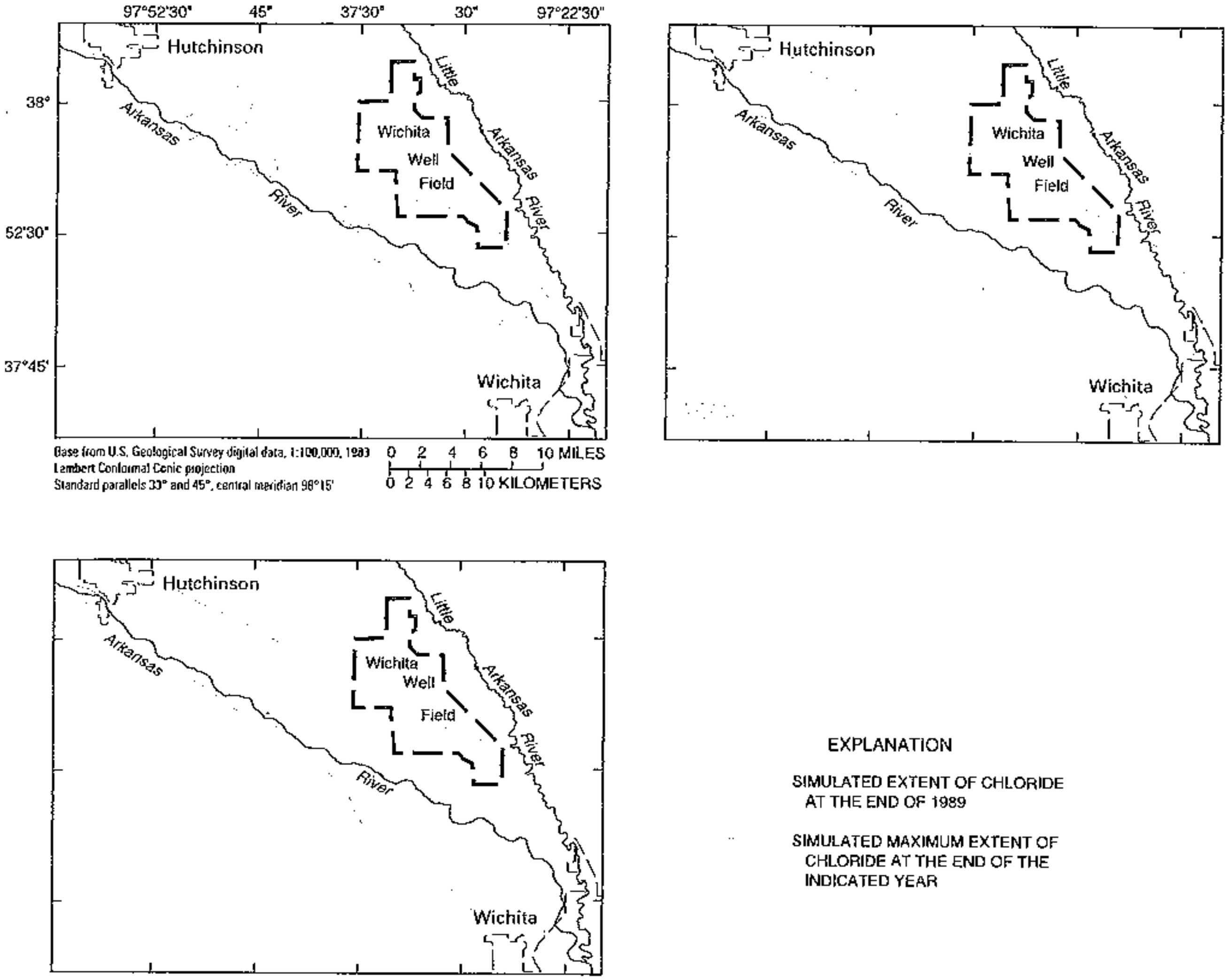
D. 3-percent per year increase in pumpage, middle model layer



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D. 3-percent per year increase in pumpage, lower model layer

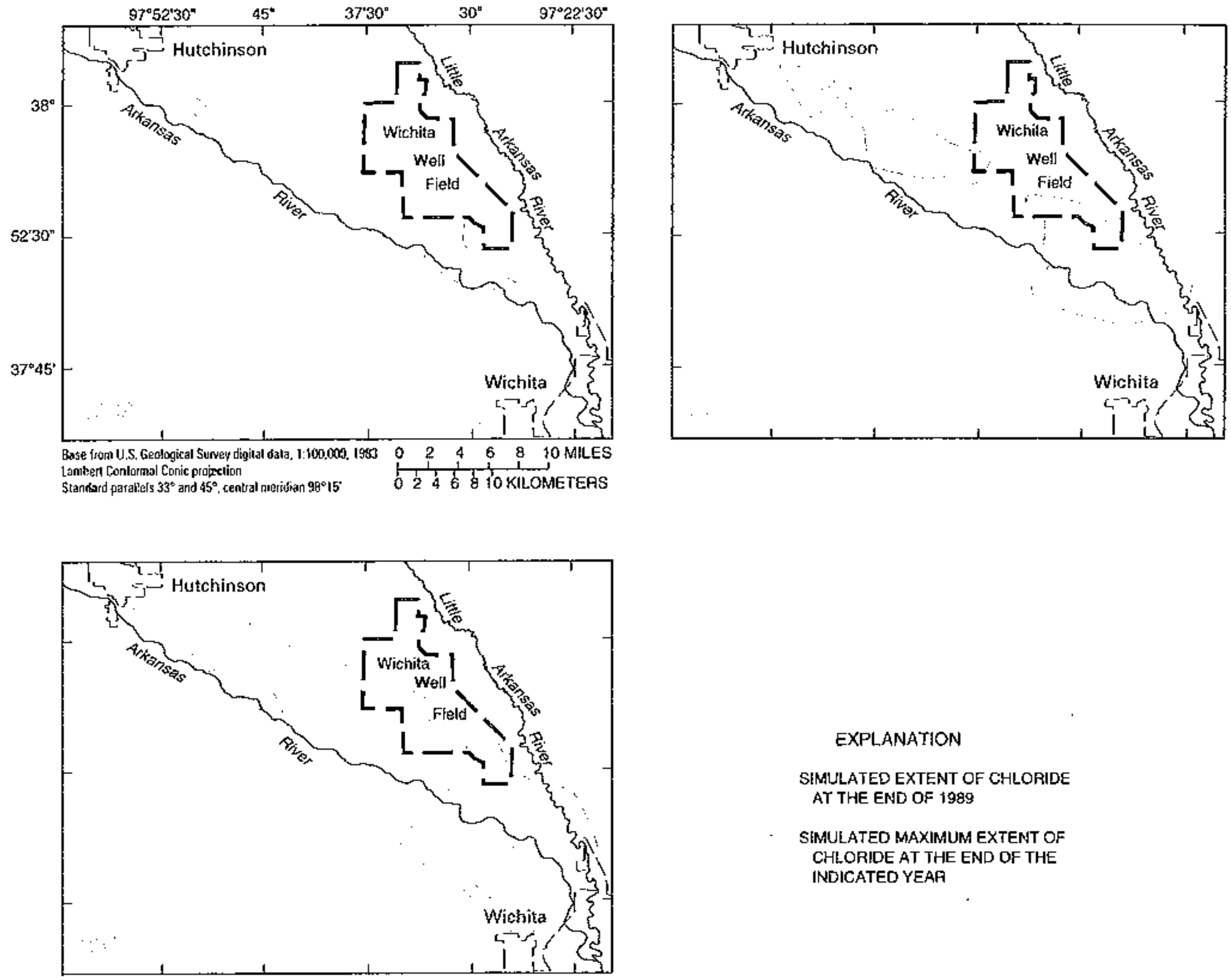

Figure 40. Simulated extent of chloride from Arkansas River at end of each stress period in upper, middle, and lower model layers assuming (*A*) no increase in pumpage since 1989, (*B*) a 1-percent per year increase in pumpage since 1989, (*C*) a 2-percent per year increase in pumpage since 1989, and (*D*) a 3-percent per year increase in pumpage since 1989—Continued.

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SUPPLEMENTAL INFORMATION

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249

[There are no lithologic logs for wells at well sites EB-211 and EB-212. Lithologic descriptions for abandoned wells near well sites USGS-H-1, USGS-H-2, USGS-H-3, USGS-H-4, NAS-1 and NAS-2 are similar to previously published lithologic logs (Williams, 1946). Wells 69, 59, 72, and 76 (Williams, 1946) correspond to well sites USGS-H-1, USGS-H-2, USGS-H-3, and USGS-H-4, respectively. Well 67 (Williams, 1946) corresponds to well sites NAS-1 and NAS-2. All altitudes are referenced to sea level and are reported to the nearest 0.1 foot. Depth of well is reported in feet below land surface]

EB-201-C.—Drilled December 10, 1986.
Altitude of land surface, 1,380.7 feet.

	Thickness, in feet	Depth, in feet
Clay, silty	5	5
Clay	10	15
Sand, coarse-grained, and gravel	10	25
Sand, coarse- to medium-grained	10	35
Clay, light-gray	8	43
Sand, medium- to fine-grained	12	55
Clay, silty, tan	8	63
Sand, coarse-grained, and gravel	7	70
Sand, medium- to fine-grained	6	76
Clay, silty, tan	3	79
Sand, medium- to fine-grained	14	93
Clay, silty	4	97
Sand, medium- to fine-grained	1	98
Clay and sand, medium- to coarse-grained	20	118
Sand, medium- to coarse-grained	8	126
Clay, silty, tan	4	130
Sand, medium- to coarse-grained	35	165
Sand and shale pieces	5	170
Shale, gray	8	178

EB-202-C.—Drilled December 6, 1986.
Altitude of land surface, 1,379.5 feet.

	Thickness, in feet	Depth, in feet
Soil, silty	2	2
Sand, fine-grained	6	8
Sand, medium- to coarse-grained	2	10
Sand, coarse-grained, and gravel	25	35
Clay, sandy, silty, tan	25	60
Clay, silty, tan to orange	18	78
Clay, silty, tan to gray	67	145
Clay, gray	3	148
Sand, fine- to medium-grained	30	178
Sand, medium- to coarse-grained	12	190
Shale, with traces of gypsum	5	195
Shale	3	198

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-203-C.—Drilled December 2, 1986.
Altitude of land surface, 1,380.8 feet.

	Thickness, in feet	Depth, in feet
Clay, silty, brown	4	4
Sand, fine- to medium-grained, and gravel	39	43
Silt, clayey, tan	34	77
Sand and gravel; clayey, arkosic	35	112
Silt, clayey, sandy, tan	48	160
Sand, medium- to coarse-grained, and gravel; arkosic	56	216
Shale, gray, with thin limestone and gypsum layers	12	228

EB-204-C.—Drilled November 18, 1986.
Altitude of land surface, 1,378.2 feet.

	Thickness, in feet	Depth, in feet
Silt, sandy	10	10
Sand, coarse-grained, and gravel	38	48
Clay, silty	10	58
Sand, medium-grained, and gravel	40	98
Sand, fine-grained, and clay	104	202
Gravel	1	203
Clay	2	205
Sand, fine- to medium-grained	28	233
Shale	4	237

EB-205-C.—Drilled November 24, 1986.
Altitude of land surface, 1,380.5 feet.

	Thickness, in feet	Depth, in feet
Silt, clayey, dark-brown	10	10
Sand, medium- to coarse-grained, and gravel	8	18
Clay, silty, gray to tan	10	28
Clay, silty; sand, coarse-grained; and gravel	10	38
Sand, coarse-grained, and gravel	10	48
Sand, medium- to coarse-grained	10	58
Sand, medium-grained, clayey	10	68
Clay, tan	7	75
Sand, fine-grained	5	80
Sand, medium- to coarse-grained	18	98
Sand, medium- to coarse-grained	75	173
Clay, hard, dark-gray	35	208
Shale	3	211

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-206-C.—Drilled November 9, 1986.
Altitude of land surface, 1,396.9 feet.

	Thickness, in feet	Depth, in feet
Silt, clayey	10	10
Sand, fine-grained, silty	25	35
Sand, coarse-grained	30	65
Sand, coarse-grained, and gravel	5	70
Clay, sandy	25	95
Sand, medium- to fine-grained	70	165
Sand, medium- to coarse-grained, and gravel	95	260
Shale	8	268

EB-207-C.—Drilled December 14, 1986.
Altitude of land surface, 1,392.8 feet.

	Thickness, in feet	Depth, in feet
Clay, brown	42	42
Sand, medium- to coarse-grained, and gravel	60	102
Silt, clayey, and clay, silty	18	120
Sand, fine- to medium-grained	30	150
Silt, clayey	10	160
Sand, medium- to coarse-grained, and gravel	30	190
Sand, medium- to coarse-grained, clayey	10	200
Sand, medium-grained	40	240
Sand, medium- to coarse-grained	6	246
Shale	4	250

EB-208-C.—Drilled October 28, 1980.
Altitude of land surface, 1,418.7 feet.

	Thickness, in feet	Depth, in feet
Soil, black	5	5
Clay, tan, and sand, fine-grained	5	10
Sand, coarse-grained, with a few thin clay layers	75	85
Clay, green	15	100
Clay, blue-gray	23	123

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-209-C.—Drilled May 26, 1987.
Altitude of land surface, 1,423.7 feet.

	Thickness, in feet	Depth, in feet
Soil	4	4
Sand, coarse-grained, arkosic	51	55
Clay, silty, sandy	5	60
Sand, fine-grained	30	90
Clay, tan, white, brown	10	100
Sand, fine- to medium-grained, with a few thin clay layers	73	173
Shale, tan, yellow, grading to dark-gray	9	182

EB-210-C.—Drilled June 1, 1987.
Altitude of land surface, 1,424.7 feet.

	Thickness, in feet	Depth, in feet
Clay, silty, sandy, red-brown	25	25
Clay, silty, sandy, gray	5	30
Sand, fine-grained	8	38
Clay, silty, sandy, gray	17	55
Sand, fine-grained	25	80
Sand, medium- to coarse-grained, arkosic	15	95
Clay, silty, tan, brown	21	116
Sand, fine- to coarse-grained, arkosic	70	186
Clay, silty, tan, brown	1	187
Sand, fine- to coarse-grained, arkosic, gypsiferous	55	242

EB-213-C.—Drilled April 6, 1988.
Altitude of land surface, 1,475.8 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, sandy	12	12
Sand and gravel; arkosic	37	49
Clay, tan, yellow	1	50
Sand and gravel; arkosic	32	82
Silt, clayey, sandy, tan, gray	21	103
Sand, fine- to medium-grained	9	112
Sand and gravel; arkosic	4	116
Clay, silty, sandy, tan, gray	6	122
Sand, fine- to medium-grained, arkosic, with a few thin clay layers	73	195
Clay, gray, and pieces of green shale	5	200
Sand and gravel; arkosic	16	216
Clay, white, grading to sand, clayey, red-brown	4	220
Sand, fine- to medium-grained	18	238
Clay, gray	2	240
Sand and gravel	2	242
Clay, silty, and silt, clayey	40	282
Shale, maroon, gray, green, weathered	23	305
Shale, dark-gray	8	313

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-214-C.—Drilled April 12, 1988.
Altitude of land surface, 1,473.1 feet.

	Thickness, in feet	Depth, in feet
Soil	4	4
Sand, fine-grained, and gravel; arkosic	54	58
Clay, silty, gray	16	74
Sand and gravel; arkosic	23	97
Clay, tan, yellow	12	109
Silt, gray	84	193
Sand, fine-grained, and gravel; arkosic	23	216
Clay, sandy, tan	19	235
Clay, silty, sandy, gray	36	271
Sand and gravel; arkosic	9	280
Silt, sandy, red to brown	8	288
Sand and gravel, some dark red-brown ironstone	3	291
Shale, gray, weathered	19	310
Shale, dark-gray	20	330

EB-215-C.—Drilled March 31, 1988.
Altitude of land surface, 1,464.9 feet.

	Thickness, in feet	Depth, in feet
Soil, brown	5	5
Silt, sandy, tan, brown	5	10
Sand and gravel; arkosic	23	33
Clay, tan, gray, yellow	3	36
Sand and gravel; arkosic, pink	12	48
Sand, fine- to medium-grained, tan	70	118
Clay, silty, gray	15	133
Sand, fine- to medium-grained, arkosic, pink	25	158
Clay, sandy, tan, yellow	9	167
Sand	5	172
Clay, tan, gray	3	175
Sand, fine- to coarse-grained, arkosic, pink	37	212
Clay, tan to gray	3	215
Sand, fine, to medium-grained, arkosic, pink	13	228
Clay, tan	2	230
Sand, fine- to medium-grained, arkosic, pink	55	285
Shale, gray	13	298

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-216-C.—Drilled November 19, 1987.
Altitude of land surface, 1,464.3 feet.

	Thickness, in feet	Depth, in feet
Clay, sandy, brown	7	7
Sand and gravel; arkosic	81	88
Clay	2	90
Sand and gravel	5	95
Clay	9	104
Sand and gravel	6	110
Clay	6	116
Sand and gravel	28	144
Clay	5	149
Sand and gravel	61	210
Clay	4	214
Sand and gravel	30	244
Clay	1	245
Sand and gravel	46	291
Clay	1	292
Sand and gravel	8	300
Sandstone	5	305
Sand, fine-grained	9	314
Shale, gray, some anhydrite	7	321

EB-217-C.—Drilled November 11, 1987.
Altitude of land surface, 1,460.0 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, sandy, brown	1	1
Sand and gravel	129	130
Clay	3	133
Sand and gravel	18	151
Clay	4	155
Sand	15	170
Clay	2	172
Sand	9	181
Clay	1	182
Sand	16	198
Clay	8	206
Sand	44	250
Shale	2	252

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-218-C.—Drilled November 5, 1987.
Altitude of land surface, 1,472.7 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, silty, tan to brown	15	15
Sand, fine- to medium-grained, and gravel; with clay, red to brown	88	103
Shale	5	108

EB-219-C.—Drilled October 23, 1987.
Altitude of land surface, 1,465.6 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, silty, tan to brown	15	15
Sand, fine- to medium-grained, tan to brown	73	88
Clay, silty, brown	25	113
Sand and gravel	17	130
Shale, gray	3	133

EB-220-C.—Drilled November 15, 1988.
Altitude of land surface, 1,337.3 feet

	Thickness, in feet	Depth, in feet
Soil, silty, brown	7	7
Sand and gravel; arkosic, pink	40	47
Shale, gray	13	60

EB-221-C.—Drilled November 7, 1988.
Altitude of land surface, 1,340.2 feet.

	Thickness, in feet	Depth, in feet
Soil, sandy, brown	2	2
Sand, fine-grained, tan	5	7
Sand and gravel, arkosic, tan to orange	44	51
Dolomite, sandy, tan to yellow	1	52
Shale, tan	6	58
Shale, gray	6	64
Sandstone, fine-grained, gray	1	65
Shale, gray	6	71

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-222-C.--Drilled November 11, 1988.
Altitude of land surface, 1,337.7 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, silty, sandy, brown	5	5
Sand and gravel; arkosic, tan	12	17
Sand and gravel; arkosic, tan to orange	62	79
Dolomite or limestone	2	81
Shale, gray	10	91

EB-223-C.—Drilled November 14, 1988.
Altitude of land surface, 1,337.6 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, brown	6	6
Sand, fine-grained, tan	5	11
Sand and gravel; arkosic, pink	32	43
Sand, clayey, yellow tan	2	45
Sand, fine- to medium-grained, tan	10	55
Clay, sandy, tan to brown	3	58
Sand, fine-grained, tan	13	71
Clay, sandy, tan to yellow	3	74
Sand, fine-grained, tan	6	80
Sand, fine-grained, tan, with a few thin clay layers	8	88
Sandstone, white	3	91
Sand, fine-grained, tan	2	93
Sandstone, white	3	96
Shale, gray	5	101

EB-224-C.—Drilled November 17, 1988.
Altitude of land surface, 1,342.2 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, sandy	11	11
Sand, medium- to coarse-grained, and gravel; arkosic, pink	57	68
Clay, sandy, tan to yellow	2	70
Clay, silty, sandy, tan	12	82
Sand, fine-grained, and silt, clayey, tan	60	142
Sand, fine-grained	12	154
Sandstone, white	6	160
Shale, gray	1	161

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-225-C.—Drilled November 22, 1988.
Altitude of land surface, 1,350.3 feet.

	Thickness, in feet	Depth, in feet
Soil, brown	8	8
Silt, clayey, brown	2	10
Sand, fine- to medium-grained, and clay, yellow; clay layers near bottom of interval	28	38
Sand, fine- to coarse-grained, and gravel	2	40
Sand, fine- to coarse-grained, tan	18	58
Clay, tan	2	60
Sand, fine- to coarse-grained, tan	12	72
Sand, coarse-grained, and gravel	9	81
Silt, sandy, tan	9	90
Clay, silty, gray	2	92
Sand, fine- to coarse-grained, tan	18	110
Sand, fine- to medium-grained, tan	7	117
Clay, sandy, yellow	4	121
Sand, fine- to coarse-grained, tan	4	125
Clay, sandy, tan	4	129
Sand, fine- to coarse-grained, tan	6	135
Clay, tan	10	145
Sand, fine-grained, tan	17	162
Clay, tan	5	167
Sand, fine- to coarse-grained, tan, and gravel	9	176
Shale, silty, gray	6	182

EB-226-C.—Drilled November 26, 1988.
Altitude of land surface. 1,354.1 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, brown	10	10
Clay, sandy, red	9	19
Sand, fine- to coarse-grained, tan	18	37
Sand, fine- to medium-grained, silty, tan	9	46
Clay, silty, tan to yellow	9	55
Sand, fine- to coarse-grained, tan	6	61
Sand, fine- to medium-grained, silty, tan	20	81
Sand, fine- to coarse-grained, and gravel, tan	30	111
Clay, sandy, yellow	1	112
Sand, fine- to medium-grained, and gravel; tan	14	126
Clay, tan	2	128
Sand, fine- to medium-grained, tan	8	136
Clay, gray	1	137
Sand, fine- to coarse-grained, tan	43	180
Shale, gray	4	184

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-227-C.—Drilled December 1, 1988.
Altitude of land surface, 1,355.1 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, brown	7	7
Clay, sandy, red to brown	10	17
Sand, fine-grained, clayey, gray	11	28
Sand and gravel; arkosic, orange	30	58
Clay, silty, sandy, yellow	2	60
Sand and gravel; arkosic, with ironstone	35	95
Clay, silty, sandy, tan to yellow to red	17	112
Sand, fine-grained, silty, tan	14	126
Shale, gypsiferous, tan to gray	19	145

EB-228-C.—Drilled December 7, 1988.
Altitude of land surface, 1,493.8 feet.

	Thickness, in feet	Depth, in feet
Clay, sandy, brown	4	4
Sand, fine- to medium-grained, tan	6	10
Sand, fine- to coarse-grained, tan	14	24
Clay, sandy, gray	6	30
Sand, fine- to coarse-grained, and gravel; tan	20	50
Sand, fine-grained, and gravel; gray	7	57
Clay, silty, green to gray	3	60
Sand, fine- to coarse-grained, and gravel; tan, with a few thin clay layers	25	85
Shale, brown to red, with some gypsum	15	100

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-229-C.—Drilled December 9, 1988.
Altitude of land surface, 1,493.8 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, silty, brown	4	4
Gravel and sand, medium- to coarse-grained; tan; with a few thin clay layers	13	17
Sand, medium- to coarse-grained, and gravel	6	23
Clay, yellow	1	24
Sand, fine- to coarse-grained, tan	15	39
Clay, sandy, gray, and gravel	6	45
Sand, fine- to coarse-grained, tan, and gravel	25	70
Clay, gray	1	71
Sand, fine- to coarse-grained, tan	4	75
Clay, gray	1	76
Sand, clayey, tan, and gravel	7	83
Clay, sandy, tan, and gravel	3	86
Sand, fine- to coarse-grained, tan	4	90
Clay, gray	1	91
Sand, fine- to coarse-grained, tan	4	95
Clay, red	1	96
Clay, yellow	1	97
Sand, fine-grained, tan	4	101
Sand, fine-grained, silty	6	107
Shale, red	14	121

EB-230-C.—Drilled December 13, 1988.
Altitude of land surface, 1,497.3 feet.

	Thickness, in feet	Depth, in feet
Soil, brown	6	6
Sand, fine-grained, tan	3	9
Sand and gravel; arkosic, orange, with a few thin clay layers	74	83
Sand, silty, red to brown	7	90
Clay, silty, sandy, gray	16	106
Sand, clayey, red to gray	9	115
Sand, fine- to medium-grained, tan	4	119
Sand, fine-grained	16	135
Sand, fine- to coarse-grained, arkosic	38	173
Clay, sandy, red to brown	4	177
Sand and gravel; arkosic, pink	9	186
Shale, silty, sandy, maroon to green	5	191

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-231-C.—Drilled December 31, 1988.
Altitude of land surface, 1,498.8 feet.

	Thickness, in feet	Depth, in feet
Clay, silty, sandy, brown, yellowish-tan	8	8
Sand, fine-grained, yellow, and gravel	9	17
Clay, yellow, tan	12	29
Clay, silty, green	4	33
Sand, fine- to coarse-grained, clayey	27	60
Clay, silty, green	1	61
Sand, fine- to coarse-grained, tan	2	63
Clay, silty, green	1	64
Sand, fine- to medium-grained, clayey, tan	9	73
Clay, sandy, yellow	3	76
Sand, fine- to medium-grained, clayey, yellow	12	88
Clay, silty, pink, green	10	98
Caliche, white	1	99
Shale, red, green	22	121

EB-232-C.—Drilled January 9, 1989.
Altitude of land surface, 1,496.6 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, sandy, brown	3	3
Sand, fine-grained, tan, and gravel	5	8
Clay, sandy, gray	10	18
Sand, fine-grained, and gravel; tan to gray	32	50
Clay, tan	4	54
Sand and gravel; arkosic	26	80
Sand, fine-grained, tan	12	92
Clay, sandy, tan	1	93
Sand, fine-grained, and gravel	17	110
Sand and gravel; tan to orange	10	120
Sand, fine-grained, tan	10	130
Clay, tan	1	131
Sand, coarse-grained, and gravel; tan to orange	12	143
Shale, green, gray, with some gypsum	11	154

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-233-C.—Drilled January 13, 1989.
Altitude of land surface, 1,553.1 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, brown	4	4
Clay, silty, tan	50	54
Sand, fine- to coarse-grained, arkosic, orange	33	87
Clay, silty, sandy, tan, gray	13	100
Sand and gravel; tan	8	108
Sand, clayey, tan	6	114
Sand, fine-grained, tan	16	130
Sand, coarse-grained, and gravel; arkosic, orange with pieces of ironstone	15	145
Shale, silty, red to green	7	152

EB-234-C.—Drilled January 17, 1989.
Altitude of land surface, 1,548.5 feet.

	Thickness, in feet	Depth, in feet
Clay, tan to gray	34	34
Sand, fine- to coarse-grained, and gravel; tan	34	68
Clay, sandy, tan, brown	22	90
Clay, sandy, with interbedded sand layers	21	111
Shale, silty	10	121

EB-235-C.—Drilled January 19, 1989.
Altitude of land surface, 1,551.4 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, silty	2	2
Clay, silty, sandy, brown, yellowish to tan	26	28
Sand, fine- to coarse-grained, tan	17	45
Clay, tan, and sand layers	17	62
Clay, tan	3	65
Shale, red, green	11	76

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-236-C.—Drilled December 9, 1989.
Altitude of land surface, 1,476.6 feet.

	Thickness, in feet	Depth, in feet
	17	17
Clay, tan, gray	1	18
Sand, fine-grained	5	23
Clay, gray	7	30
Sand, fine- to medium-grained	32	62
Sand, fine- to coarse-grained, tan, and gravel; arkosic	6	68
Clay, silty, sandy, tan	4	72
Clay, gray	21	93
Sand and gravel; tan	7	100
Sand, fine- to coarse-grained, clayey, and gravel; tan	120	220
Sand	25	245
Shale, gray		

EB-237-D.—Drilled December 17, 1989.
Altitude of land surface, 1,516.8 feet.

	Thickness, in feet	Depth, in feet
	3	3
Soil, clayey, silty, brown	31	34
Clay, tan, red to brown	9	43
Clay, silty, tan, red-brown	2	45
Shale, red; weathered	31	76
Shale, silty, red to maroon, green	1	77
Dolomite (?)	8	85
Shale, silty, red to maroon		

EB-238-C.—Drilled February 12, 1990.
Altitude of land surface, 1,394.0 feet.

	Thickness, in feet	Depth, in feet
	2	2
Soil, silty, brown	26	28
Clay, silty, red to brown	33	61
Sand and gravel; arkosic, orange	7	68
Clay, tan to white	21	89
Sand and gravel; clayey, orange	11	100
Shale, gray-green		

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-239-C.—Drilled November 11, 1989.
Altitude of land surface, 1,512.2 feet.

	Thickness, in feet	Depth, in feet
Soil, sandy, gray	2	2
Silt, clayey, gray	9	11
Sand, arkosic, tan to orange	39	50
Clay, tan	2	52
Sand and gravel; tan to orange	19	71
Shale, silty, red to maroon	4	75

EB-240-C.—Drilled November 14, 1989.
Altitude of land surface, 1,485.7 feet.

	Thickness, in feet	Depth, in feet
Soil, sandy, brown	5	5
Sand, fine-grained, tan to orange	4	9
Sand and gravel; arkosic, tan to orange	51	60
Clay, sandy, silty, tan	10	70
Sand, fine- to medium-grained, tan	11	81
Clay, gray	3	84
Sand, fine-grained, tan	11	95
Clay, gray	35	130
Sand and gravel; tan	30	160
Sand, fine-grained, clayey, lignitic, tan	9	169
Clay, lignitic, tan to gray	3	172
Sand, fine-grained, clayey	8	180
Sand, fine- to medium-grained, tan; with calcium carbonate cemented layers at 192 to 193 and 260 to 261 feet.	112	292
Sand, fine-grained, tan; with some ironstone gravel	21	313
Shale, gray, maroon	27	340

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-241-C.—Drilled January 13, 1990.
Altitude of land surface, 1,444.4 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, sandy, brown	5	5
Sand and gravel; arkosic, orange	35	40
Clay, sandy, tan to brown	14	54
Sand and gravel; clayey, tan to orange	17	71
Clay, sandy, silty, tan to gray	14	85
Sand and gravel; arkosic, tan to orange	30	115
Clay, tan to gray	8	123
Sand, arkosic, orange	34	157
Silt, clayey, black	13	170
Sand, fine-grained, tan to gray	60	230
Clay, sandy, tan	10	240
Sand, tan	36	276
Sandstone, white	3	279
Sand and gravel; orange	11	290
Sand, clayey, tan	5	295
Sand and gravel; arkosic, orange	26	321
Shale, gray	4	325

EB-242-C.—Drilled January 4, 1990.
Altitude of land surface, 1,462.6 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, brown	5	5
Sand, coarse-grained, and gravel; arkosic, tan	47	52
Clay, silty, tan to yellow	11	63
Silt, clayey, tan to yellow	19	82
Sand and gravel; arkosic, tan to orange	26	108
Sand, coarse-grained, and gravel; arkosic	12	120
Silt and sand, fine-grained, micaceous, tan to gray	10	130
Clay, gray, lignitic	29	159
Sand, fine- to medium-grained, tan	13	172
Silt, clayey, gray	6	178
Sand and gravel, arkosic, tan to orange	39	217
Sandstone, white, calcareous cement	2	219
Clay, silty, tan	10	229
Sand and gravel; arkosic, tan to orange	41	270
Clay, silty, sandy, tan	10	280
Sandstone, calcareous cement	1	281
Silt and sand, fine, tan	19	300
Sand and gravel, some ironstone	6	306
Shale, gray	19	325

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-243-C.—Drilled January 24, 1990.
Altitude of land surface, 1,434.2 feet.

	Thickness, in feet	Depth, in feet
Sand, silty, brown	7	7
Sand and gravel; arkosic, orange	14	21
Clay, sandy, yellow to gray	5	26
Sand, fine- to medium-grained, tan	6	32
Clay, silty, sandy, tan to yellow	6	38
Sand and gravel	13	51
Clay, silty, tan	10	61
Sand and gravel; arkosic, tan to orange	57	118
Clay, silty, tan to gray	7	125
Sand, fine-grained, tan	68	193
Clay, tan to brown	9	202
Sand and gravel; arkosic, tan to orange	25	227
Shale, gray to maroon	13	240

EB-244-C.—Drilled February 1, 1990.
Altitude of land surface, 1,393.8 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, sandy, brown	4	4
Sand, fine- to medium-grained, arkosic, tan to orange	37	41
Clay, tan to yellow	22	63
Sand and gravel; arkosic, tan to orange	57	120
Shale, gray	5	125

EB-245-C.—Drilled February 6, 1990.
Altitude of land surface, 1,378.5 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, clayey, brown	3	3
Clay, tan to gray	4	7
Clay, silty, sandy, tan to gray	13	20
Sand, fine- to medium-grained, tan to orange	15	35
Sand and gravel; arkosic, orange, with clay layers at 50 to 52, 60–61, 80–81, and 102–103 feet.	119	154
Clay, sandy, gray to tan	7	161
Sand and gravel; arkosic, tan to orange	34	195
Sand, arkosic, tan to white	31	226
Dolomite or limestone, gray	1	227
Shale, gray	13	240

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-246-C.—Drilled February 9, 1990.
Altitude of land surface, 1,347.5 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, sandy, brown	6	6
Sand and gravel; arkosic, tan to orange	21	27
Clay, tan to yellow	8	35
Sand, fine-grained, tan	10	45
Clay, sandy, brown to tan	25	70
Sand, fine-grained, and silt; tan	16	86
Sand, fine-grained, tan	12	98
Clay, tan to gray	28	126
Sand, fine-grained, and silt; tan	27	153
Sand, white	1	154
Shale, gray	11	165

EB-247-C.—Drilled January 29, 1990.
Altitude of land surface, 1,431.9 feet.

	Thickness, in feet	Depth, in feet
Soil, silty, brown	3	3
Clay, gray	5	8
Sand, arkosic, tan to orange	78	86
Clay, sandy, tan	5	91
Sand, clayey, tan	10	101
Clay, sandy, tan	9	110
Sand, with caliche	17	127
Shale	13	140

EB-248-C.—Drilled November 6, 1989.
Altitude of land surface, 1,522.3 feet.

	Thickness, in feet	Depth, in feet
Soil, clayey, gray	1	1
Clay, sandy, gray to tan	10	11
Clay, sandy, brown to tan	6	17
Sand, fine- to coarse-grained, and gravel	38	55
Clay, sandy, gray	4	59
Sand, fine- to medium-grained, tan	3	62
Clay, tan	13	75
Sand, silty, tan	30	105
Sand, fine- to coarse-grained, tan, with red chert pieces	15	120
Clay, sandy, tan	9	129
Sand, fine- to coarse-grained, and gravel	6	135
Shale, green to gray to red	10	145

Table 7. Lithologic logs of deep wells at well sites EB-201 through EB-249—Continued

EB-249-C.—Drilled December 2, 1989.
Altitude of land surface, 1,476.3 feet.

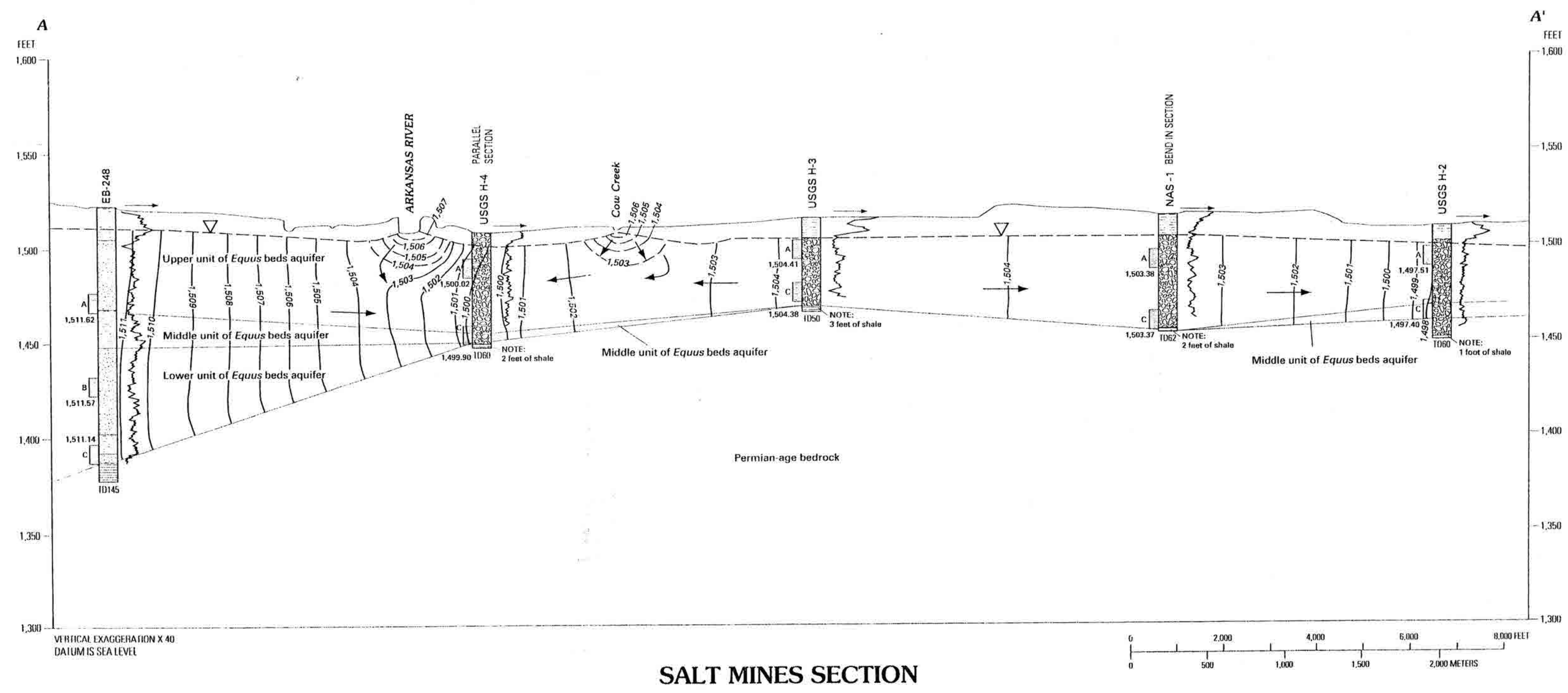
	Thickness, in feet	Depth, in feet
Sand, fine- to coarse-grained, gray to brown	5	5
Sand and gravel, silty, tan to brown	5	10
Clay, sandy, gray, tan	8	18
Sand and gravel; tan	9	27
Clay, tan	15	42
Sand, fine-grained, white	4	46
Clay, sandy, tan to gray	30	76
Sand, fine- to coarse-grained, tan	26	102
Clay, sandy, gray	23	125
Sand, fine- to medium-grained, orange	54	179
Clay, black	10	189
Sand, clayey, tan	12	201
Sand, fine- to medium-grained, tan	11	212
Clay, sandy, tan	7	219
Sand, fine- to coarse-grained, tan	21	240
Clay, sandy, white	6	246
Sand, fine- to coarse-grained, white to tan	69	315
Sandstone, white	1	316
Shale, gray	9	325

EXHIBIT H

U.S. DEPARTMENT OF THE INTERIOR
U.S. GEOLOGICAL SURVEY

Prepared in cooperation with the
KANSAS WATER OFFICE, the *EQUUS* BEDS GROUNDWATER MANAGEMENT
DISTRICT NO. 2, the BUREAU OF RECLAMATION, and the U.S. DEPARTMENT OF THE INTERIOR

WATER-RESOURCES INVESTIGATIONS REPORT 95-4191
Hydrogeologic sections *A–A'* through *D–D'*—PLATE 1
Myers, N.C., and others, 1996, Hydrologic and chemical interaction of the Arkansas River and the *Equus* beds aquifer between Hutchinson and Wichita, south-central Kansas



SALT MINES SECTION

EXPLANATION

LITHOLOGY

- Soil
- Gravel
- Sand
- Silt
- Clay
- Shale

Approximate water table

Geologic contact—Dashed where approximately located, contact location based primarily on gamma log correlations

1,415 **Line of equipotential**—Shows altitude at which water level would have stood in tightly cased well, June 1990. Dashed where approximately located. Interval 1 and 5 feet. Datum is sea level

Gamma-ray logs—Arrow indicates increasing radioactivity

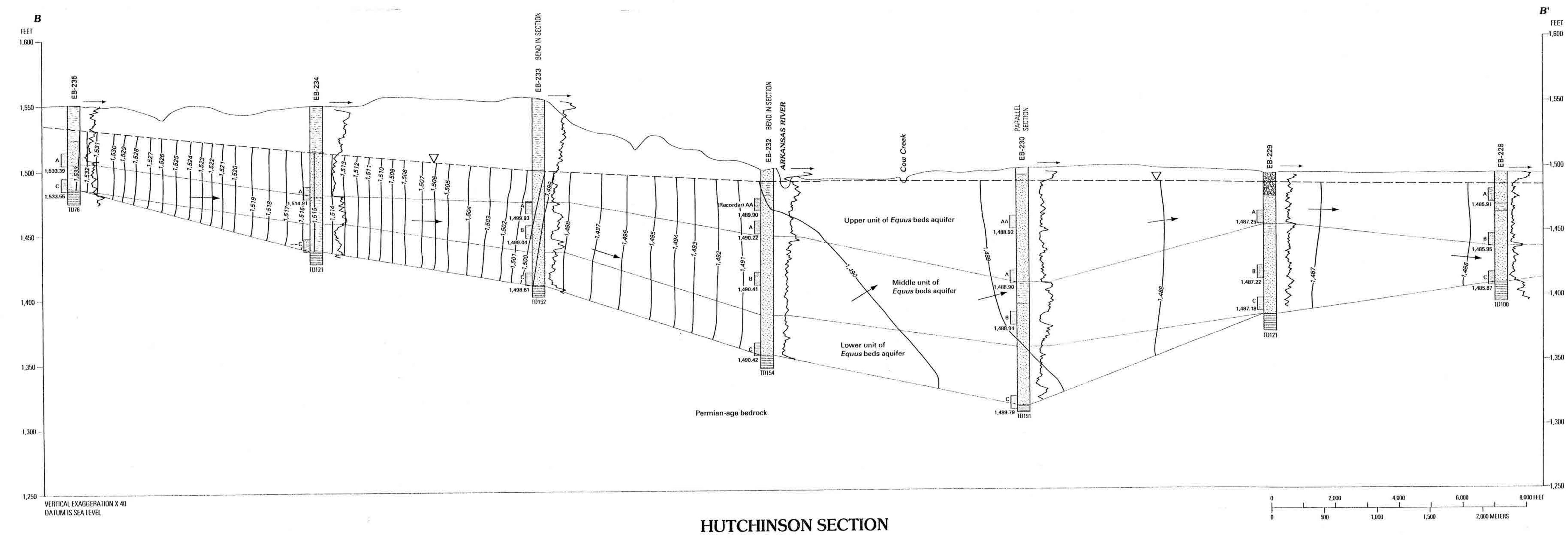
A 1,456.15 **Well-screen location, letter designation of observation well, and June 1990 water altitude, in feet above sea level, if available**

EB-225 **Observation-well site number**

TD182 **Total depth (TD) of well below land surface, in feet**

Direction of ground-water flow

NOTE: Lithologies shown for wells have been generalized from the lithologic descriptions (table 7), to show only the dominant lithology. There is no dividing line in the lithologic column unless there is a change in the dominant lithology. Also, layers less than 5 feet thick were combined with adjacent layers.



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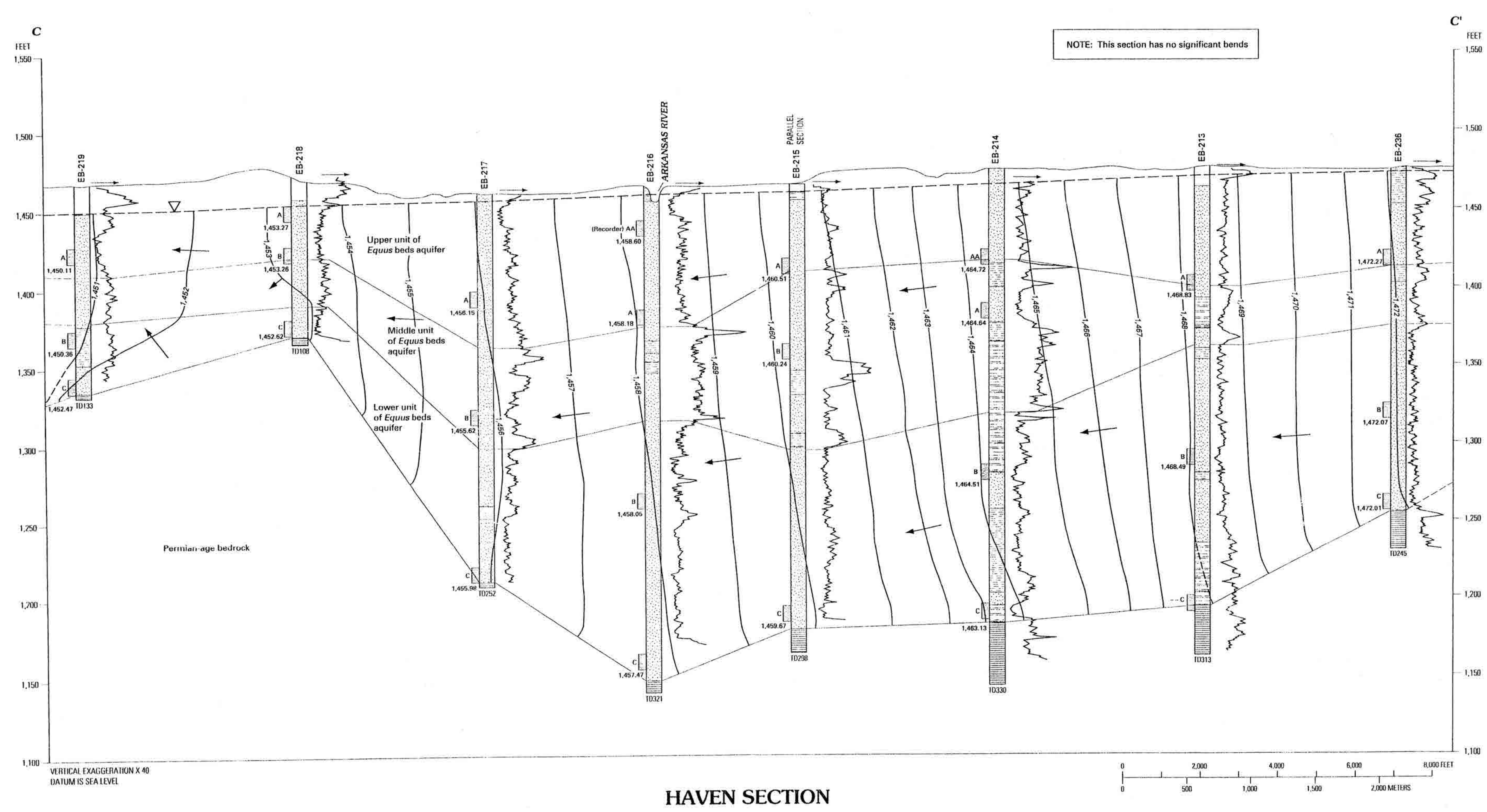

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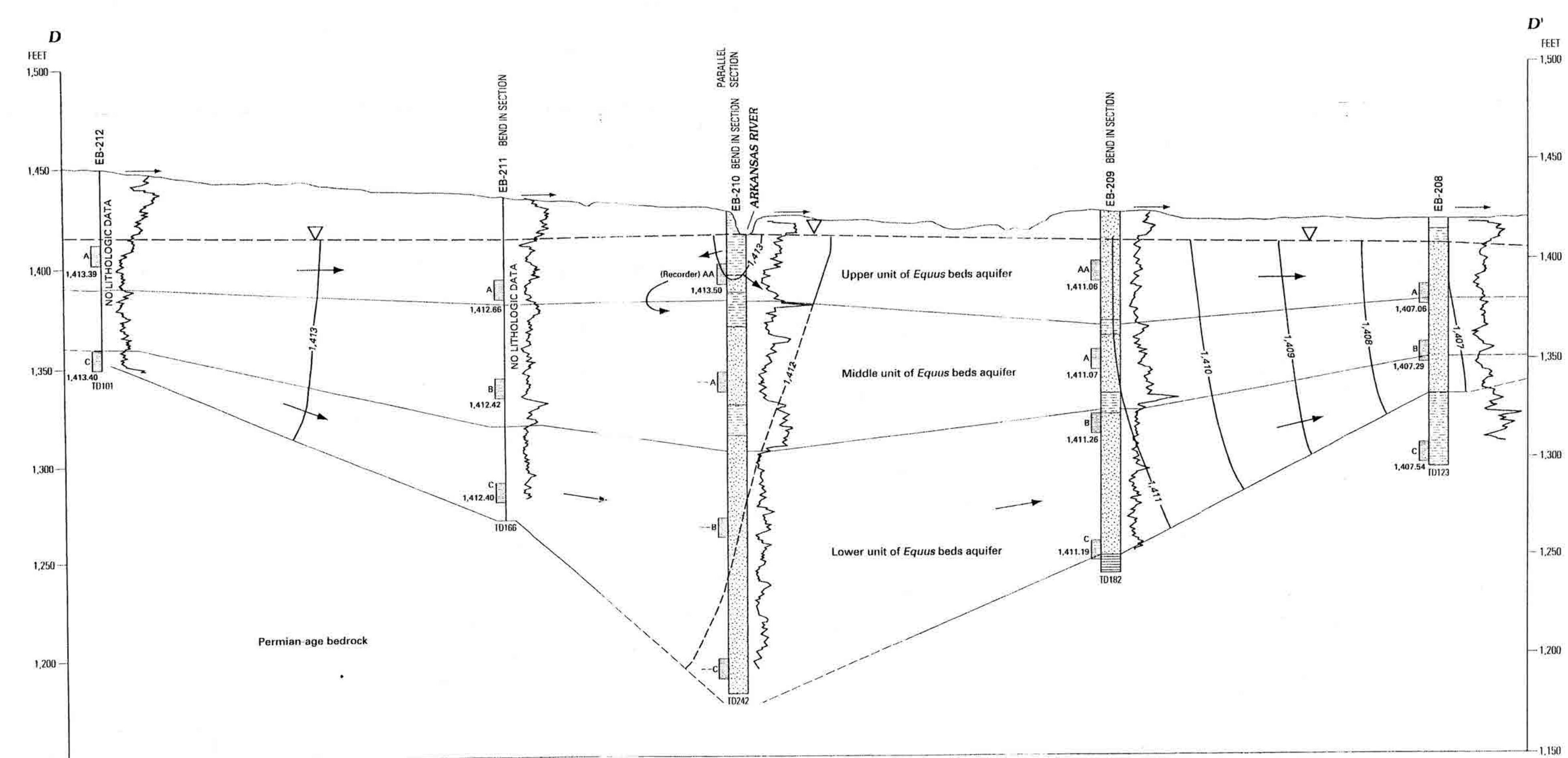
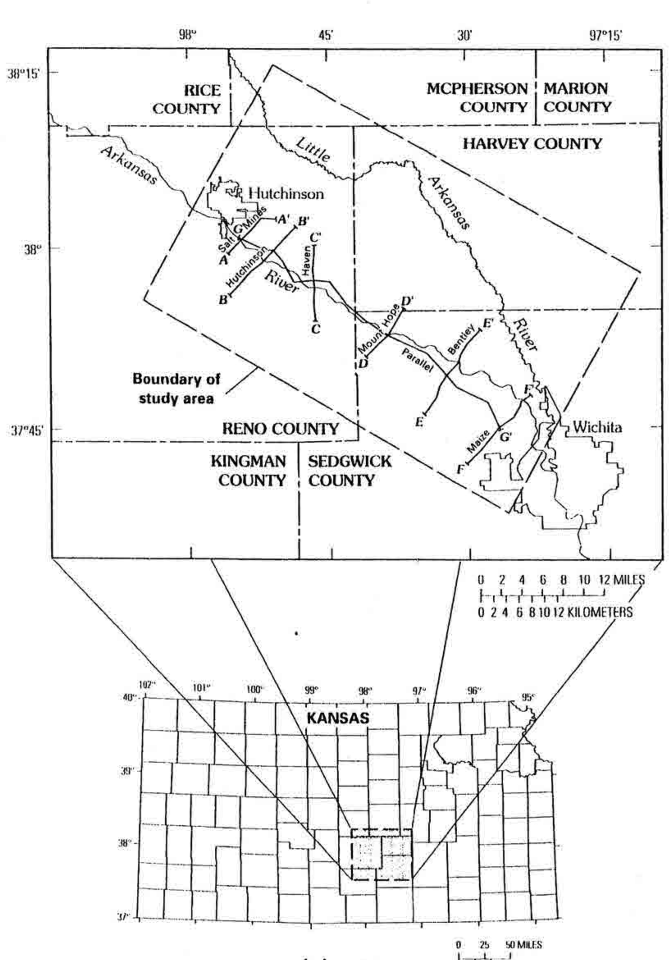


MOUNT HOPE SECTION



Index map

HYDROGEOLOGIC SECTIONS *A–A'* THROUGH *D–D'* IN VICINITY OF THE ARKANSAS RIVER BETWEEN HUTCHINSON AND WICHITA, SOUTH-CENTRAL KANSAS

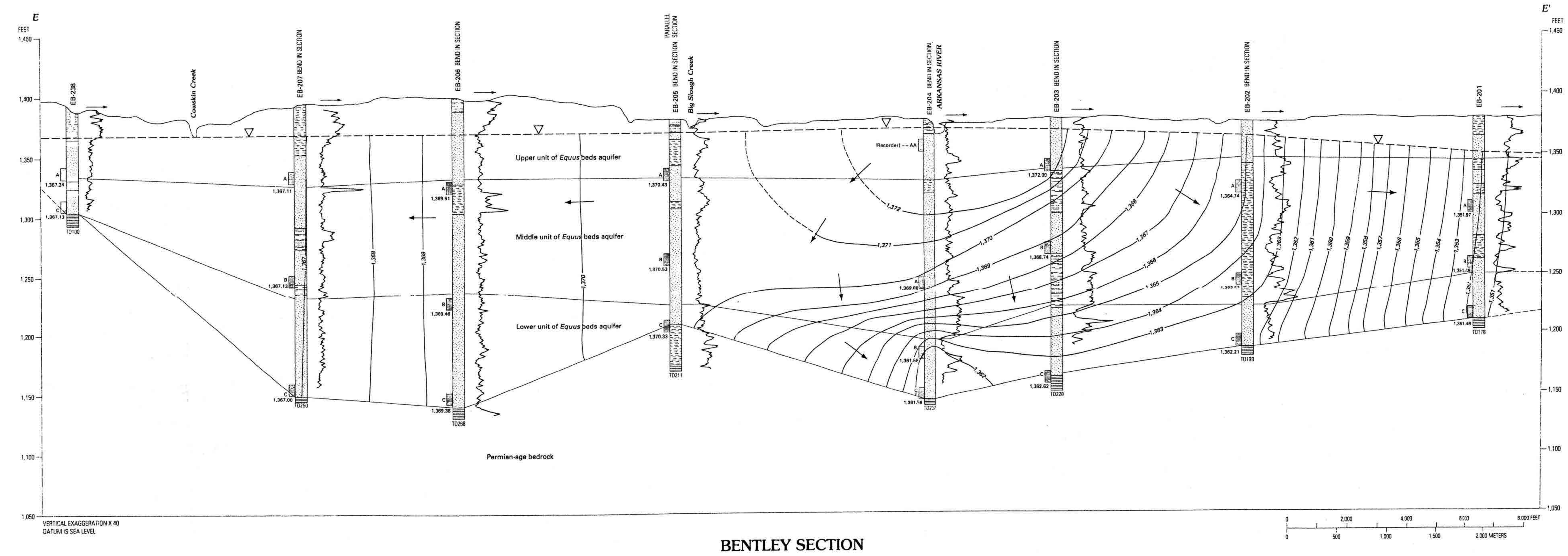
By
N.C. Myers, G.D. Hargadine, and J.B. Gillespie
1996

EXHIBIT H

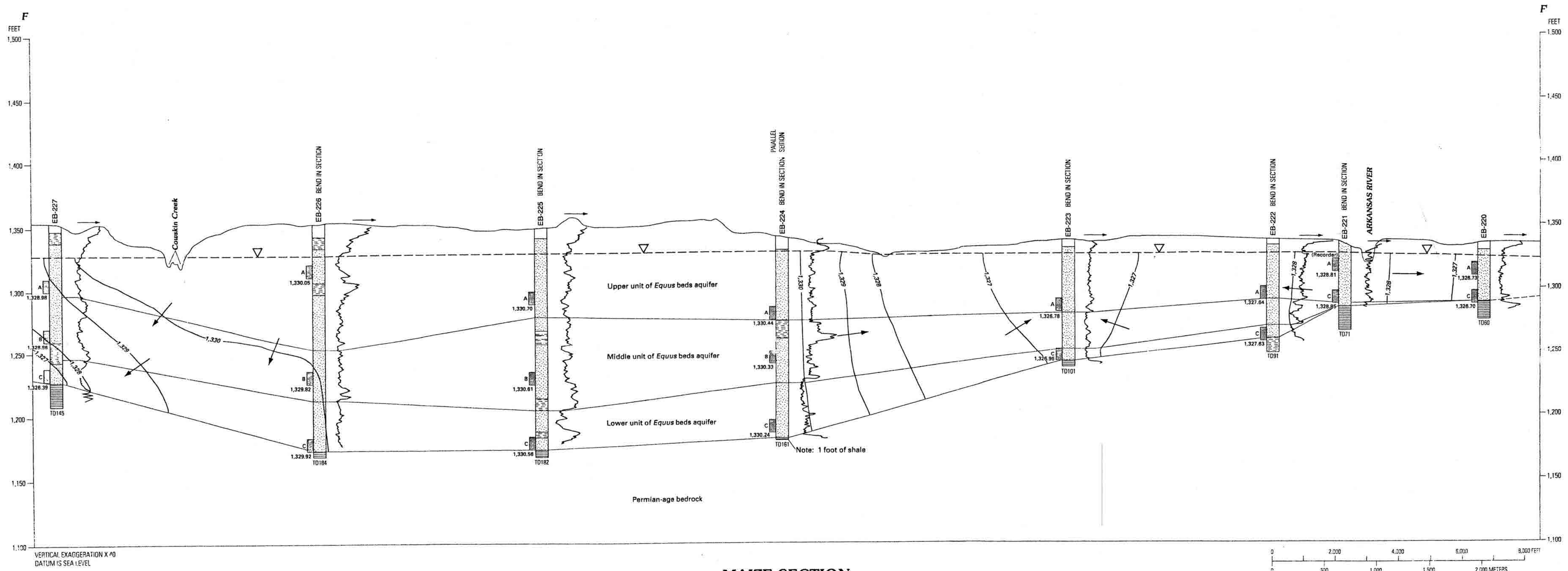
U.S. DEPARTMENT OF THE INTERIOR
U.S. GEOLOGICAL SURVEY

Prepared in cooperation with the
KANSAS WATER OFFICE, the *EQUUS* BEDS GROUNDWATER MANAGEMENT
DISTRICT NO. 2, the BUREAU OF RECLAMATION, and the U.S. DEPARTMENT OF THE INTERIOR

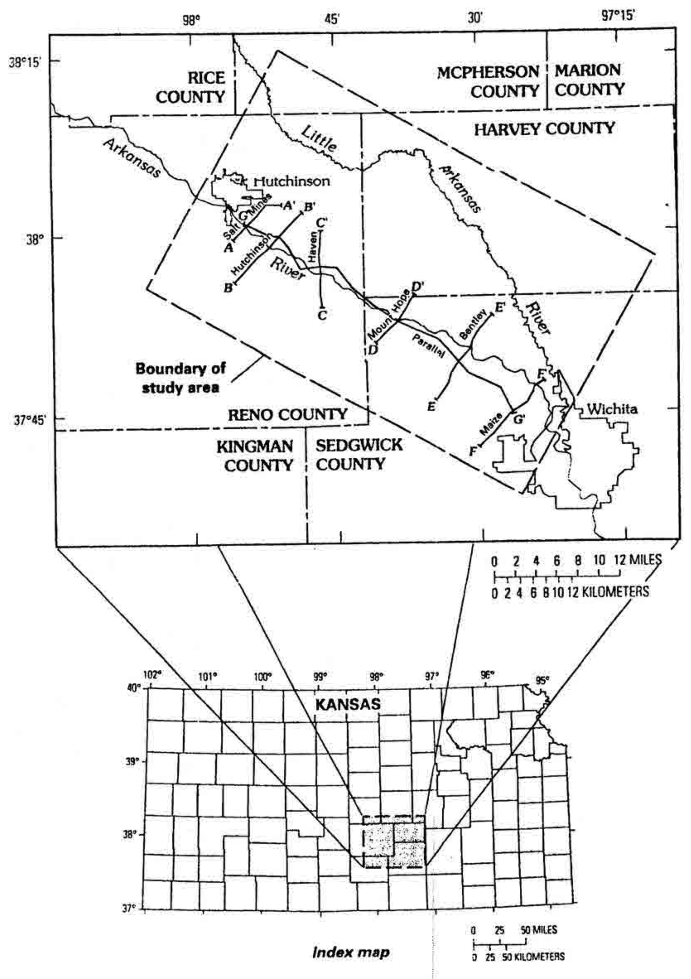
WATER-RESOURCES INVESTIGATIONS REPORT 95–4191
Hydrogeologic sections *E–E'* through *G–G'*—PLATE 2
Myers, N.C., and others, 1996, Hydrologic and chemical interaction of the Arkansas River and the *Equus* beds aquifer between Hutchinson and Wichita, south-central Kansas

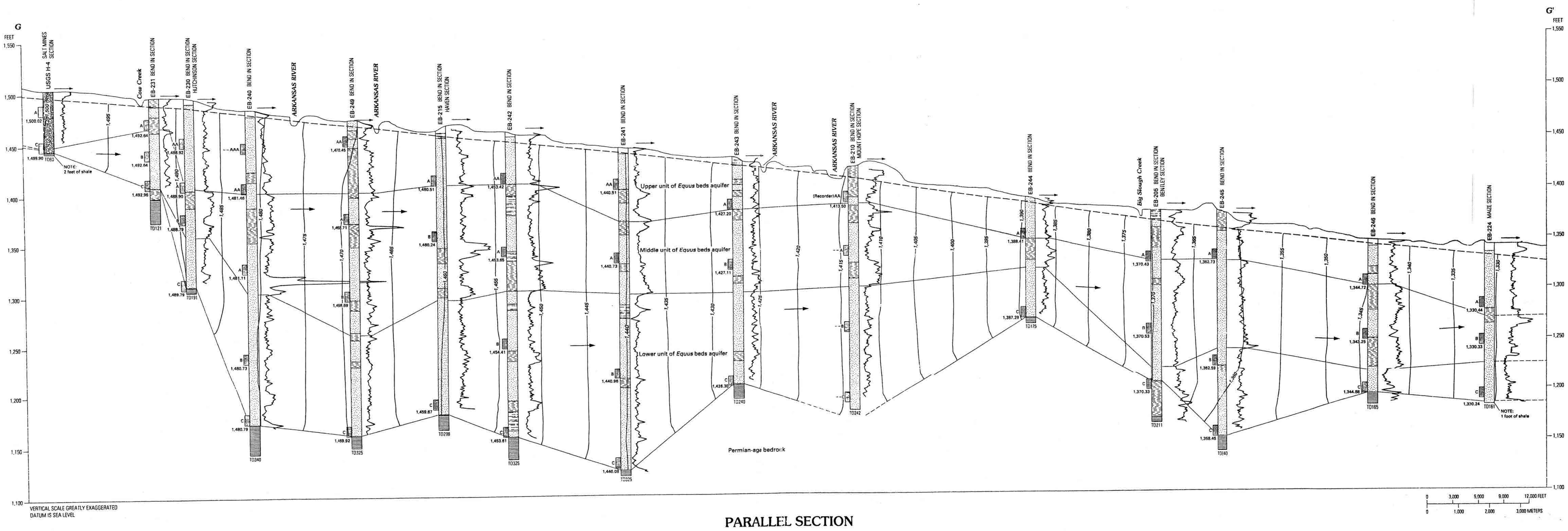


BENTLEY SECTION



MAIZE SECTION





PARALLEL SECTION

EXPLANATION

LITHOLOGY

- Soil
- Gravel
- Sand
- Silt
- Clay
- Shale

Approximate water table

Geologic contact—Dashed where approximately located, contact location based primarily on gamma log correlations

1,415 **Line of equipotential**—Shows altitude at which water level would have stood in tightly cased well, June 1990. Dashed where approximately located. Interval 1 and 5 feet. Datum is sea level

Gamma-ray logs—Arrow indicates increasing radioactivity

A 1,456.15 **Well-screen location, letter designation of observation well, and June 1990 water altitude, in feet above sea level, if available**

EB-225 **Observation-well site number**

TD182 **Total depth (TD) of well below land surface, in feet**

Direction of ground-water flow

NOTE: Lithologies shown for wells have been generalized from the lithologic descriptions (table 7), to show only the dominant lithology. There is no dividing line in the lithologic column unless there is a change in the dominant lithology. Also, layers less than 5 feet thick were combined with adjacent layers.

HYDROGEOLOGIC SECTIONS *E–E'* THROUGH *G–G'* IN VICINITY OF THE ARKANSAS RIVER BETWEEN HUTCHINSON AND WICHITA, SOUTH-CENTRAL KANSAS

By
N.C. Myers, G.D. Hargadine, and J.B. Gillespie
1996

EXHIBIT I

Burrton IGUCA Remediation Study

Prepared for

Kansas Corporation Commission

1997

97-183-4

002008

Burns & McDonnell

Burrton IGUCA Remediation Study

Prepared for

Kansas Corporation Commission

1997

97-183-4

Burns & McDonnell

February 18, 1998

Kansas Corporation Commission
130 S. Market Room 2078
Wichita, Kansas 67202-3802

KCCBURTN
Burrton IGUCA Remediation Study
Project 97-183-4

Ladies and Gentlemen:

Presented herewith is the report titled "*Burrton IGUCA Remediation Study,*" in accordance with our contract for professional engineering services dated April 1, 1997. This engineering study evaluates potential high chloride remediation alternatives using groundwater and contaminant transport modeling and includes cost estimates for various alternatives.

Based on study results, large conventional remediation schemes have relatively small impacts on the mass of salt in the Burrton brine plume. Large volume pumping for plume remediation also has the negative impact of inducing flow of high chloride water from the Arkansas River into the area.

Study findings emphasize the long-term nature of remediation measures and the need for careful coordination of remediation, water supply, and management actions for the Equus Beds Aquifer. Recommendations are presented for immediate actions to enhance assessment of the suitability of the water for beneficial use and for a pilot plant remediation project which are needed prior to commitment to large-scale remediation efforts.

The assistance provided by the staff of the Kansas Corporation Commission during the course of this study is greatly appreciated. The project team remains ready to discuss the details of this report at your convenience.

Sincerely,

Joel A. Cerwick, P.E.
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KANSAS CORPORATION COMMISSION
BURRTON IGUCA REMEDIATION STUDY
PROJECT NO. 97-193-4

INDEX AND CERTIFICATION PAGE

REPORT INDEX

PART NO.	PART TITLE	NUMBER OF PAGES
S	SUMMARY	9
IN	INTRODUCTION	2
I	SITE DESCRIPTION	14
II	MODELING	16
III	REMEDIATION TECHNOLOGIES	12
IV	REMEDIATION ALTERNATIVES	7
V	CONCLUSIONS AND RECOMMENDATIONS	7

CERTIFICATION(S)

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KANSAS
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2/18/98

Table of Contents

TABLE OF CONTENTS

SUMMARY S-1

INTRODUCTION IN-1
A. PURPOSE IN-1
B. SCOPE IN-1

PART I - SITE DESCRIPTION I-1
A. GENERAL I-1
B. STUDY AREA I-1
C. SOURCES OF ELEVATED CHLORIDE CONCENTRATIONS I-2
D. EXTENT OF CONTAMINATION I-4
E. GEOLOGY I-6
1. Physiography I-6
2. Topography and Drainage I-7
3. Bedrock I-7
4. Unconsolidated Deposits I-8
5. Soils I-9
6. Structure I-9
F. HYDROGEOLOGY I-11
1. Recharge I-12
2. Discharge I-12
3. Aquifer Parameters I-13
4. Surface Hydrology I-14

PART II - MODELING II-1
A. GENERAL II-1
B. GROUNDWATER MODEL II-2
1. Conceptual Model II-2
2. Regional Model II-4
3. Subregional Model II-6
C. CONTAMINANT TRANSPORT MODEL II-7
1. Subregional Model II-7
2. Boundary Conditions and Initial Concentrations II-7
D. MODELING SCENARIOS II-8
E. MANAGEMENT SIMULATION RESULTS II-9
1. No Action (1a) II-9
2. Plume Interceptor Wells (2a, 2b, and 2c) II-9
3. Plume Control (3a, 3b and 3c) II-11
4. Plume Control with Recharge Basins (4a) II-13
5. Plume Control Impacts of the Equus Beds Groundwater Recharge Project (5a, 5b and 5c) II-13

6. Equus Beds Recharge with Plume Control (6a, 6b, 6c and 6d) II-14
7. Pilot Study (7a) II-15
F. PARTICLE TRACKING ANALYSIS II-16

PART III - REMEDIATION TECHNOLOGIES III-1
A. GENERAL III-1
B. MEMBRANE TREATMENT PROCESSES III-1
1. Reverse Osmosis III-3
a. Waste Minimization (High Pressure Membranes) III-4
b. Reverse Osmosis Plant Pilot Study III-4
C. BENEFICIAL REUSE III-5
1. Public Water District III-5
2. City of Wichita III-5
D. DISPOSAL TECHNIQUES III-8
1. Deep Well Injection III-9
2. Discharge to Surface Water III-10
3. Evaporation Ponds III-12

PART IV - REMEDIATION ALTERNATIVES IV-1
A. GENERAL IV-1
B. CAPITAL COSTS FOR ALTERNATIVE COMPONENTS IV-3
1. Extraction Wells IV-3
2. Gradient Control Wells IV-3
3. Class II Disposal Wells IV-4
4. Reverse Osmosis Plant IV-4
5. Recharge Basins IV-5
6. Pipelines IV-6
C. REMEDIAL ALTERNATIVES COST ESTIMATES IV-6

PART V - CONCLUSIONS AND RECOMMENDATIONS V-1
A. CONCLUSIONS V-1
B. RECOMMENDATIONS V-5

REFERENCES REF-1

LIST OF FIGURES

Figure No.	Title	Follows Page No.
I-1	Study Area	I-1
I-2	Chloride Concentration Map for 1948 Less Than 50 Feet Deep	I-4
I-3	Chloride Concentration Map for 1948 Greater Than 50 Feet Deep	I-4
I-4	Chloride Concentration Map for 1982 Less Than 50 Feet Deep	I-4
I-5	Chloride Concentration Map for 1982 Greater Than 50 Feet Deep	I-4
I-6	1996 Chloride Concentrations in Level A	I-4
I-7	1996 Chloride Concentrations in Level B	I-4
I-8	1996 Chloride Concentrations in Level C	I-4
I-9	Chloride Concentrations in Wichita City Well No. 45	I-6
I-10	Altitude of Bedrock Surface	I-8
I-11	Study Area Showing Geologic Cross-Sections and Well Locations	I-11
I-12	Geologic Cross-Sections	I-11
I-13	1996 Water Table	I-12
II-1	Subregional Conceptual Model	II-3
II-2	Subregional Model Boundary Conditions	II-6
II-3	Level A Chloride Concentrations - 1 No Action for 50 Years	II-9
II-4	Level B Chloride Concentrations - 1 No Action for 50 Years	II-9
II-5	Level C Chloride Concentrations - 1 No Action for 50 Years	II-9
II-6	Location of USBR Interceptor Wells	II-9
II-7	Level A Chloride Concentrations - 2c Plume Interception 4,000 gpm for 50 Years	II-9
II-8	Level B Chloride Concentrations -2c Plume Interception 4,000 gpm for 50 Years	II-9
II-9	Level C Chloride Concentrations -2c Plume Interception 4,000 gpm for 50 Years	II-9
II-10	USBR Interceptor Wells - Chloride Concentrations for No. 5B	II-10
II-11	USBR Interceptor Wells - Chloride Concentrations for No. 5C	II-10
II-12	3c Plume Control Well Locations	II-11
II-13	Level A Chloride Concentrations -3c Plume Control 4,000 gpm for 50 Years	II-12
II-14	Level B Chloride Concentrations -3c Plume Control 4,000 gpm for 50 Years	II-12
II-15	Level C Chloride Concentrations -3c Plume Control 4,000 gpm for 50 Years	II-12
II-16	Chloride Concentrations - Gradient Control Well 3 - Level B	II-12
II-17	Chloride Concentrations - Gradient Control Well 3 - Level C	II-12
II-18	4a Plume Control Showing Locations of Recharge Basins	II-13
II-19	Level A Chloride Concentrations -4a Plume Control with Recharge Basins 4,000 gpm for 50 Years	II-13

Figure No.	Title	Follows Page No.
II-20	Level B Chloride Concentrations -4a Plume Control with Recharge Basins 4,000 gpm for 50 Years	II-13
II-21	Level C Chloride Concentrations -4a Plume Control with Recharge Basins 4,000 gpm for 50 Years	II-13
II-22	Level A Chloride Concentrations - 5c Equus Beds Recharge Project	II-14
II-23	Level B Chloride Concentrations - 5c Equus Beds Recharge Project	II-14
II-24	Level C Chloride Concentrations - 5c Equus Beds Recharge Project	II-14
II-25	Level A Chloride Concentrations - 6c Equus Beds Recharge Project with Plume Control, 4,000 gpm for 50 Years	II-14
II-26	Level B Chloride Concentrations -6c Equus Beds Recharge Project with Plume Control, 4,000 gpm for 50 Years	II-14
II-27	Level C Chloride Concentrations -6c Equus Beds Recharge Project with Plume Control, 4,000 gpm for 50 Years	II-14
II-28	Level A Chloride Concentrations - 6d Equus Beds Recharge Project with Plume Control - 8,000 gpm for 50 Years	II-15
II-29	Level B Chloride Concentrations -6d Equus Beds Recharge Project with Plume Control - 8,000 gpm for 50 Years	II-15
II-30	Level C Chloride Concentrations -6d Equus Beds Recharge Project with Plume Control - 4,000 gpm for 50 Years	II-15
II-31	50 Year Capture Area for Typical Gradient Control Well, Extraction Well, and Municipal Well	II-16
II-32	Migration of Water Particles from Level A to Level C Wichita Well No. 41	II-16
III-1	RO Schematic	III-3
III-2	Class II Disposal Well Schematic	III-9
IV-1	Alternative 1	IV-1
IV-2	Alternative 2	IV-1
IV-3	Alternative 3	IV-1
IV-4	Alternative 4	IV-2
IV-5	Alternative 5	IV-2
IV-6	Pilot Study	IV-2

LIST OF TABLES

Table No.	Title	Follows Page No.
II-1	Management Simulations	II-8
II-2	Total Land Area Above High Chloride Levels in Groundwater	II-9
II-3	Summary of Model Salt Budget (Tons of Salt)	II-10
III-1	RO Feed Water Characteristics	III-3
III-2	Potential Usable Flow (MGD) from Burrton Plume Control Wells	III-7
IV-1	Capital Costs for Extraction Wells	IV-3
IV-2	Capital Costs for Gradient Control Wells	IV-3
IV-3	Capital Costs for Class II Disposal Wells	IV-4
IV-4	Capital Costs for Reverse Osmosis Plant	IV-4
IV-5	Capital Costs for Recharge Basins	IV-6
IV-6	Alternative 1 Costs	IV-6
IV-7	Alternative 2 Costs	IV-6
IV-8	Alternative 3 Costs	IV-6
IV-9	Alternative 4 Costs	IV-6
IV-10	Alternative 5 Costs	IV-6
IV-11	Alternative 6 Costs	IV-6

Summary

SUMMARY

The Kansas Corporation Commission (KCC) is currently considering alternatives involving the control, treatment, beneficial use, and disposal of high chloride groundwater in the vicinity of the Burrton Intensive Groundwater Use Control Area (IGUCA). Past oil field operations near Burrton, Kansas have impacted the Equus Beds Aquifer in this region, resulting in a plume of elevated chloride concentration levels greater than 2,000 mg/L. This groundwater plume of high chlorides is slowly expanding and migrating southeast within the Equus Beds Aquifer, threatening downgradient irrigation and municipal supply wells. Current data indicates that chloride levels in several of the Wichita wells are increasing. Modeling indicates that portions of the plume are likely to reach the Wichita municipal well field in less than 50 years.

The primary objectives of this evaluation are to establish potential remediation alternatives using groundwater and contaminant transport modeling and develop cost estimates for the alternatives. Work includes refining the hydrogeology of the study area and reviewing historical chemical data to create a subregional model for predicting chloride concentrations and the effectiveness of the remediation scenarios.

SITE DESCRIPTION

The study area established for this evaluation is depicted by the subregional model boundary shown in Figure I-1 and is established to address the majority of chloride impacted groundwater originating from the former Burrton oil field. The model of the study area is derived from a larger regional groundwater model developed by the U.S. Geological Survey (USGS) and refined by the U.S. Bureau of Reclamation (USBR).

Both natural and man-made chloride sources have impacted the groundwater within the study area. Two natural sources include recharge from the Arkansas River and saltwater intrusion from the Wellington Formation. Three man-caused sources of chlorides occurring throughout the region include brine of evaporation pans from salt mining activities, brine from oil field

operations, and migration of saltwater from the Wellington Formation through faulty well casings. The primary source of the groundwater salinity in the project area originates from the Burrton and Hollow-Nikkel oil field activities.

MODELING

Several modeling studies have been conducted by the USGS, USBR, and others for the Equus Beds Aquifer due to its importance as a source of water for municipal, agricultural, and industrial use. The majority of modeling parameters used in this evaluation were derived from those utilized in the previous models. A regional model, shown in Figure S-1, was refined from a recent USBR model. The regional model was updated to include the 1990 to 1994 chloride concentrations, current river levels, well pumpage and recharge parameters, based on 1990 through 1994 data. The study area subregional model was developed with finer grid spacing than earlier models, to contribute to a more detailed evaluation of chloride movement. Three hydrostratigraphic units were delineated for the regional and subregional models as follows:

- Level A -- upper sand and gravel unit generally less than 50 feet below land surface (BLS)
- Level B -- middle fine grained sand, silt and clay unit generally greater than 50 feet and less than 170 feet BLS
- Level C -- lower coarse grained unit generally greater than 170 feet BLS and above bedrock.

Numerous modeling scenarios were established to determine the impact of control well pumpages and recharge on plume migration control and remediation. All scenarios were modeled for a 50 year time period beginning with the 1996 chloride distribution. A base, or no action, scenario assumes that no action was taken to remediate or control the chloride plume for the entire 50 year time period and current pumping stresses are constant. The result of a no action scenario was lateral spreading and migration of the chloride plume in the study area generally to the east, towards the Wichita well field. The plume tended to migrate to the lower levels of the aquifer, due to recharge of precipitation in the upper zone and pumping from the lower zones. The peak chloride concentrations decreased as the plume was diluted by precipitation and mixing with the surrounding groundwater. Results of other plume management scenarios were compared with the results of the base or no action case.

Several types of plume control wells or recharge features were included in the various modeled scenarios. These include:

- Interceptor and extraction wells - relatively low capacity wells installed at locations of high chloride concentrations. Pumping rates varied from about 100 to 400 gpm.
- Gradient control wells - located upgradient of the plume to reduce water levels, lowering the gradient and slowing plume migration. The gradient control wells were located in areas of medium to low chloride concentrations so the water could be beneficially reused with little treatment. These wells were simulated at up to a 400 gpm pumping rate.
- Recharge basins - recharge of pumpage from the gradient control wells into the formation to raise water levels downgradient of the plume to reduce the plume migration rate.

Initial modeling scenarios indicated that very high pumping and recharge rates would be required for complete plume management. Several scenarios were evaluated with plume control pumping rates of 4,000 gpm which resulted in the greatest impact on the plume. These scenarios can be summarized as follows:

- **Interceptor wells, 4,000 gpm:** 20 wells in the middle and lower layers of the aquifer pumping at a total rate of 4,000 gpm. The locations of these wells were similar to those located in the 1993 USBR model (Pruitt model, 1993). This was the highest pumping rate evaluated in the USBR study.

- **Plume control wells, 4,000 gpm:** 6 wells (gradient control wells) located upgradient of the plume and screened in Levels B and/or C and 17 wells (extraction wells) in the high concentration areas screened in Levels A and/or B pumping at a total rate of 4,000 gpm.

- **Plume control wells with recharge basins:** gradient control and extraction wells described above with the addition of four one-half acre recharge basins receiving 75% of the pumpage from the extraction well network assuming 25% is lost during water treatment. The recharge basins were located downgradient of the plume.

- **Base scenario with recharge effects from the Equus Beds Recharge Project with active recharge into Level C:** No action for 50 years with water levels 10 feet higher in the vicinity of the Wichita wells due to the recharge effects from the Equus Beds Recharge Project with active injection into level C at 388 gpm through 22 recharge wells.

- **Plume control wells, 4000 gpm, with recharge effects from the Equus Beds Recharge Project with active recharge into Level C:** 6 wells upgradient of the plume screened in levels B and/or C and 17 wells in the high concentration areas screened in levels A and/or B pumping at a total rate of 4000 gpm for 50 years with water levels 10 feet higher in the vicinity of the Wichita wells due to the recharge effects from the Equus Beds Recharge Project with active injection into Level C at 388 gpm through 22 recharge wells.

- **Plume control wells, 8000 gpm, with recharge effects from the Equus Beds Recharge Project with active recharge into Level C:** 6 wells upgradient of the plume screened in levels B and/or C and 17 wells in the high concentration areas screened in levels A and/or B pumping at a total rate of 8000 gpm for 50 years with water levels 10 feet higher in the vicinity of the Wichita wells due to the recharge effects from the Equus Beds Recharge Project with active injection into Level C at 388 gpm through 22 recharge wells.

The interceptor well scenarios were duplicated for the subregional model from previous modeling by the USBR which simulated the effects of placing withdrawal wells intercepting the highest concentrations areas of the plume, east of Burrton. These remediation scenarios removed a small portion of the plume and slows, but did not prevent, the impact of the chloride plume on the Wichita Well Field.

The layout of the plume control wells combined the use of extraction wells located in the high concentration zones of the chloride plume and higher capacity wells upgradient of the plume. The low-capacity extraction wells are designed to remove groundwater from areas of highest chloride concentration within Levels A and B. The gradient control wells are designed to control the local groundwater flow gradient restricting the migration of the high chloride plume. Plume migration control allows greater chloride removal efficiencies. Like the interceptor wells scenario, the modeled plume control layout slows the impact of the chloride plume, without significant chloride removal.

The modeling simulation of plume control wells with four recharge basins located along the 200 mg/L chloride isocontour shows some additional control over the plume, however, like the previous scenarios, the beneficial impacts are limited. In all of the above cases, the control measures resulted in recovery of 8 to 10 percent of the salt believed to have entered the aquifer from oil field activities.

Final modeling scenarios were conducted to reflect the effects of a recharge project being undertaken by the City of Wichita. The project involves recharge of the Equus Beds Aquifer with above-base flow from the Little Arkansas River. A modeling scenario was set up with the assumption that the water levels rose five or ten feet along the eastern portion of the study area.

Additional no action scenarios were modeled with the effects from the Equus Beds Recharge Project to assess the natural movement of the plume. The results of these modeling scenarios indicated that rising groundwater levels caused by the Equus Beds Recharge project significantly effect migration of the chloride plume. In Levels B and C, the leading edge of the plume was pushed more to the north toward Wichita City Wells Numbers 3 and 4. The addition of the plume control well network with a total pumping rate of 4,000 gpm counteracted the movement of the plume to the north. The reduced flow gradient in the aquifer allowed this remediation scenario to have the greatest, yet still minimal, impact on the control and removal of the plume.

REMEDIAL ALTERNATIVES

Several remedial options were considered for the use, treatment or disposal of the recovered water. These options included:

- Beneficial use of the gradient control pumpage by blending with a municipal water supply
- Treatment of extraction well pumpage by reverse osmosis (RO) and subsequent use or recharge
- Beneficial use of a blend of RO permeate and gradient control pumpage as a municipal water supply
- Disposal of extraction well pumpage to deep geologic formations
- Disposal of gradient control pumpage to the Arkansas River
- Disposal of blended extraction and gradient control pumpage to the Arkansas River

The RO process is the preferred technology for the treatment of groundwater in the vicinity of the study area. The beneficial reuse of RO permeate and/or pumpage from gradient control wells is through blending for use as a municipal water supply. The proximity of the potential recipient of the treated and/or blended water affects total pipeline costs. The preferred interconnect for the water is at the well field of the City of Wichita, due to proximity, water demand and the ability of the relative large system to provide greater dilution and thus, less stringent influent requirements.

The preferred disposal technique is through Class II disposal wells, primarily due to cost effectiveness compared to other options. This disposal option is suitable for relatively small pumpage volumes of particularly high chloride concentrations, including RO concentrate and pumpage from extraction wells. The Arkansas River is the nearest significant surface water body to the control well network, and is thus evaluated to receive pumpage from remediation of the chloride plume.

Several remediation alternatives were developed to provide control and removal of the groundwater plume, treatment of the pumpage, if applicable, and disposal. Capital and annual costs were subsequently estimated and compared among the alternatives. The combination of control techniques were similar for each alternative, involving a network of gradient control wells to control migration of the plume, and a network of extraction wells to remove groundwater exhibiting high chloride concentrations. Each alternative included a well and pipeline preliminary plan. The locations of these wells correspond to plume control well layout used in the subregional model. The sizing of alternative components is based on a 4,000 gpm flow rate from the entire control wells system. The alternatives are summarized as follows:

- **Alternative 1:** Discharge from the extraction well network is continuously directed to four Class II disposal wells. Effluent from the gradient control wells is pumped to a connection at the Wichita well field and blended with the Wichita water supply, as appropriate (Figure IV-1).

- **Alternative 2:** Discharge from the extraction well network is continuously directed to four Class II disposal wells. Gradient control well discharge is continuously directed to an outfall along the Arkansas River (Figure IV-2).

- **Alternative 3:** Discharge from the extraction and gradient control networks is discharged at an outfall along the Arkansas River (Figure IV-3).

- **Alternative 4:** Discharge from the extraction well network is piped to an RO treatment plant. RO concentrate is pumped to one Class II disposal well for disposal. RO permeate and gradient control pumpage is blended and discharged to a connection with the Wichita Well Field (Figure IV).

- **Alternative 5:** Discharge from the extraction well network is piped to an RO treatment plant. RO concentrate is pumped to one Class II disposal well. RO permeate and gradient control pumpage is blended and discharged to four one-half acre recharge basins (Figure IV -5).

- **Pilot Study:** This alternative is a pilot-scale installation of two 250 gpm extraction wells with disposal of pumpage to one Class II disposal well (Figure IV-6).

Estimates of capital costs, operation, maintenance, energy and equipment replacement costs are itemized for each of the remedial alternatives in Tables IV-6 through IV-11. Annual costs, involving energy and labor rates, were obtained from sources local to the study area. Operational costs were estimated to include one to four full-time employees, depending on the alternative, to monitor the well field, pipelines, collect samples, oversee maintenance, and monitor a Supervisory Control and Data Acquisition (SCADA) system. Operational and energy costs comprised the majority of annual expenditure for the alternatives. The capital and annual costs for the six alternatives are summarized below:

	Total Capital Costs	Total Annual Costs
Alternative 1	$7,470,000	$538,000
Alternative 2	$6,486,000	$453,000
Alternative 3	$6,803,000	$503,000
Alternative 4	$11,903,000	$1,643,000
Alternative 5	$10,275,000	$1,828,000
Pilot Study	$642,000	$123,000

Because of the high capital costs of Alternatives 1 through 5, no further cost analysis (i.e., present worth calculation) was performed. Groundwater modeling indicated that the pumpage rates for the evaluated alternatives were not adequate for control or remediation of the chloride plume. Therefore, complete plume remediation is likely to cost several times those given in above table. The pilot study was developed in response to the previous alternative costs estimates, as requested by KCC. This final alternative is intended as a initial step in addressing the need for action regarding the plume and to develop operating data to refine the assumptions used in this analysis. The pilot study operation will be the basis for further evaluation of remedial alternatives and possible design of larger systems.

STUDY FINDINGS

Previous studies by the Kansas Geological Survey (KGS) indicates that a withdrawal rate in access of 30,000 ac-ft/year (18,000 gpm) may be required to completely control the plume. In general, the Equus Beds Recharge Project reduces the flow gradients in the aquifer which aides the management of the plume. Use of the recovered water as a municipal supply is a positive beneficial use. Samples of groundwater from throughout the study area need to be obtained and tested for a complete EPA Safe Drinking Water Act set of parameters to evaluate the suitability of the water to be used as a municipal drinking water source. Part of the plume control strategy should include an evaluation of the high pumping rates down-gradient of the City of Wichita well field operation and analysis of methods to optimize pumping operations to mitigate the impacts of the salt plume.

The results of this study are important in showing that large scale pumping to control or remediate the Burrton brine plume would induce high chloride water into the study area from the Arkansas River and deep bedrock sources. This means that careful planning, siting of a remediation system, and coordination with other Equus Beds Aquifer Projects is critical on developing a good long term solution that does not cause negative impacts in other parts of the study area.

* * * * *

Introduction

INTRODUCTION

A. PURPOSE

Groundwater in the vicinity of Burrton, Kansas has been impacted by elevated chloride concentrations of greater than 2,000 mg/L, primarily caused by past oil field operations in the region. The plume of high chloride groundwater is expanding and migrating southeast in the Equus Beds Aquifer, threatening to contaminate a larger area of the aquifer which is used for municipal, industrial, and agricultural water supplies. The contaminated area around Burrton was designated as the Burrton Intensive Groundwater Use Control Area (IGUCA) by the state of Kansas in 1985. The IGUCA is within the boundaries of Equus Beds Groundwater Management District No. 2 (GMD2) which was formed in 1975 to manage the aquifer. To address the high chloride groundwater problem, the Kansas Corporation Commission (KCC) is seeking alternatives for the control, disposal, and/or beneficial use of high chloride water from the Burrton IGUCA.

The objectives of this evaluation are to define the hydrogeology of the groundwater in the study area, model the effects of various remedial alternatives on the chloride concentrations, and to outline potential alternatives for the control, disposal, and/or beneficial use of high-chloride groundwater from the Burrton IGUCA.

B. SCOPE

The major tasks performed in this study include the following:

- Prepare a subregional model including the Burrton IGUCA using a modified version of the USGS MODFLOW groundwater flow model to determine recovery well capacity, spacing and well/screen design criteria.

- Establish remediation scenarios for modeling, including no-action, pump-and-treat, deep well injection, and aquifer recharge components.

- Develop a contamination transport model, using the MT3D modeling program, to evaluate chloride migration in the Equus Beds Aquifer with no action and selected remediation scenarios.

- Use results of modeling scenarios to help determine potential alternatives to control, use and/or disposal of high-chloride groundwater.

- Coordinate up to two meetings with KCC, state regulatory agencies that have regulatory authority and local governments that may be impacted by the continued migration of the high chloride plume.

- Prepare estimates of capital, operation and maintenance costs for each feasible alternative developed in the engineering evaluation.

- Meet with KCC personnel to review alternatives and incorporate appropriate comments into the engineering evaluation.

- Prepare an engineering report summarizing the evaluations performed with exhibits, cost estimates, conclusions and recommendations.

* * * * *

Part I - Site Description

PART I

SITE DESCRIPTION

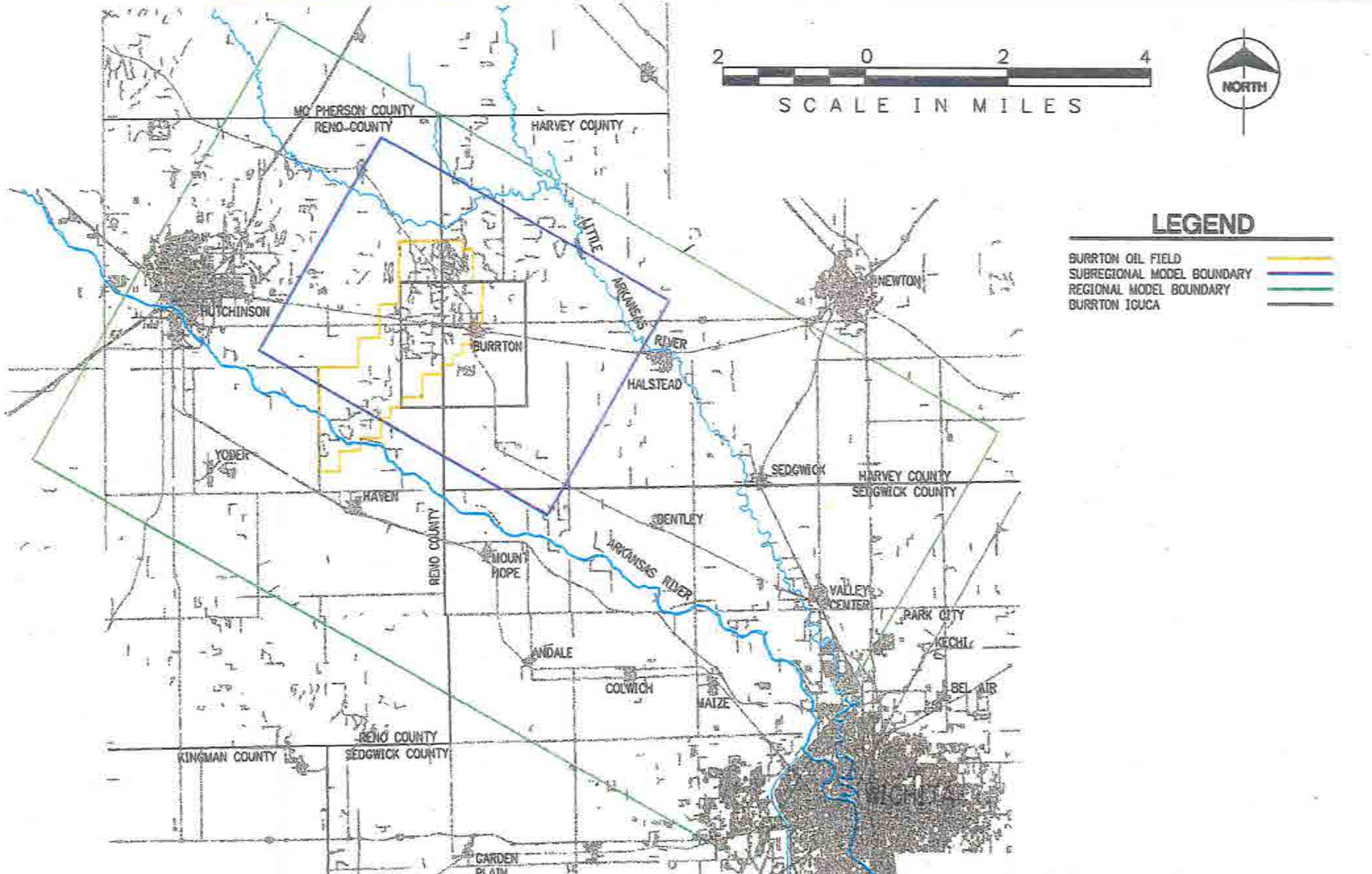
A. GENERAL

The Equus Beds Aquifer is the primary source of potable and irrigation water in south-central Kansas. The quality of the groundwater in the aquifer is generally very good, although salinity, indicated by the presence of chloride, has entered the aquifer from several sources. In the Burrton area, past oil field practices have resulted in a large area of high chloride groundwater. The hydrogeologic setting and large volume of groundwater pumpage within the study area have caused migration of the chloride plume, impacting agricultural and municipal wells.

B. STUDY AREA

The study area for this evaluation is shown in Figure I-1 as the subregional model boundary. This area occupies approximately 190 square miles and encompasses the majority of chloride impacted groundwater originating from the Burrton oil field. The subregional model is a portion of a larger regional groundwater model developed by the US Geological Survey (USGS) and refined by the US Bureau of Reclamation (USBR) to study the hydrogeology and salt migration in the area. The boundary of the regional model is located in south-central Kansas in parts of Reno, Harvey, and Sedgwick Counties.

The study area exhibits characteristics of a continental climate, with large variations in seasonal temperatures, moderate precipitation, and windy conditions. Temperatures range from daily averages of 2 °F in January to 81.4 °F in July. Temperatures reach a maximum of greater than 100 °F in the summer to less than -20 °F in winter. Average annual precipitation is about 30 inches per year, mostly in the form of rainfall. Approximately 15 inches of snow per year falls between December and March.



Burns & McDonnell

Figure I–1

STUDY AREA

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BURTON=burton.dwg

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Topographic variation in the area is minimal, with land surfaces gently sloping toward major streams in the area.

C. SOURCES OF ELEVATED CHLORIDE CONCENTRATIONS

There are two natural sources of salinity entering the Equus Beds Aquifer, the infiltration of surface water from the Arkansas River and saltwater intrusion from the Wellington Formation. Three other sources of chlorides in the groundwater include brine from evaporation-pans of salt mining activities, brine from oil field operations, and migration of saltwater from the Wellington Formation through improperly constructed wells.

The Arkansas River receives saltwater discharge from Permian formations upstream of Hutchinson, Kansas. This is the source of high chloride concentrations in the river water which averages about 630 mg/L of chloride near the project area. Wells in the Equus Beds Aquifer located near the Arkansas River exhibit higher chloride concentrations than wells located further from the river (Spinazola, 1985). Infiltration of high chloride water from the river continues to impact the aquifer. Previous groundwater modeling studies (Myers, 1996, Pruitt, 1993, and Burns and McDonnell, 1994) demonstrate the interaction of the Arkansas River with the Equus Beds Aquifer and the impact of high chloride river water migrating into the aquifer.

High concentrations of chlorides are found in the deeper parts of the aquifer (a bedrock low) near the Arkansas River near the study area. These chlorides may have migrated from the Hutchinson Salt Member through fractures in the upper shale of the Wellington Formation. Chloride concentrations in this water have been detected as high as 4,000 mg/L (Whittemore, 1990).

During the late 1890's and early 1900's, salt companies in Hutchinson conducted salt-solution mining of the Hutchinson Salt Member of the Wellington Formation. Waste from evaporation-pans is reported to be a significant source of salinity in some parts of the Equus Beds Aquifer (Whittemore, 1990).

Permian saltwater and oil field brine migrating up around poorly cased disposal wells or poorly plugged boreholes are other possible sources of chlorides in the aquifer (Whittemore, 1990). Prior to the practice of injecting oil field brines into deep formations, the brines were commonly injected into the Permian Wellington Formation that underlies the Equus Beds Aquifer. In areas where the potentiometric surface in the Permian is greater than the water table elevations in the Equus Beds Aquifer, these wells and boreholes provide a potential conduit for the flow of brine from the Permian into the Equus Beds.

The primary source of groundwater salinity in the project area originates from the Burrton and Hollow-Nickel oil field activities. During the first half of the 20th century, numerous oil and gas wells were drilled in the vicinity of Burrton, Kansas. Saltwater brine, a by-product of oil and gas production, was pumped from the subsurface along with oil and gas. The brine was separated and discharged into nearby creeks and streams during original well discovery. In the 1930's and early 1940's, this practice was prohibited. The brine was then disposed of in evaporation pits or shallow injection wells until this practice was outlawed in the 1950's and the remaining pits were closed. Secondary sources are return flow from shallow disposal wells and the leaks in pipelines leading to these wells.

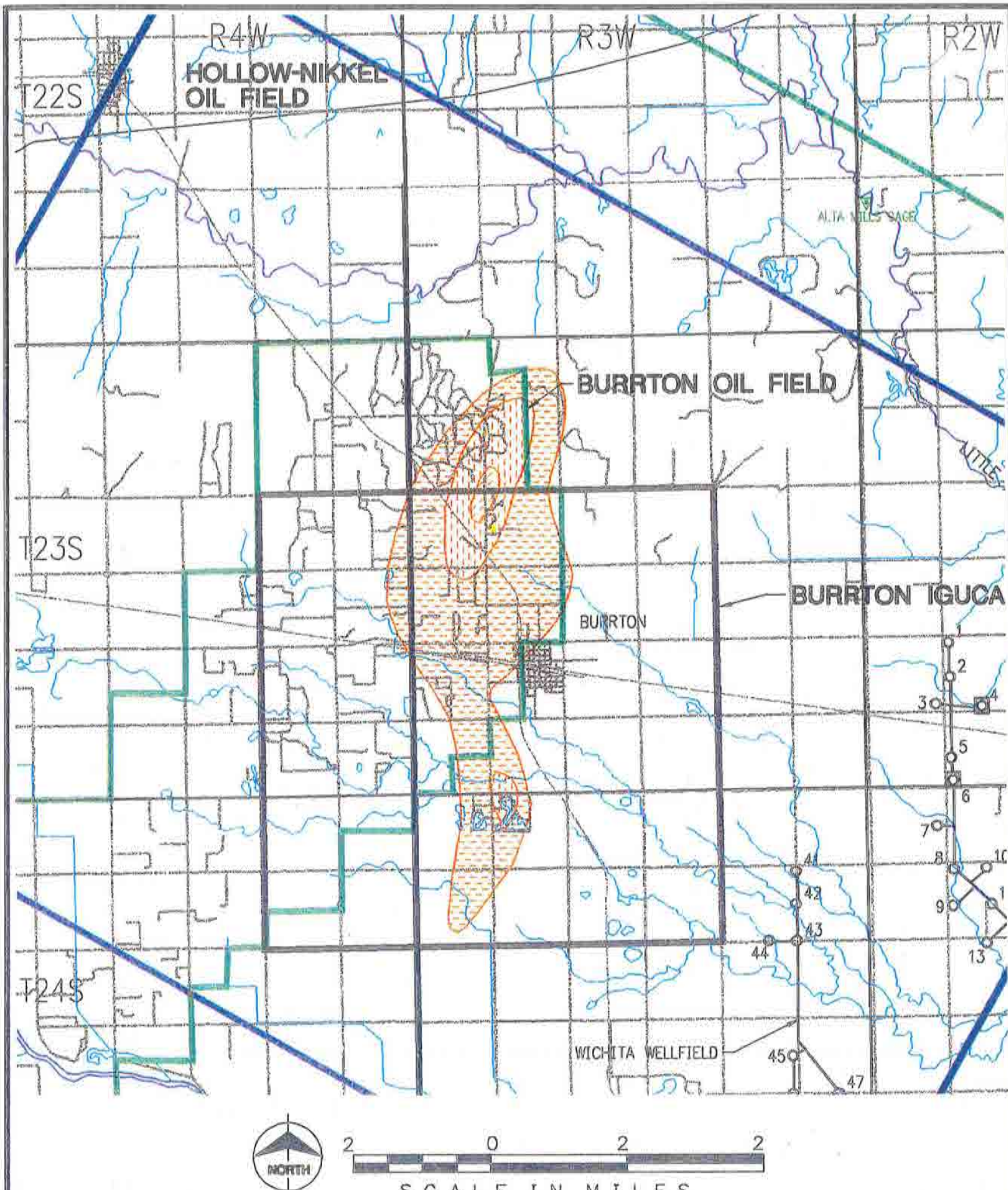
Several million barrels of brine were disposed of, much of which migrated into the shallow aquifer system. The resulting groundwater contamination plume is characterized by chloride concentrations currently exceeding 2,000 mg/L in some areas of the aquifer. Large areas of the aquifer have chloride concentrations exceeding the maximum contaminate level (MCL) of 250 mg/L. The Report of the Burrton Task Force (1984) concluded that about 1.9 million tons of salt was produced by oil field operations and that a large percentage of the salt has entered the aquifer. Additionally, some unknown amount is probably tied up in the soil due to closure of the brine ponds (Burrton Task Force, 1984).

D. EXTENT OF CONTAMINATION

A chloride plume in the Equus Beds Aquifer originating from the Burrton oil field area was recognized in the late 1940's. From 1948 until the use of shallow injection wells and evaporation pits ceased, the area of the plume expanded and concentrations levels increased. After the pits and shallow wells were closed, the plume continue to spread, migrating downgradient and moving to lower levels in the aquifer. The Kansas Corporation Commission (KCC) developed chloride concentration maps for the aquifer zones less than 50 feet and greater than 50 feet deep for 1948 and 1982 as reported in the Report of the Burrton Task Force (1984). These maps were reproduced and are shown in Figures I-2 through I-5.

Chloride concentration maps were developed for the study area based on data obtained during 1996 from Equus Beds Groundwater Management District No. 2 (GMD2) monitoring wells in the area. GMD2 monitoring wells have been constructed to track chloride values in three levels of the aquifer. A map was created for each of three hydrostratigraphic units described below:

- Level A, upper unit -- generally less than 50 feet below land surface (BLS) (Figure I-6)
- Level B, middle unit -- generally greater than 50 feet and less than 170 feet BLS (Figure I-7)
- Level C, lower unit -- generally greater than 170 feet BLS and above bedrock (Figure I-8)



CHLORIDE CONCENTRATION, mg/l

GREATER THAN 1,000 CL

GREATER THAN 500 CL

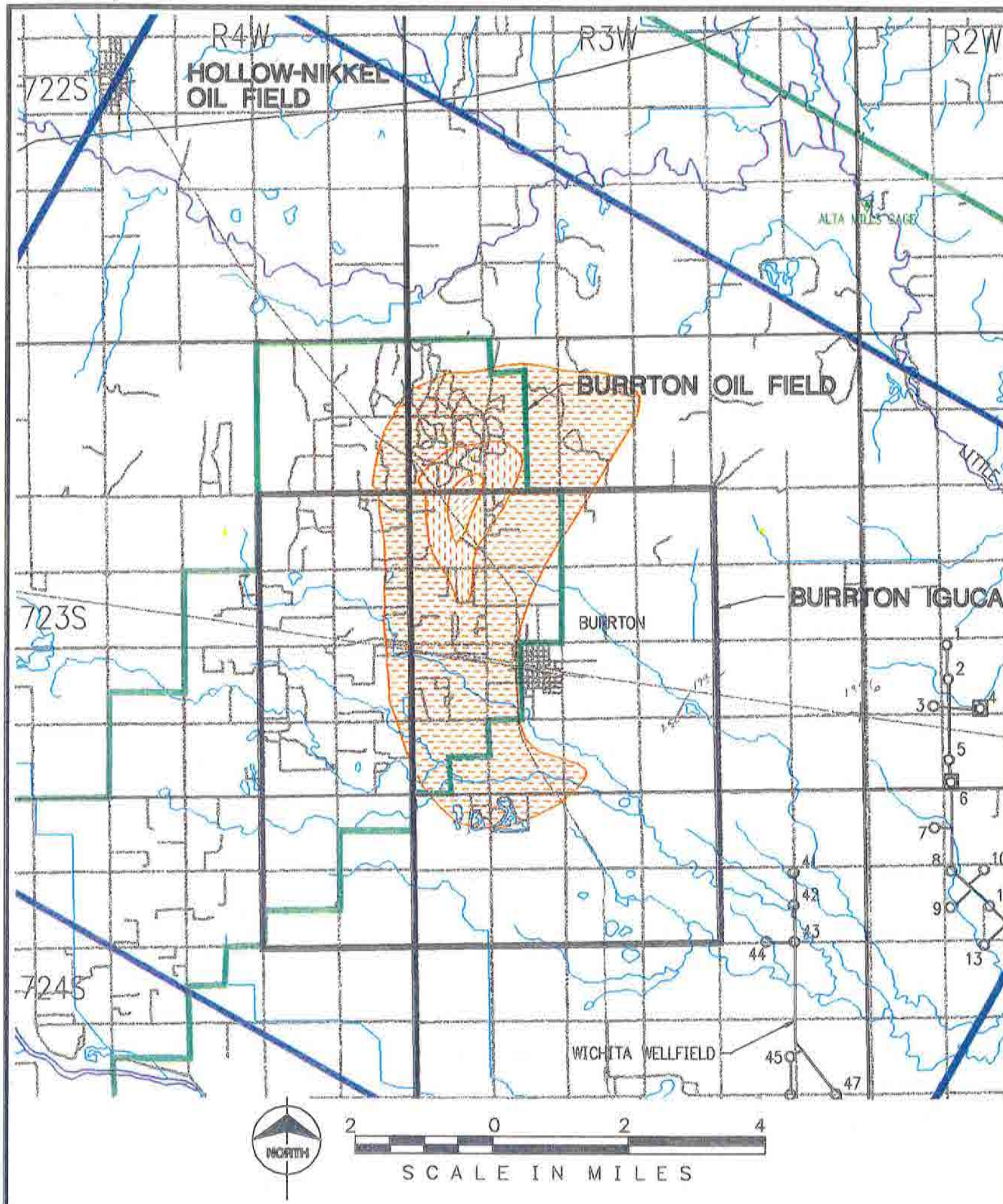
GREATER THAN 250 CL

SOURCE: BURRTON TASK FORCE REPORT (1984)

Burns & McDonnell

Figure I-2

CHLORIDE CONCENTRATION DATA 1948 LESS THAN 50 FT. DEEP



CHLORIDE CONCENTRATION, mg/l

GREATER THAN 1,000 CL

GREATER THAN 500 CL

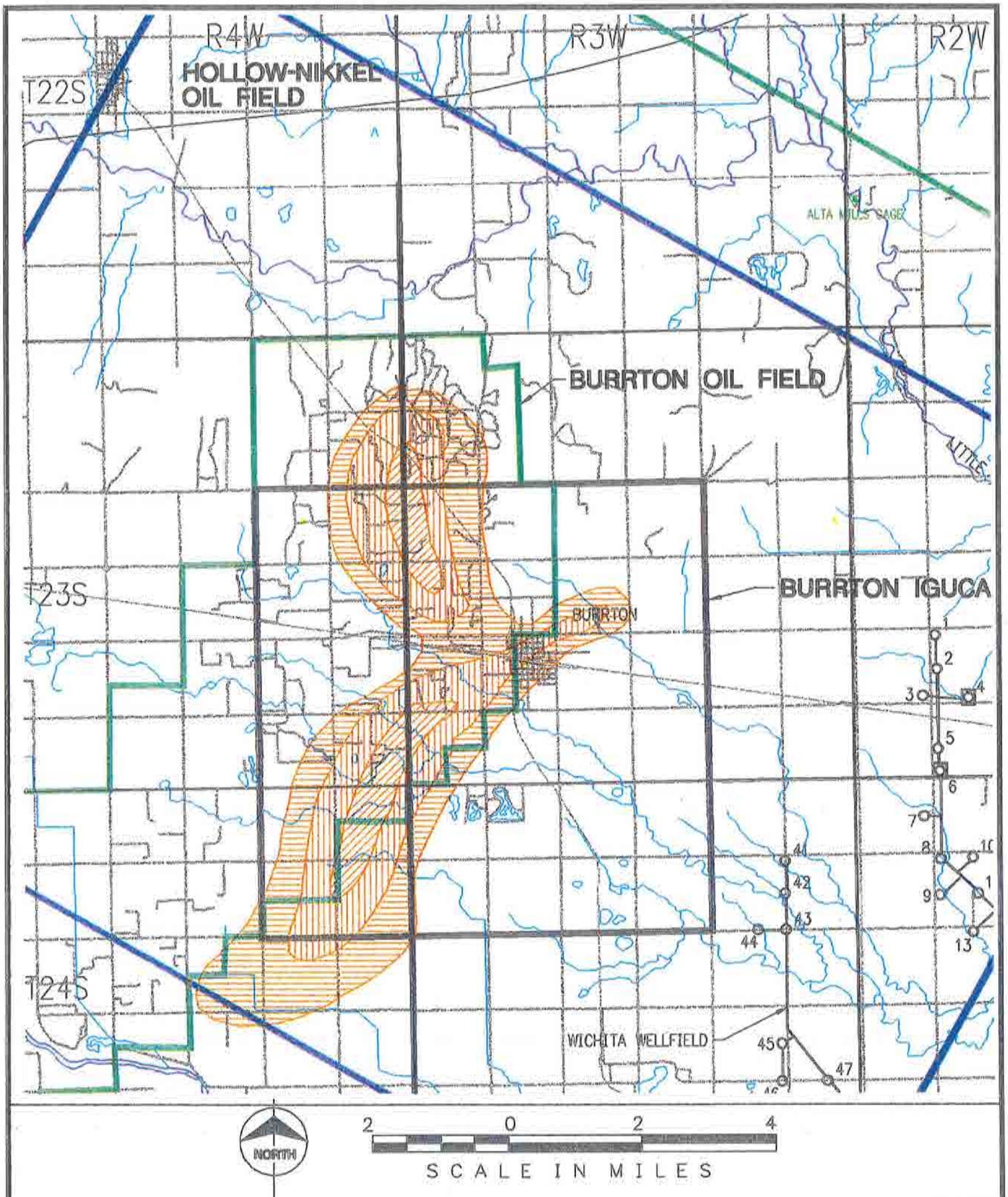
GREATER THAN 250 CL

SOURCE: BURRTON TASK FORCE REPORT (1984)

Burns & McDonnell

Figure I-3

CHLORIDE CONCENTRATION
DATA 1948
DEEPER THAN 50 FT.



CHLORIDE CONCENTRATION, mg/l

GREATER THAN 1,000 CL

GREATER THAN 500 CL

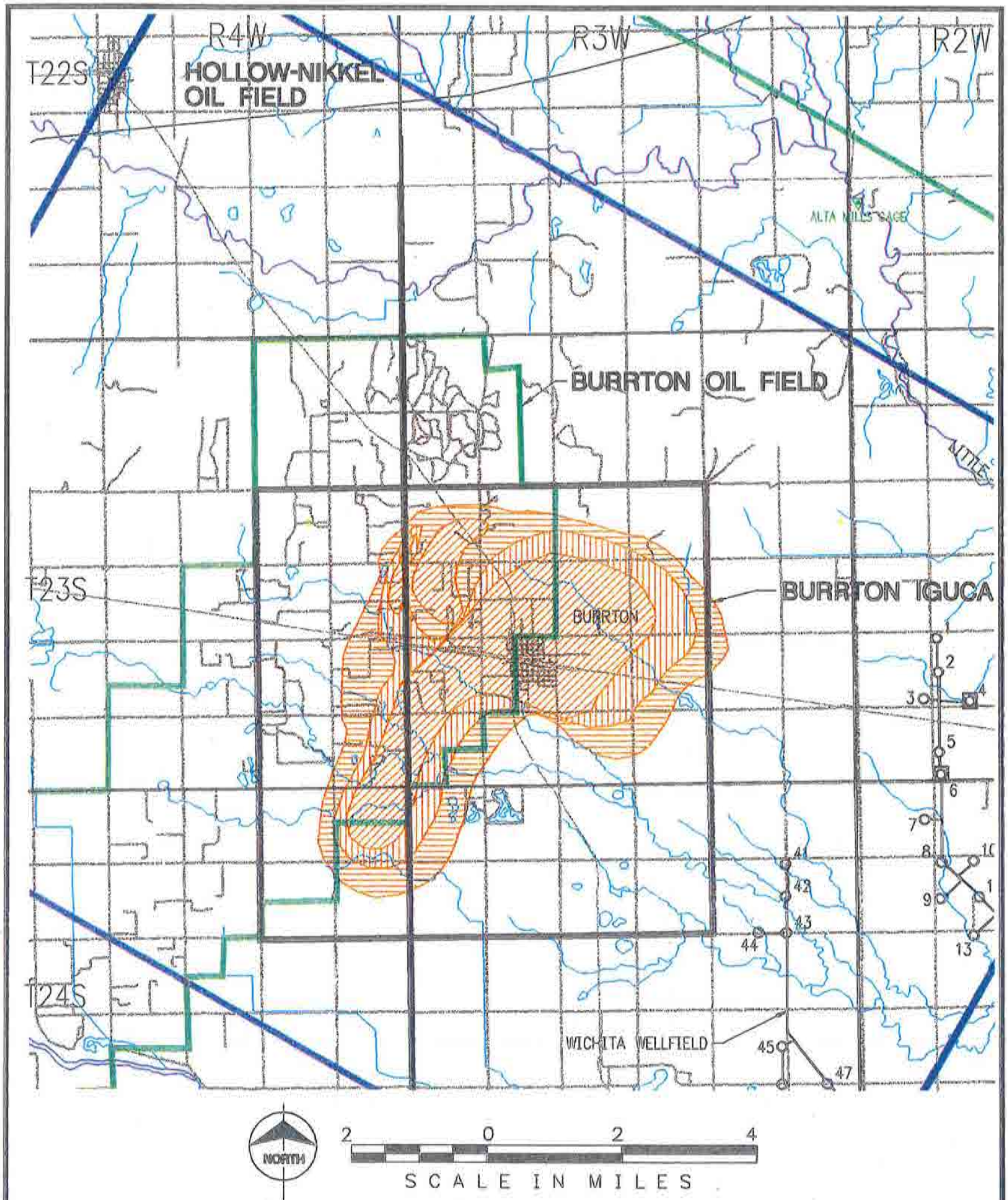
GREATER THAN 250 CL

SOURCE: BURRTON TASK FORCE REPORT (1984)

Burns & McDonnell

Figure I-4

CHLORIDE CONCENTRATION DATA 1982 LESS THAN 50 FT. DEEP



CHLORIDE CONCENTRATION, mg/l

- GREATER THAN 1,000 CL
- GREATER THAN 500 CL
- GREATER THAN 250 CL

SOURCE: BURRTON TASK FORCE REPORT (1984)

Burns & McDonnell

Figure I–5

CHLORIDE CONCENTRATION DATA 1982 DEEPER THAN 50 FT.

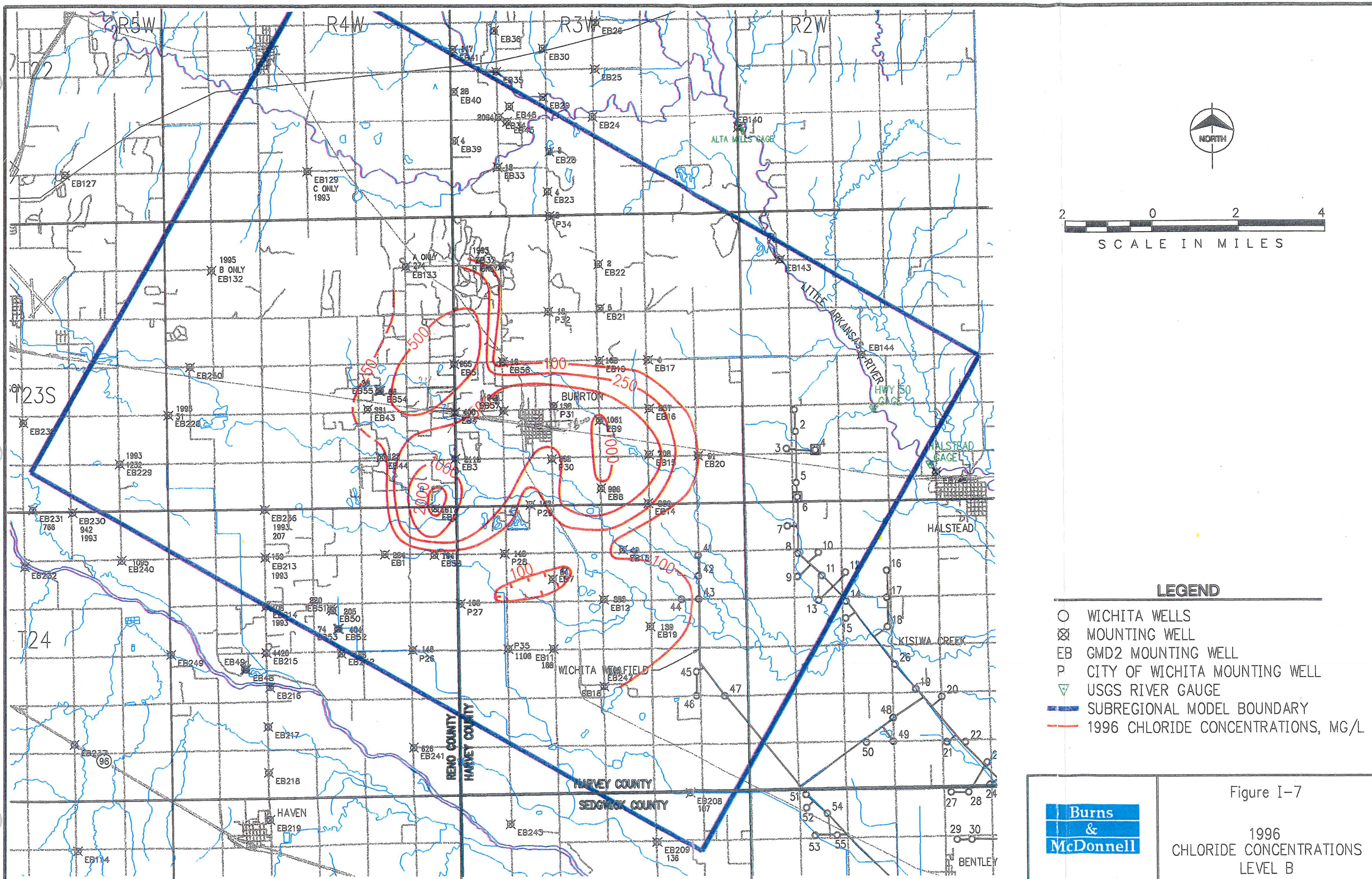
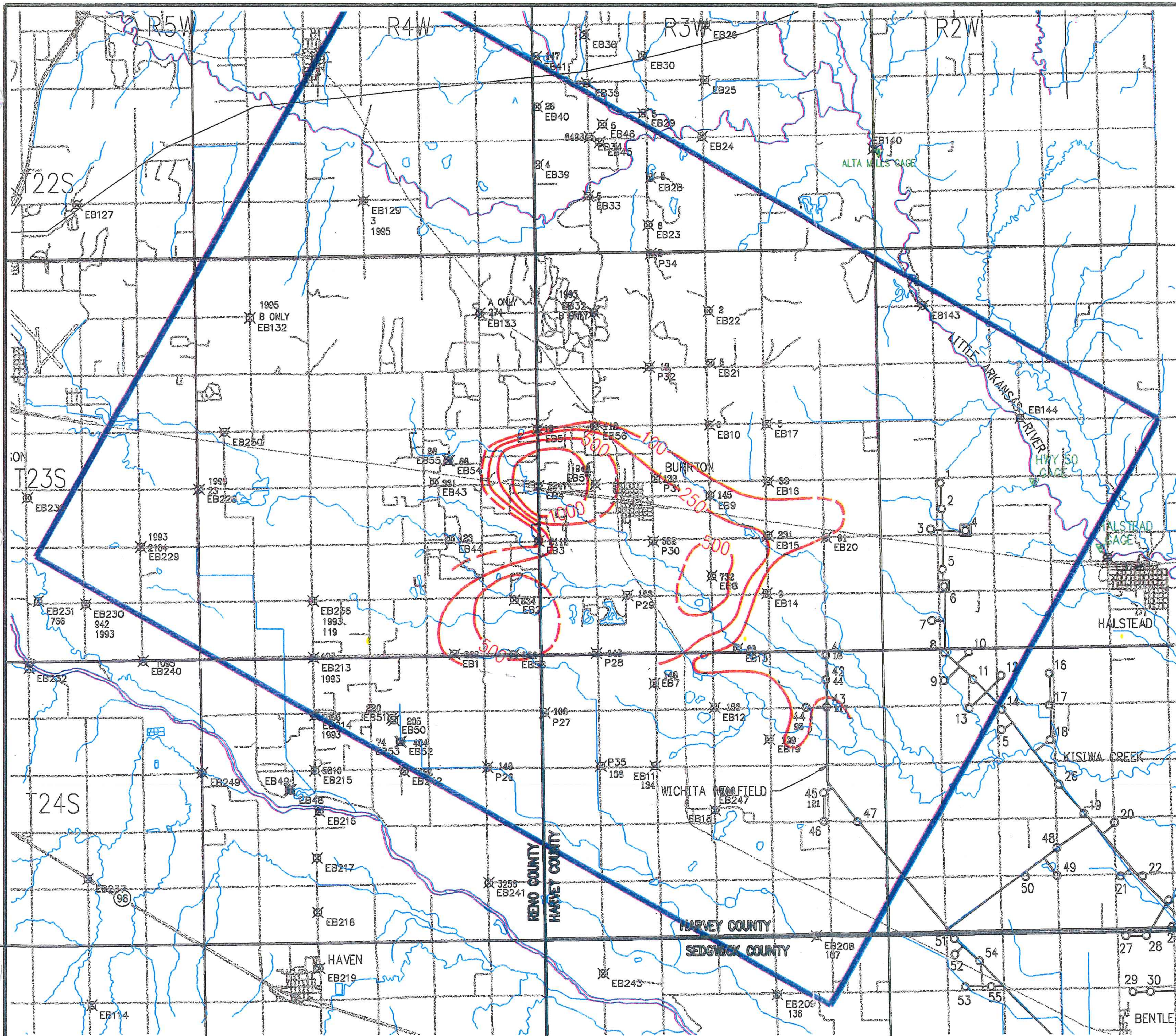


Figure I-7

1996
CHLORIDE CONCENTRATIONS
LEVEL B



2 0 2 4

SCALE IN MILES

LEGEND

- ○ WICHITA WELLS
- ⊠ MOUNTING WELL
- EB GMD2 MOUNTING WELL
- P CITY OF WICHITA MOUNTING WELL
- ▽ USGS RIVER GAUGE
- SUBREGIONAL MODEL BOUNDARY
- 1996 CHLORIDE CONCENTRATIONS, MG/L

Burns & McDonnell

Figure I-8

1996 CHLORIDE CONCENTRATIONS LEVEL C

From 1982 through 1996, the zone of contamination has expanded to the east particularly in Level A, the upper unit of the aquifer. Clay layers in the aquifer tend to perch the high chloride water, slowing its downward migration and causes the salt water to spread laterally in the shallower layers. As the salt water moves to the edges of the clay layers it can migrate downward to the next clay layer in a "stair-step" manner.

Work by the Kansas Geological Survey had shown that some salt remains in the unsaturated zone beneath the old pit areas. Little or no salt was found in control borings located outside the pit area. The salt in the unsaturated zone can provide a continuing source of chlorides to the aquifer due to leaching by recharge from precipitation, especially in areas with a high density of old brine evaporation pits. Although this source of salt is expected to slow natural attenuation of the plume, the impacts are expected to be relative minor compared to the magnitude of the salt plume (Don Whittemore, personal communication).

The Equus Beds Aquifer is the principal source of raw water for surrounding municipalities including Burrton, Halstead, and Wichita. The City of Wichita, Kansas has a large well field consisting of 55 wells. The City has typically used 60% groundwater from the Equus Beds Aquifer for municipal water supply. Groundwater withdrawal rates from the aquifer in the study area have steadily increased since the 1940's. At the present time, water rights and pumpage of all users exceed the natural recharge rate of the aquifer, estimated to be three to six inches per year. This overdevelopment of the aquifer has resulted in a reduction of static water levels. Originally, the depth to water in the Equus Beds Aquifer was relatively shallow, ranging from 10 to 20 feet below land surface. Currently depth to water ranges from 10 to 60 feet. In the area downgradient of the Burrton Intensive Groundwater Use Control Area (IGUCA), water rights have been filed to pump approximately 27.8 billion gallons per year by approximately 450 wells. Forty-eight percent of these water rights are for municipal use, and 51 percent are for irrigation use (Burns & McDonnell, 1994).

Water obtained from the Equus Beds Aquifer is generally of good quality. Changes in the groundwater flow gradients, caused by pumping, is expected to cause changes in groundwater quality by increasing infiltration from the Arkansas River and causing faster migration of the Burrton salt plume. Groundwater modeling performed by the Bureau of Reclamation (Pruitt, 1993) indicates that the average chloride concentration in the Wichita well field will increase from 55 mg/L at the present time to approximately 95 mg/L in year 2010 and 145 mg/L in year 2050. Maximum chloride levels could exceed 300 mg/L in some areas, well above the sensitivity level of 200 mg/L for agricultural uses and 250 mg/L, the Secondary Maximum Contaminant Level (SMCL) for municipal uses.

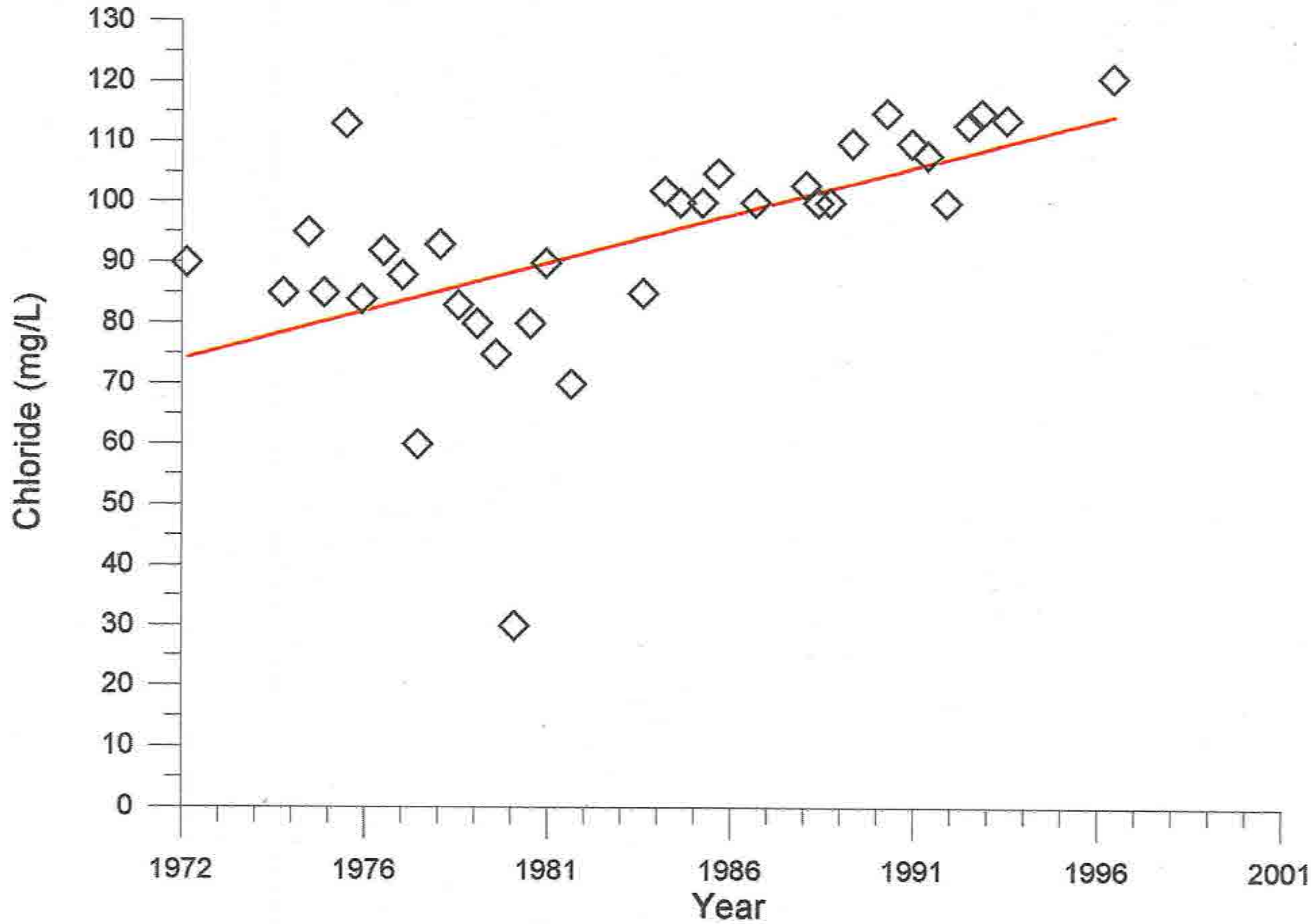
The Wichita City Wells Nos. 1 through 15 and 41 through 45 are downgradient of the Burrton chloride plume. They are also downgradient of the chlorides migrating from the Arkansas River and the deep saltwater originating from the Wellington Formation. Since 1972, the chloride levels in Wichita City Well No. 45 have risen from 75 mg/L to 113 mg/L (Figure I-9).

E. GEOLOGY

1. Physiography

The study area is located near the boundary of the Great Bend Prairie physiographic region and the High Plains region of the Central Lowlands physiographic province. Based on aquifer characteristics, the Great Bend Prairie and western areas of Kansas underlain by the Ogallala Formation have been grouped into one groundwater region and is considered to be part of the Great Bend region. The Great Bend Prairie physiographic province (also known as the Wellington and McPherson Lowlands) is characterized by large areas of low topographic relief.

This area of low relief is disrupted by a belt of sand dunes trending northwest-southeast along the northeast side of the Arkansas River Valley (Williams and



Burns & McDonnell

Figure I-9

CHLORIDE CONCENTRATIONS
WICHITA CITY WELL NO. 45

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Lohman, 1949 and Hathaway et al, 1981). Wind-blown sand and silt form another major belt of sand dunes extending southwestward from Rice County across Reno County, between the northern edge of the Arkansas River Valley and the Little Arkansas River to an eastern terminus northeast of Burrton in Harvey County, Kansas. In addition, small isolated sand dunes occur locally in the area. Soils in the area include excessively drained soils with loamy or silty subsoil on the uplands, well-drained soils with clayey subsoil on ridges and side slopes, imperfectly drained and loamy soils with clayey subsoils in well-drained sandy soils on level plains, and deep loamy soil over sandy or gravelly material in the breaks and along alluvial lands.

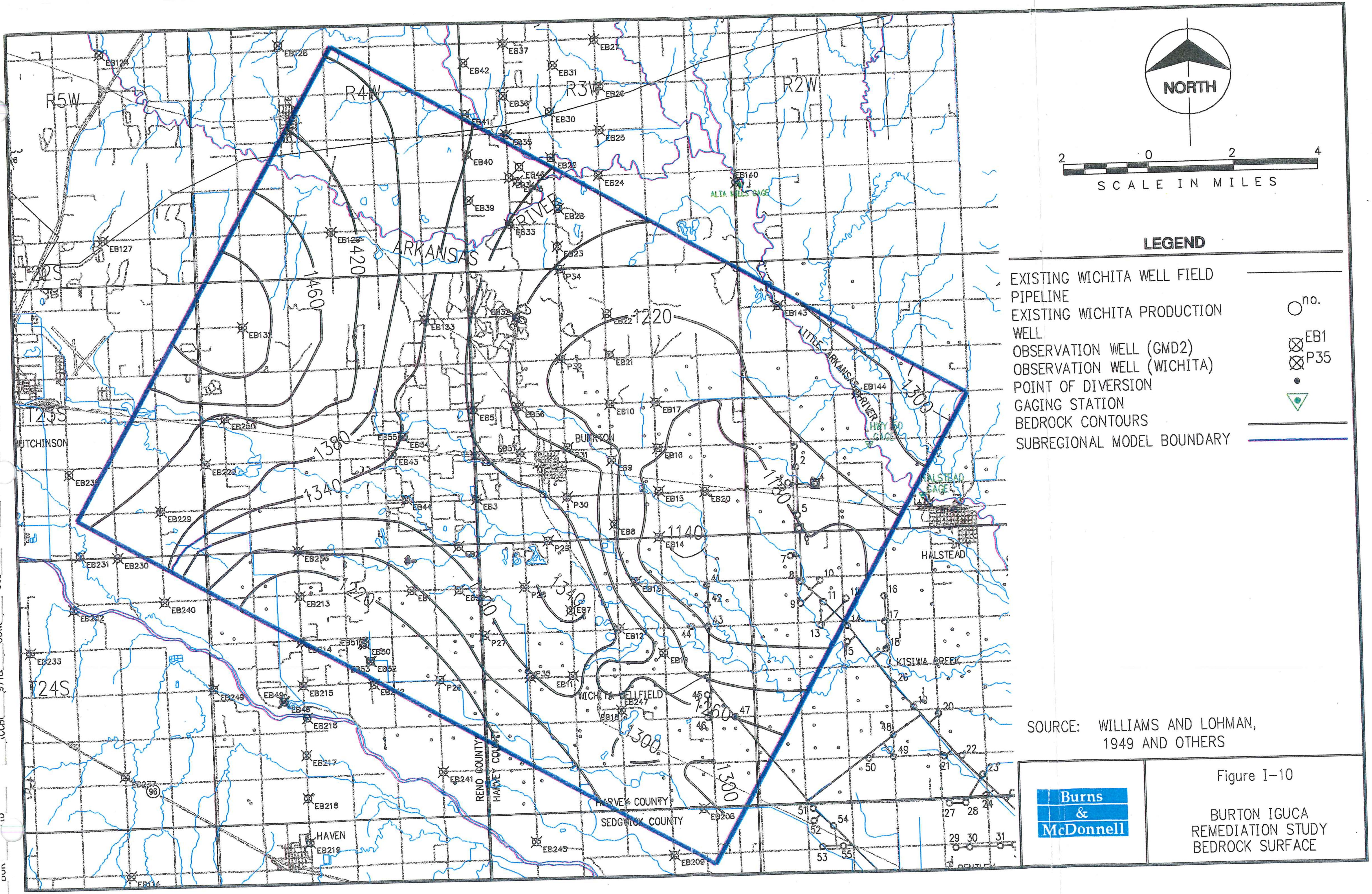
2. Topography and Drainage

Land surface elevation in the study area range from a low of about 1380 feet above mean sea level (MSL) where the Little Arkansas River leaves the study area, to a high of about 1500 feet MSL near the northwest corner of the study area. Therefore, topography generally slopes downward to the southeast.

The major streams in the vicinity of the study area are the Arkansas River and the Little Arkansas River. The Arkansas River flows to the southeast in a relatively straight, slightly braided channel and is typically entrenched 5 to 10 feet below the adjacent land surface. The Little Arkansas River meanders, flowing east and southeast, with its channel entrenched 15 to 20 feet below the adjacent land surface. The confluence of the Little Arkansas River with the Arkansas River is in Wichita.

3. Bedrock

The bedrock underlying the unconsolidated deposits in the study area consists primarily of early Permian age (approximately 240 million years old) shales of the


NORTH
2 0 2 4
SCALE IN MILES
LEGEND
EXISTING WICHITA WELL FIELD
PIPELINE
EXISTING WICHITA PRODUCTION WELL
no.
OBSERVATION WELL (GMD2)
EB1
OBSERVATION WELL (WICHITA)
P35
POINT OF DIVERSION
GAGING STATION
BEDROCK CONTOURS
SUBREGIONAL MODEL BOUNDARY
SOURCE: WILLIAMS AND LOHMAN, 1949 AND OTHERS
Burns & McDonnell
Figure I-10
BURTON IGUCA REMEDIATION STUDY BEDROCK SURFACE
R5W
R4W
R3W
R2W
T22S
T23S
T24S
HUTCHINSON
ARKANSAS RIVER
LITTLE ARKANSAS RIVER
BURRTON
HALSTEAD
HAVEN
KISIWA CREEK
WICHITA WELLFIELD
RENO COUNTY
HARVEY COUNTY
SEDGWICK COUNTY
ALTA MILLS GAGE
HWY 50 GAGE
HALSTEAD GAGE
1460
1420
1380
1340
1300
1260
1220
1180
1140

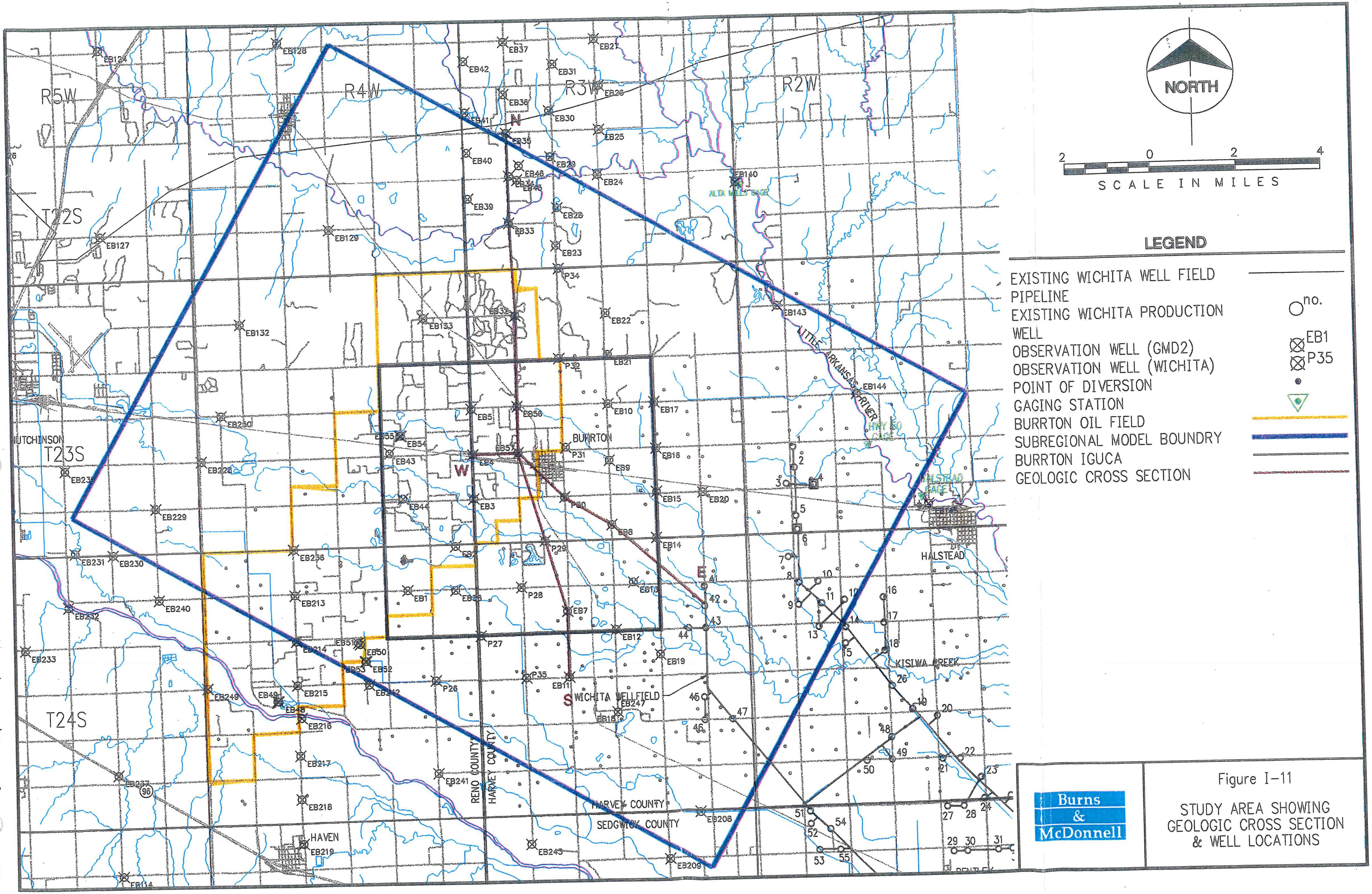


Figure I-11

STUDY AREA SHOWING GEOLOGIC CROSS SECTION & WELL LOCATIONS

J:\KCCBURTN\971834\PROCWS\FIGI-6.DWG 10-23-1997 MJC eastward from the McPherson Valley. Zeller (1968) classified the Pleistocene age sediments and indicates the McPherson Formation is comparable to the Grand Island Formation. A complete description of the unconsolidated materials have been provided by Zeller.

The sand dunes that occupy part of the area between the Little Arkansas River and the Arkansas River have been in the process of formation since early Pleistocene time. The older dunes probably were beginning to form while the streams depositing the McPherson formation were flowing southward through the area. The dunes consist mainly of fine- to medium-grained, well rounded quartz sand. Silt, clay, and organic material are locally mixed forming zones which may represent the formation of soils in quiescent periods and interdune pond areas. The higher sand dunes have been developed by prevailing southerly and southwesterly winds (Williams and Lohman, 1949).

5. <u>Soils</u>

Soils in the area tend to be sandy and loamy with poorly defined surface drainage features. Soil associations were mapped by Hoffman, Dowd, and others by combining data for individual counties, grouping various soil associations presented on county soil maps, and some regrouping of mapping units to better reflect the proportional pattern of soils in natural landscapes for a multi-county area. Hathaway et al (1981) provides an extensive description of the soils in this area.

6. <u>Structure</u>

The strata were effected by the building of the Rocky Mountains during which the rocks were uplifted and tilted. Erosion following uplift (approximately 63 million years ago) removed overlying rocks and exposed the salt-bearing part of the Wellington Formation to solution by ground and surface waters. A depressional area resulted in the eastern part of the region and became progressively deeper as solution and erosion proceeded westward down the dip of the Wellington. Early in the Pleistocene Epoch (2 million years ago) a stream flowing southwestward entered this area near Lindsborg and continued past McPherson and Mound Ridge southward, west of Wichita forming a channel. The channel extends southward where it converges with another channel. Toward the end of the first stage of the Pleistocene Epoch, the streams which had been active in eroding the McPherson Valley apparently became overloaded and deposited their suspended load and bed load over much of the valley. A large unconformity occurs between the older deposits of the McPherson Formation and the channel deposits (Williams and Lohman, 1949).

Because of either a smaller supply of water, an abundance of available detritus, or subsidence of the bedrock surface due to dissolution, the valleys formed by stream erosion began to be filled. After this valley filling had been in progress, it is believed that the stream flowing southward into the area was captured by headward erosion of an eastward-flowing stream in the vicinity of Salina, and the direction of flow was reversed.

The study area is located at the conjunction of the Salina and Sedgwick Basins. The Salina Basin, or Central Nebraska Basin, is limited on the east by the Nemaha Anticline, on the west by the Cambridge Arch and Central Kansas Uplift, and on the south by an indistinct, unnamed saddle. The axis of this post-Mississippian syncline trends northwest and plunges northward into the deeper part of the basin in north-central Kansas. The basin extends over an area of about 12,700 square miles and is the second largest basin in Kansas. Minor structures in the basin include the Abilene Anticline, the northern part of the Voshell Anticline, and the Wilson-Burns Element (Merriam, 1963).

The Sedgwick Basin is a shelf-like, southerly plunging area in south-central Kansas, with an area of about 8,000 square miles. It is a major pre-Desmoinesian, post-Mississippian feature bounded on the east by the Nemaha Anticline, and the Pratt Anticline to the west. An indistinct saddle separates it from the Salina Basin on the north. Minor structures, approximately parallel with the Nemaha Anticline, have been recognized in the basin. These include the Bluff City Anticline, Conway Syncline, Elbing Anticline, Halstead-Graber Anticline, and the southern end of the Voshell Anticline (Merriam, 1963).

Faults are located along the eastern and western boundaries of the basins associated with the Nemaha Anticline to the northeast and the Central Kansas Uplift to the west. A normal fault, approximately 35 miles in length, trends in a northwest corner of Harvey County, and into central McPherson County (Merriam, 1963). One small earthquake occurred in the Salina Basin, as well as a moderate one in the area between the Sedgwick and Salina Basins.

F. HYDROGEOLOGY

The Equus Beds Aquifer is the eastern-most portion of the High Plains Aquifer system in Kansas. The Equus Beds are named for the equine fossils found in the unconsolidated sediments in this area. Three hydrogeologic units are generally recognized in the area; an upper sand and gravel unit; a middle fine grained (fine sand, silt, and clay) unit; and a lower coarse grained unit. These units are not consistent and vary greatly throughout the area. Typical cross-sections were developed. Their locations are shown in Figure I-11. The cross-sections are presented in Figure I-12.

Variations in bedrock elevations cause large variations in the saturated thickness of the Equus Beds Aquifer in the Wichita well field area. Saturated thickness within the study area ranges from less than 100 feet to over 200 feet.

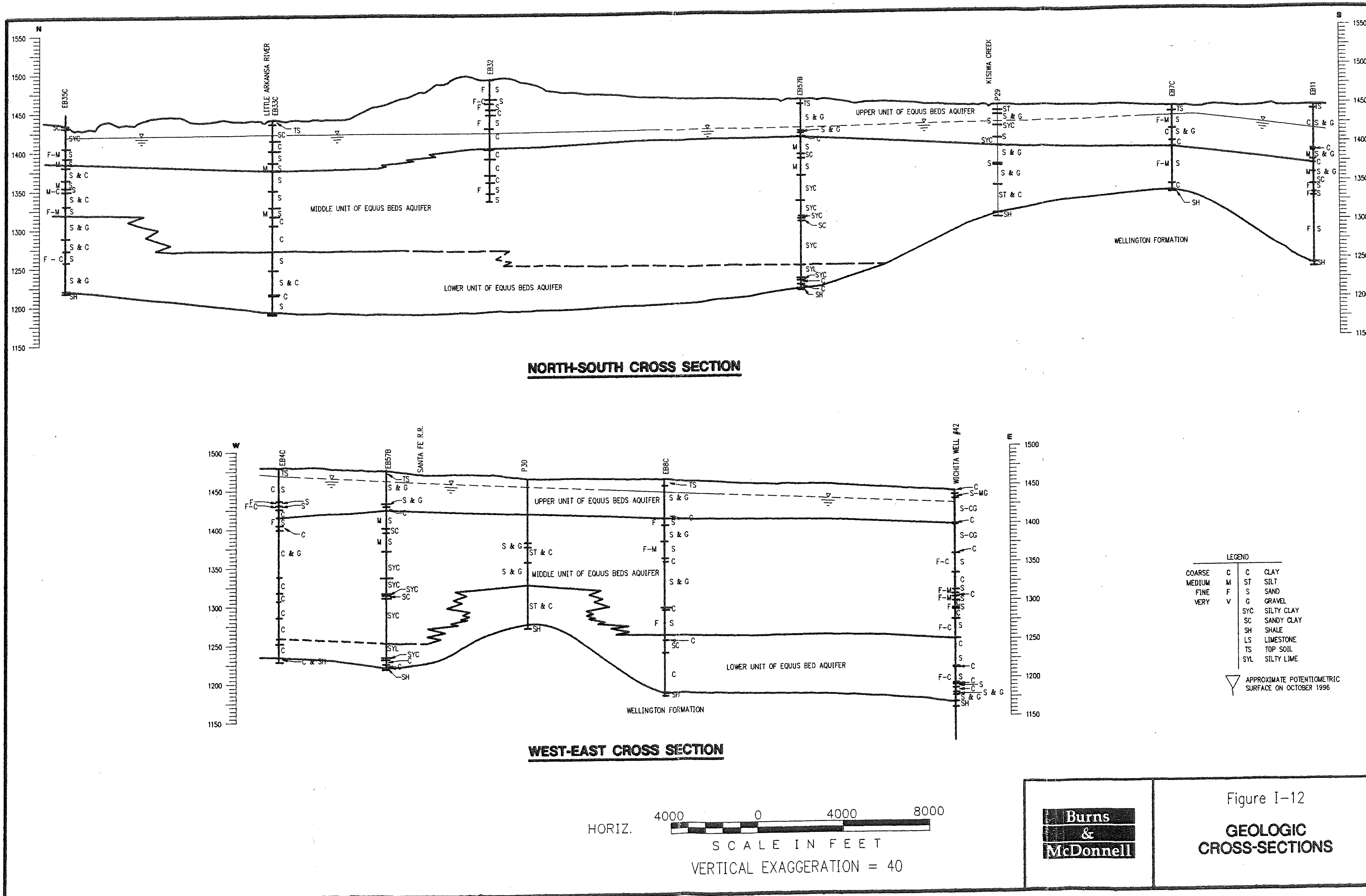


Figure I-12

GEOLOGIC CROSS-SECTIONS

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Originally, the depth to water in the Equus Beds Aquifer was relatively shallow, ranging from 10 to 20 feet below land surface. After decades of municipal and irrigation pumping, current depth to water ranges from 10 to 50 feet. The original water table had a southeast gradient of approximately 6 feet per mile. Today, the groundwater gradient is modified by pumping; however, there is still an eastward gradient allowing flow toward the Little Arkansas River. Figure I-13 shows the surface of the water table in 1996.

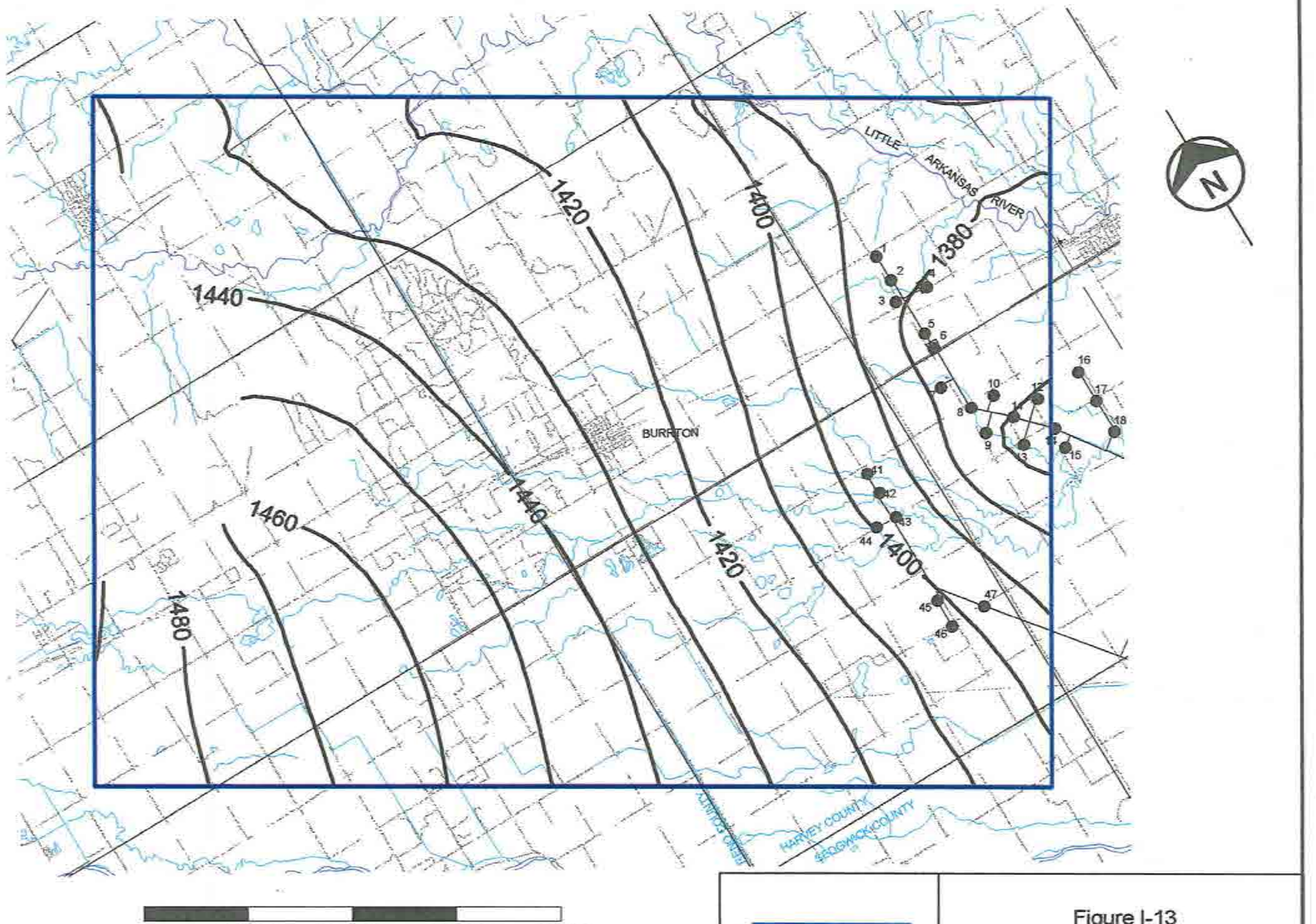
1. Recharge

The primary sources of recharge to the Equus Beds Aquifer are precipitation and inflow from the Arkansas River. Recharge from precipitation is estimated to range from three to six inches per year which is approximately 10 to 29 percent of the annual precipitation of 30 inches. The flat topography and generally sandy nature of the surface soils southwest of the Little Arkansas River allows rapid infiltration of precipitation in the region.

The Equus Beds Aquifer is also recharged by underflow from aquifer materials from the west and stream losses in areas where the groundwater is lower than the surface water levels. The Arkansas River is currently believed to be a losing stream in the reach between Hutchinson and Wichita. Recent groundwater modeling estimates an average of 50 cubic feet per second (cfs) is entering the aquifer from the Arkansas River through this reach (Burns & McDonnell, 1994).

2. Discharge

Water is lost from the Equus Beds Aquifer from evapotranspiration, pumping, underflow out of the aquifer, and discharge to streams as baseflow. In areas that the water table is near land surface, groundwater can be lost directly from evaporation. Groundwater is also taken up by vegetation with deep root systems that intercept the water table, allowing absorbed water to be lost through transpiration to the atmosphere.



Burns & McDonnell

Figure I-13

MODELED WATER TABLE
1996

Note: Based on Pruitt Model (1996)

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The Equus Beds Aquifer within the study area was developed with wells beginning in the late 1930's and early 1940's. The initial water rights were established for municipal use. Later in the late 1960's and 1970's, a large volume of water rights were filed for irrigation. Within the Wichita well field area, water rights have been filed to pump approximately 27.8 billion gallons per year by approximately 450 wells. Forty-eight percent of these water rights are for municipal use and 51 percent are for irrigation use (Burns & McDonnell, 1994).

The bedrock high occurring north and east of the Little Arkansas River generally prevents underflow out of the Equus Beds in that direction. A small amount of underflow loss occurs to the southeast into alluvium of the Arkansas and Little Arkansas Rivers near the confluence of the two rivers (Burns & McDonnell, 1994).

Where the groundwater levels are higher than stream levels, water is lost from the aquifer to the stream as baseflow. Baseflow in the Little Arkansas River is provided by discharge from the Equus Beds Aquifer. As more water is removed by pumping, less water is available for seepage into the river as baseflow. Alternately, during times of higher groundwater levels due to lower pumping and recharge, baseflow is increased. Previous computer modeling suggests that pre-development baseflow to the Little Arkansas River was about 60.5 cfs and the recent baseflow to be about 27.4 cfs, a 33.1 cfs reduction over 50 years (Burns & McDonnell, 1994).

3. Aquifer Parameters

A number of aquifer pumping tests have been collected by the USGS from wells constructed in the Equus Beds Aquifer and evaluated to determine hydrogeological parameters throughout the aquifer. Aquifer parameters summarized from the USGS Open-File Report 85-200 are as follows:

	Average	Range
Transmissivity (feet^2/day)	13,100	34,000-7,300
Storage Coefficient (dimensionless)	0.03	0.16-0.0008

The upper materials generally act as an unconfined aquifer and materials below the intermediate fine grained materials (where present) act in a confined or semi-confined manner. In areas where fine-grained materials cause the aquifer to react as a confined system, such as the northern end of the Wichita well field area, large changes in water levels are noted with during pumping or from recharge.

4. Surface Hydrology

The study area is located several miles north of the Arkansas River and encompasses a portion of the Little Arkansas River Basin. The Little Arkansas River has a mean annual flow of 284 cubic feet per second (cfs) (205,600 acre-feet per year) at Valley Center, Kansas, based on 69 years of historical records. The Arkansas River has a mean annual flow of 1,014 cfs at the Wichita gage, based on 56 years of records. The gage at Wichita is below the confluence of the two rivers and therefore includes the flows from the Little Arkansas River (Burns & McDonnell, 1994).

Groundwater and river flow interact and move depending on water levels of the groundwater and river. The interaction is influenced to some extent by the conductivity of the river bed materials. Sediments in the Arkansas and Little Arkansas Rivers are relatively coarse, allowing rapid infiltration of water into the riverbank or rapid exfiltration of water to the stream.

* * * * *

Part II - Modeling

PART II
MODELING

A. GENERAL

Many studies have been conducted on the Equus Beds Aquifer because of its importance as a source of water for municipal, agricultural, and industrial use. The occurrence of chloride in parts of the aquifer, in streams, and bedrock has been a major concern and has resulted in several groundwater flow and transport modeling studies. The early models simulated the Equus Beds Aquifer as a single layer and focused on specific aspects of the groundwater flow system and the transport of chloride from specific concentrated sources (Myers et al, 1996). Groundwater flow models have been used to determine the long-term safe yield of the aquifer (Green and Pogge, 1977) and to describe the groundwater flow (Spinazola et al, 1985). The migration of chlorides in the Equus Beds aquifer has been studied using solute transport models, also (Sophocleous, 1983, Spinazola, 1985).

The US Geological Survey (USGS) was the first organization to develop a groundwater model of the Equus Beds Aquifer using MODFLOW, a USGS three-dimensional, finite-difference, groundwater flow model written by McDonald and Harbaugh (McDonald and Harbaugh, 1988). MODFLOW is a well documented model that is widely used and accepted by many regulatory agencies. This model uses a modular method of data entry to simulate specific aspects of the aquifer system such as wells, rivers, recharge, and evapotranspiration, along with aquifer properties.

The USGS office in Lawrence, Kansas developed a groundwater flow model to study the stream-aquifer system of the Arkansas River and the Equus Beds Aquifer (Myers, 1996). The USGS model area includes the current study area for the Burrton Intensive Groundwater Use Control Area (IGUCA) Remediation Study. In the study conducted by the USGS, the hydrologic and chemical interaction of the Arkansas River with the Equus Beds Aquifer was modeled. Steady state and transient simulations were conducted. A steady state model was developed to represent aquifer and stream conditions during the late 1930's. Transient modeling was used to simulate the conditions from 1940 through 1989. Transient modeling was also used for projections beyond 1989.

The U. S. Bureau of Reclamation (USBR), under contract with Equus Beds Groundwater Management District No. 2 (GMD2), modified the USGS model in order to conduct a contaminant transport study (Pruitt, 1993). The purpose of the study was to evaluate the potential for the migration of saline water from the Arkansas River, deep natural saltwater, and brine from the Burrton oil field operations into the Equus Beds Aquifer. To improve the accuracy of the transport modeling, the Bureau of Reclamation refined the model grid spacing and made the grid cells more square-shaped.

In an additional study conducted by the USBR, under contract with the City of Hutchinson, the USGS MODFLOW model was expanded to the west and south and grid spacing was refined. The purpose of this study was to determine the potential impacts of increased pumpage in the Hutchinson area on water quality and supply (Pruitt, 1996). An additional stress period was added to the model to represent the time period from 1990 to 1994.

For the current study, the first USBR transient MODFLOW and contaminant transport models were utilized (Pruitt, 1993). These models were updated to include an additional stress period to represent 1990 to 1994 time period. From this regional model, a subregional model was created for the Burrton IGUCA.

B. GROUNDWATER MODEL

1. <u>Conceptual Model</u>

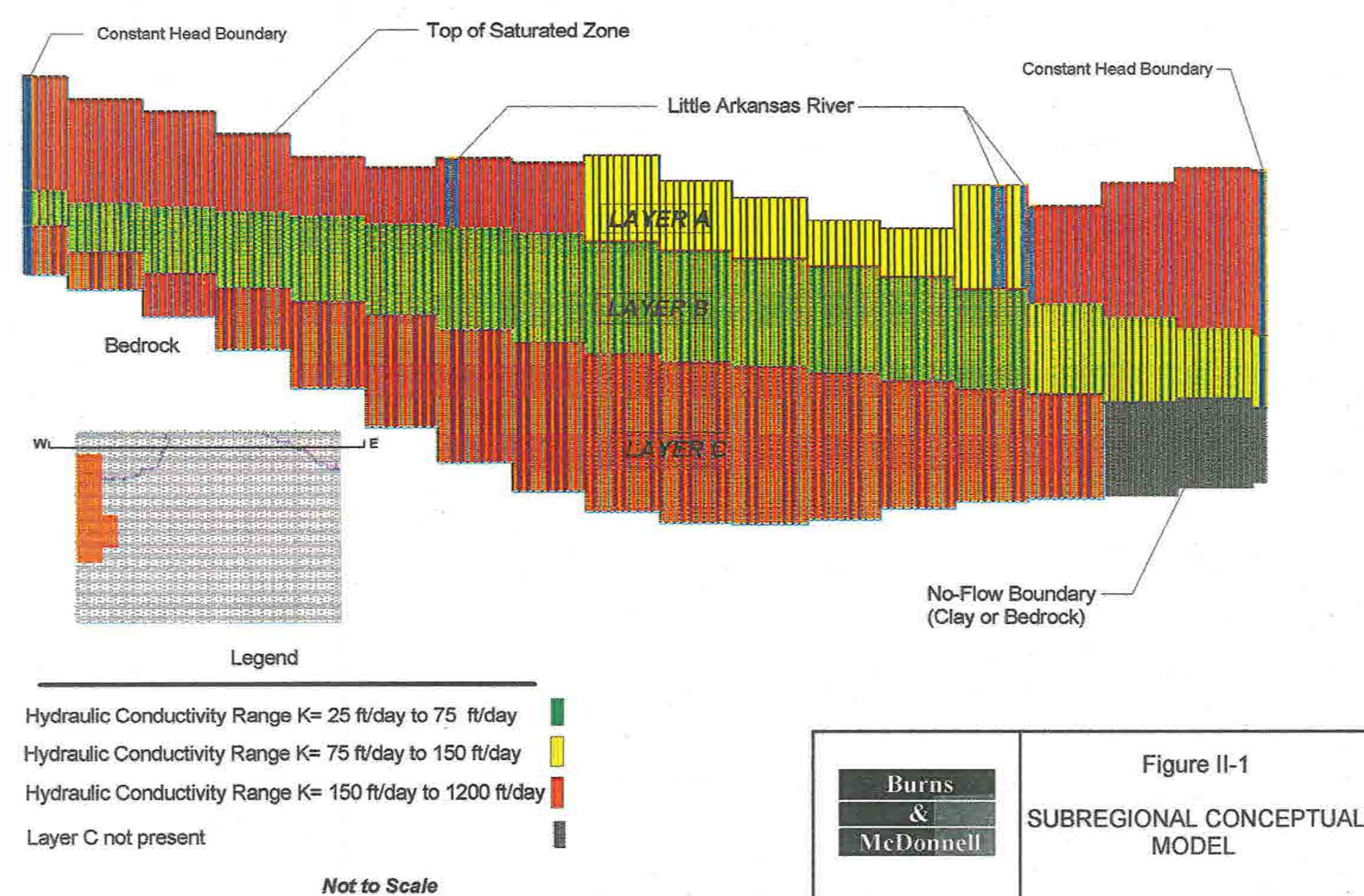
A conceptual model is a block diagram showing how geological conditions are simplified for computer modeling simulations. The Equus Beds Aquifer has three recognized hydrogeologic units. It receives recharge from precipitation, through overlying rivers, and as underflow from surrounding formations. The block diagram in Figure II-1 shows a simplified cross-section of the general aquifer configuration and is the basis for the MODFLOW model construction.

Typically, models are constructed using natural flow boundaries as the boundaries for the model. However, due to the detailed information required for this study, the entire aquifer could not be included. The subregional model utilizes natural boundaries where possible. The Equus Beds Aquifer overlies the Wellington Formation and Ninnescah Shale which have low permeabilities that limit groundwater flow. These natural boundaries are used in the model as a lower boundary and portions of the lateral boundaries. In the areas where the natural aquifer extends beyond the model boundaries, constant head cells were used to simulate the effects of distant parts of the aquifer.

River cells in the top layer of the models were used to simulate the interaction of the groundwater and surface water bodies in the model area. The Arkansas River and the Little Arkansas River and their tributaries are the major water bodies in this area. In the subregional model, only the Little Arkansas River is present.

The aquifer properties used in this model were collected and evaluated for the previous groundwater models. These properties were not modified for this model. Aquifer properties for the models include horizontal and vertical hydraulic conductivity, storage coefficient, and specific yield. Horizontal hydraulic conductivity ranges from 55 to 1,000 feet/day, and vertical conductivities range from 50 to 1,200 feet/day. (Myers, 1996). Storage coefficients range between 0.0004 and 0.16 with specific yields of 0.0006 to 0.22 (Myers, 1996).

Recharge to the Equus Beds Aquifer from precipitation occurs across the study area except where shale outcrops. The amount of recharge from precipitation is



Burns & McDonnell

Figure II-1

SUBREGIONAL CONCEPTUAL MODEL

J:\kccburtn\971834\engineer\models\gwvsub\john\xsec.grf the total precipitation minus surface runoff and evapotranspiration. Mean annual recharge values for the area range from 0.44 to 6.02 inches (Myers, 1996).

Like recharge, discharge through evapotranspiration occurs across the study area. The two components of evapotranspiration are discharge from the unsaturated zone and phreatophytic consumption of the saturated zone. The recharge values set for the regional model allow for evapotranspiration from the unsaturated zone. Phreatophytic consumption of the saturated zone is simulated separately (Myers, 1996).

Pumpage is a major source of discharge from the Equus Beds Aquifer. Groundwater is pumped from the aquifer for irrigation, municipal, and industrial use. The average groundwater withdrawal from the study area through wells during 1990 to 1994 was approximately 31,920 acre-feet/year.

2. Regional Model

For this study, the primary area of interest is the Burrton IGUCA. This area is included in the USBR model, however the grid spacing in this area is too large to provide a detailed evaluation of the migration of chlorides in the Burrton area. In addition, the model is only current through 1989. In order to operate with current conditions, Burns & McDonnell updated the USBR model to include the 1990 to 1994 time period. An additional stress period was created to represent this time period. In the MODFLOW model, the stream, recharge, and well packages were updated with information for this time period. In addition, current chloride concentrations in the Arkansas River were included in the transport model.

In the USGS model, recharge was a function of 1940 precipitation, soil type, and thickness of clay in the unsaturated zone. The same recharge distribution was maintained for the new 1990 to 1994 stress period of the regional model. The precipitation data for 1990 through 1994 was obtained from the Hutchinson, McPherson, Newton, and Wichita climatic stations and from the Reno, McPherson, Harvey, and Sedgwick GMD2 climatic stations. The recharge rates were determined from this data by following the method used to determine the rates for the original USGS model (Myer, 1996). First, the recharge for each steady state cell was divided by the mean annual precipitation for the steady state time period (1995 to 1939). The resulting values were multiplied by the mean annual precipitation for the 1990 to 1994 time period for the model area. The mean annual precipitation for this period for the regional model area was 28.1 inches per year.

Discharge from the aquifer through wells for the regional model area was adjusted from the 1990 to 1994 pumpage data used for the second USBR model (Pruitt, 1996). The data set was modified to include only the wells in this study area and redistributed for the grid geometry of the original USBR model.

The base flow was updated for the starting reach of the Arkansas River and Little Arkansas River. The stream routing module (Prudic, 1988) calculated the stream flows for the downstream cells. The stream flow for the Arkansas River was determined from data collected at the gaging station on the Arkansas River near Hutchinson for the 1990 to 1994 period. Data for the Little Arkansas River was obtained from the Little Arkansas River at Valley Center for the same time period. The original USGS model assumed base flow to be the stream flow that was exceeded 70 percent of the time. This assumption was applied to the stream flow data for the 1990 to 1994 period. Stream flows for the starting reaches of the Arkansas and Little Arkansas Rivers were determined by trial and error until the stream flows simulated by the stream routing model approximated the base flow at the gages at Hutchinson and Valley Center.

A comparison between water budgets and predicted heads from the updated USBR model was used to evaluate the modifications and updates of the regional model used for this study. The predicted heads for both models are similar and the water budget for the study's regional model has volumetric water budget discrepancy of 0.01 percent. Volumetric water budget discrepancy less than 1 percent is generally adequate for modeling studies.

3. Subregional Model

In order to provide the necessary detail with current conditions, Telescopic Mesh Refinement (TMR), a modeling technique, was used to create a subregional model with finer grid spacing for the Burrton IGUCA study area. Figure I-1 shows the location of the boundary for the subregional model along with the boundary for the regional model.

A uniform grid spacing was used across the model to give good resolution of all parts of the study area. Row spacing was set at 508 feet and column spacing at 500 feet to provide adequate resolution for the study. The boundary conditions for the subregional model represent zones of no flow and the regional groundwater and surface water flow conditions for 1994. Starting groundwater and surface water flow conditions for the subregional model were determined by the regional model. Figure II-2 shows the boundary conditions for the subregional model.

During the development of the subregional model, the stream cells of the regional model were converted to river cells and restricted to cells that overlie the Little Arkansas River. The values for river stage were determined from the regional model.

Recharge, evapotranspiration, and the aquifer properties were not modified from the regional model for the development of the subregional model.

C. CONTAMINANT TRANSPORT MODEL

1. Subregional Model

MT3D (Papadopulos and Associates, Inc., 1992), a modular three-dimensional transport model, was used to predict the migration of chlorides in the Equus Beds Aquifer. The regional contaminant transport model developed by the Bureau of Reclamation (Pruitt, 1993) was used to develop the subregional model used for this study. The subregional contaminant transport model focuses on the chloride plume in the Burrton oil field area. The migration of chlorides was evaluated under conditions of no action and with several alternatives to control and remediate the plume.

The MT3D program is used to model advection, dispersion, and chemical reactions of contaminants in groundwater, and is a companion program to MODFLOW. Output from the groundwater flow simulations is used in the contaminant transport modeling.

A detailed description of the MT3D program and the assumptions for applying the model to the migration of chlorides in the Equus Beds Aquifer is contained in the USBR Technical Report on modeling chlorides in the Equus Beds Aquifer (Pruitt, 1993).

2. Boundary Conditions and Initial Concentrations

The boundary conditions established in the groundwater flow model were maintained for the contaminant transport model. Data obtained from GMD2 and from previous modeling (Pruitt, 1996) was used to establish the initial concentration of chloride for the study area. Chloride associated with the Burrton

IGUCA was the main focus of the model. Figures I-2, I-3, and I-4 show the concentration of chloride present in the aquifer in 1996. This chloride distribution was used as the initial concentration condition for all model simulations.

The impacts of chlorides migrating from the Arkansas River and the deep natural saltwater were also considered. The Arkansas River provides a continuous source of chloride into the aquifer. Constant concentration cells located along the southern boundary of the top layer of the model represent the influx of chlorides from the Arkansas River into the model area.

Like the Arkansas River, the deep natural saltwater migrating from the Wellington Formation is a continuous source of chloride. Constant concentration cells at the southern boundary of the deepest layer simulate the intrusion of deep saltwater from the bedrock into the study area.

D. MODELING SCENARIOS

The modeling runs for the management simulations were established for a 50 year time period beginning with the 1996 conditions. The stresses and chloride concentrations from the Arkansas River and deep aquifer were assumed to remain constant during the projected time period. The changes in water level demonstrates the impact that pumpage and recharge has on the aquifer. The base scenario assumes that no action is taken to remediate or control the chloride plume for the 50 year time period. The results from the plume management simulations were compared with the results from base (no action) scenario.

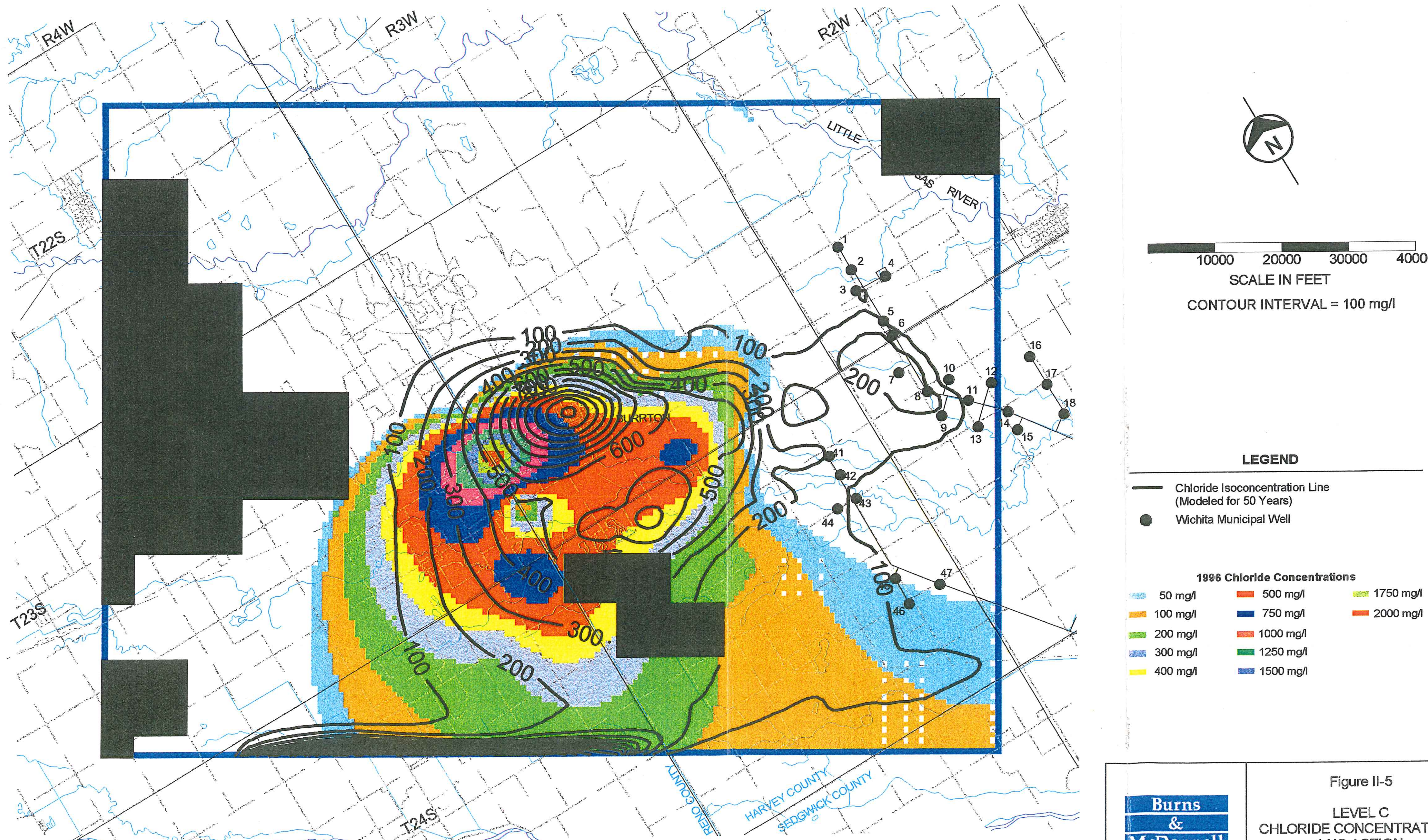
Management simulations conducted for this study are described in the following Table II-1.

Table II-1
MANAGEMENT SIMULATIONS

Run	Management Action
1a	**No Action.** 1996 stresses and conditions held constant for 50 years.
Plume Interception	
2a	Reproduced from the USBR study (Pruitt, 1993). Ten wells screened in levels B and C with a total pumping rate of 1000 gallons per minute (gpm).
2b	Reproduced from the USBR study (Pruitt, 1993). Ten wells screened in levels B and C with a total pumping rate of 2000 gpm.
2c	Reproduced from the USBR study (Pruitt, 1993). Ten wells screened in levels B and C with a total pumping rate of 4000 gpm.
Plume Control	
3a	Six gradient control wells and 17 extraction wells with a total pumping rate of 1000 gpm.
3b	Six gradient control wells and 17 extraction wells with a total pumping rate of 2000 gpm.
3c	Six gradient control wells and 17 extraction wells with a total pumping rate of 4000 gpm.
Plume Control with Recharge Basins	
4a	Six gradient control wells and 17 extraction wells with a total pumping rate of 4000 gpm. 75% of pumpage from the extraction wells (1500 gpm) recharged downgradient of the plume.
Equus Beds Recharge Project	
5a	Water levels five feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project.
5b	Water levels ten feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project.
5c	Water levels ten feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project with active injection into level C through 22 wells at a total rate of 8,530 gpm.

Table II-_ (continued)
MANAGEMENT SIMULATIONS

Run	Management Action
Equus Beds Recharge Project with Plume Control	
6a	Water levels five feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project, and Six gradient control wells and 17 extraction wells with a total pumping rate of 4000 gpm.
6b	Water levels ten feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project, and Six gradient control wells and 17 extraction wells with a total pumping rate of 4000 gpm.
6c	Water levels ten feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project with active injection into level C through 22 wells at a total rate of 8,530 gpm, and Six gradient control wells and 17 extraction wells with a total pumping rate of 4000 gpm.
6d	Water levels ten feet higher in the vicinity of the Wichita Well System due to recharge effects from the Equus Beds Recharge Project with active injection into level C through 22 wells at a total rate of 8,530 gpm, and Six gradient control wells and 17 extraction wells with a total pumping rate of 8000 gpm.
Pilot Installation	
7a	Two wells screened in levels A and B pumping at a total rate of 500 gpm.



Burns & McDonnell

Figure II-5

LEVEL C
CHLORIDE CONCENTRATIONS
1 NO ACTION
FOR 50 YEARS

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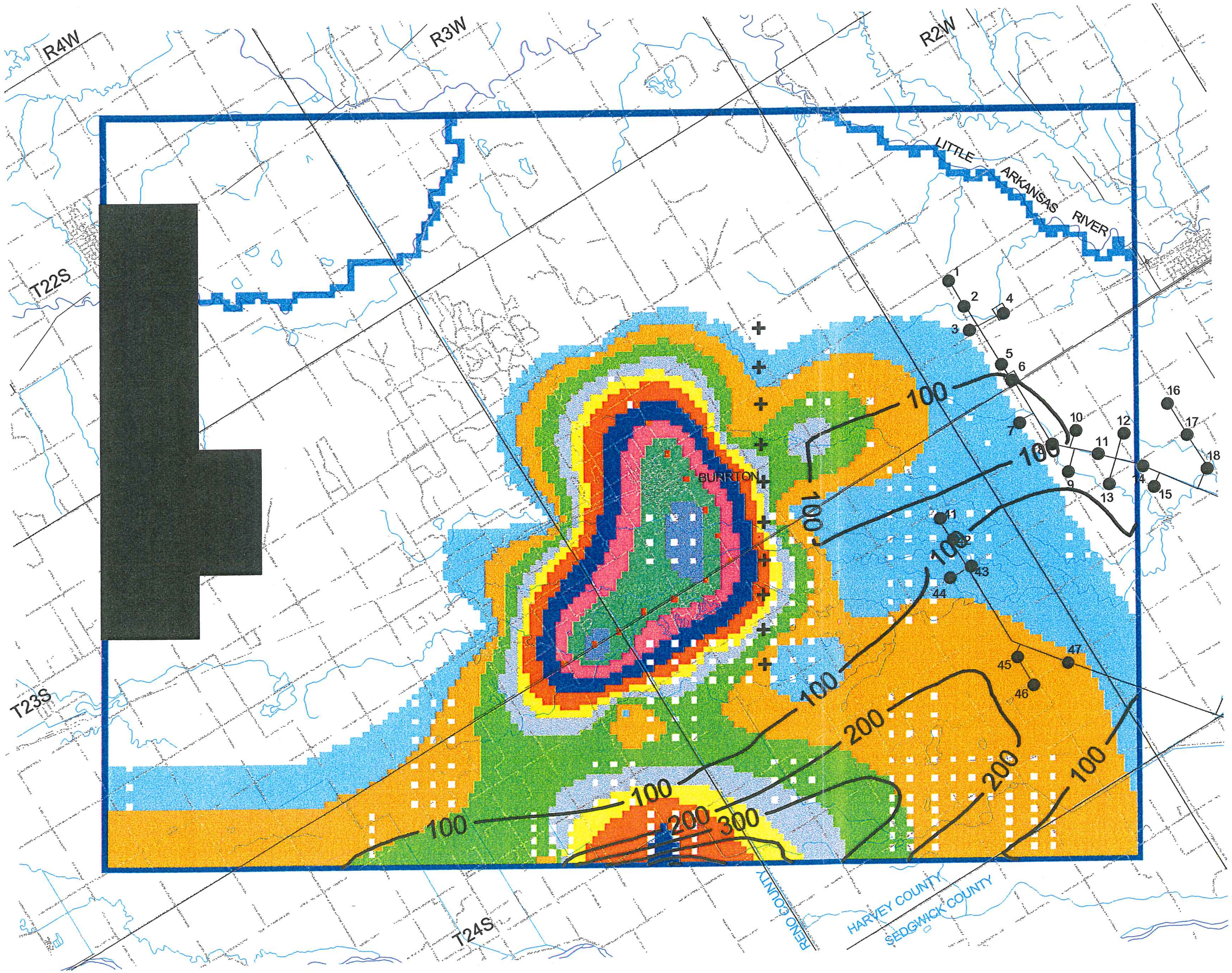
Table II-2

Total Land Area
above
High Chloride Levels in Groundwater
(square miles)

Scenario	Level A	Level B	Level C
Current (measured 1996 concentrations)			
250 mg/L	18.8	20.8	16.7
500 mg/L	9.8	13.5	3.9
Modeled Simulations			
After 50 years (500 mg/L)			
1 - no action	-	2.9	11.8
2c - USBR interceptor wells at 4,000 gpm	-	0.5	8.1
3c - Plume Control (extraction and gradient control wells)	-	1.7	7.7
4a - Plume Control with recharge basins	-	1.6	8.6
5c - Equus Bed Recharge Project with no plume control	-	3.8	11.7
6c - Equus Beds Recharge Project with 4,000 gpm plume control	-	2.0	8.5
6d - Equus Beds Recharge Project with 8,000 gpm plume control	-	1.3	6.3



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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

Chloride Isoconcentration Line (Modeled for 50 Years)

Wichita Municipal Well

Interceptor Well (Modeled by Pruitt, 1993)

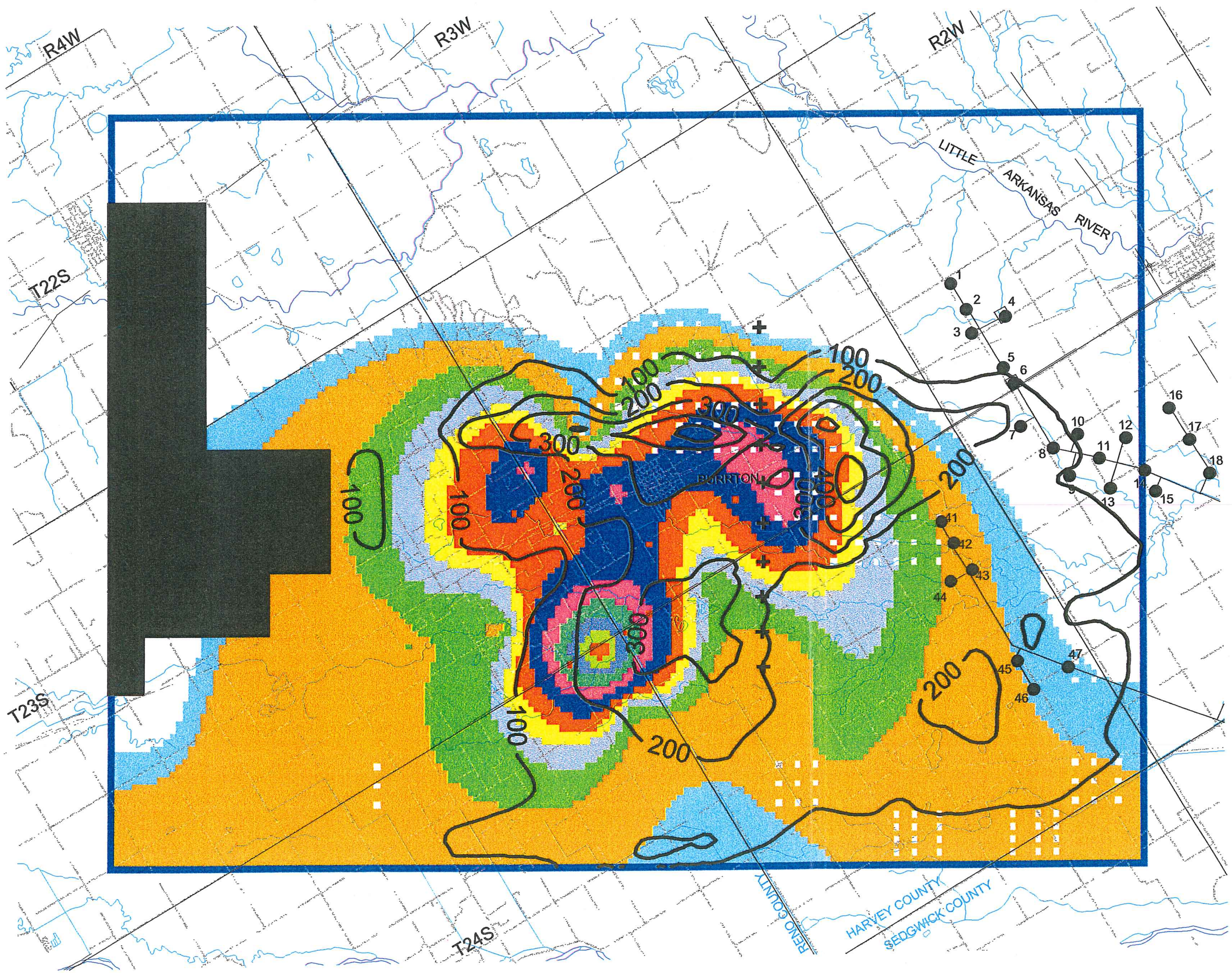
1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

Burns & McDonnell

Figure II-7

LEVEL A
CHLORIDE CONCENTRATIONS
2C PLUME INTERCEPTION
4,000 GPM FOR 50 YEARS



N

10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

Chloride Isoconcentration Line (Modeled for 50 Years)

Wichita Municipal Well

Interceptor Well (Modeled by Pruitt, 1993)

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-8

LEVEL B
CHLORIDE CONCENTRATIONS
2C PLUME INTERCEPTION
4000 GPM FOR 50 YEARS

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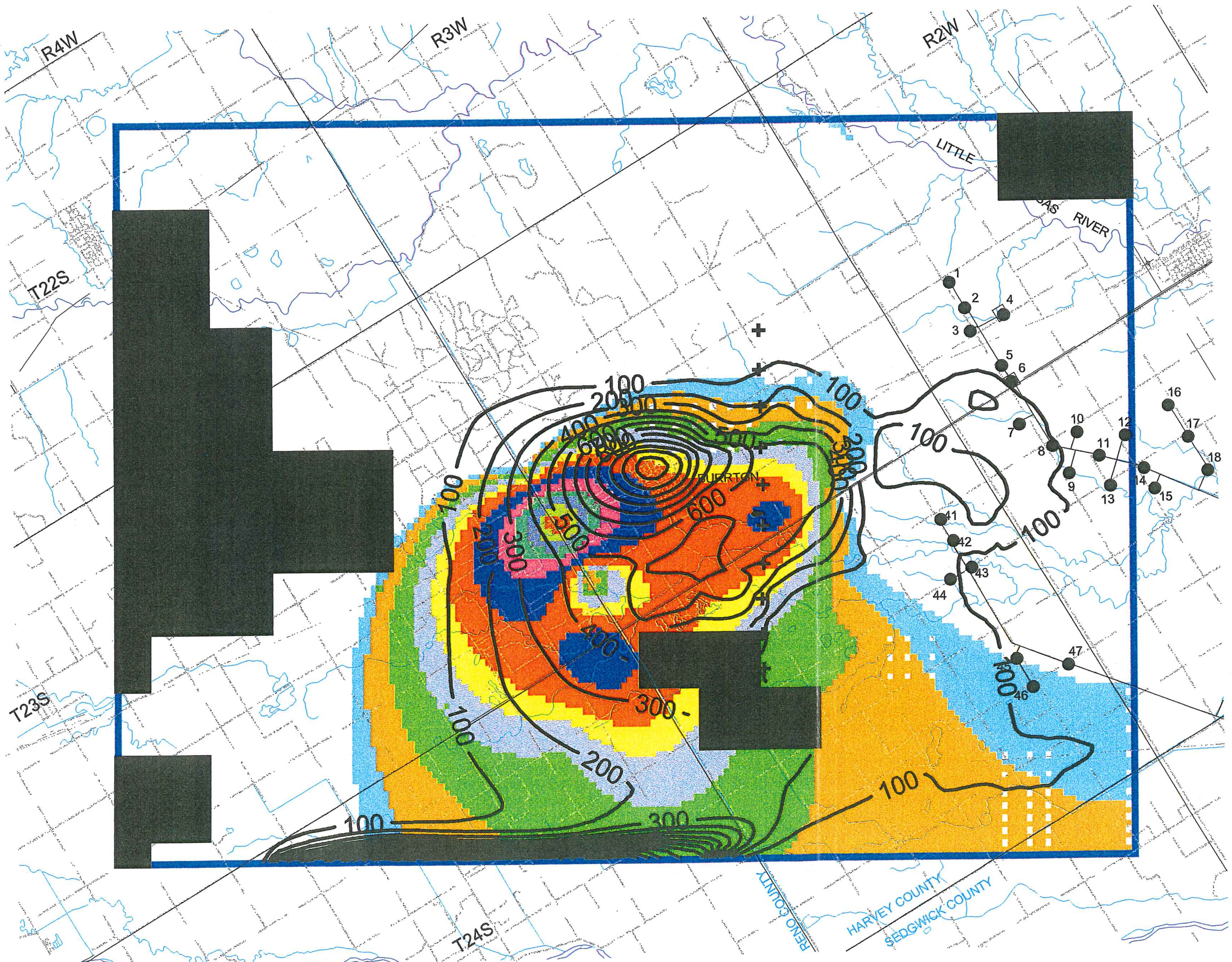

10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- Interceptor Well (Modeled by Pruitt, 1993)

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-9

LEVEL C
CHLORIDE CONCENTRATIONS
2C PLUME INTERCEPTION
4000 GPM FOR 50 YEARS

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With the USBR simulation well layout, the area impacted by chloride concentrations exceeding 500 mg/L is reduced to 0.5 and 8.1 square miles in levels B and C, respectively. The use of the intercepter wells also reduces the impact of the chloride plume on the Wichita Well Field.

Over the 50 year time period, the MT3D modeling predicts that approximately 175,000 tons of chlorides as salt (NaCl) will be removed by the interceptor wells pumping at a total rate of 4,000 gpm for the management simulation 2c.

Table II-3 lists selected management simulations and shows the effects each scenario on the predicted chloride balance within the model area. The second column of the table shows the mass of brine removed from the aquifer by the remediation wells (interceptor wells, extraction wells, and/or gradient control wells) in terms of salt (NaCl). The table also lists the amount of salt induced into the model area in response to the remediation pumping. The source of the salt (chlorides) in-flow to the model area is either the Arkansas River and/or the deep Permian saltwater located in the bedrock channel under the Arkansas River.

Graphs of the chloride concentrations over the 50 year model period were created for simulated well No. 5 which is centrally located in the line of interceptor wells. The simulated well withdraws water from the middle and lower layers of the aquifer (Figures II-10 and II-11). In layer B, the high concentration zones of the plume migrate past the well, and in layer C, the lowest layer, the high concentration zones remain in the vicinity of the well. In comparison with the no action scenario, the layer B chloride concentrations are reduced by approximately 200 mg/L. However, in layer C, these wells have very little impact on the chloride concentrations.

Table II-3

SUMMARY OF MODEL SALT BUDGET[1]
(Tons of salt)

Management Simulation	Salt Removed by Extraction and Gradient Control Wells	Salt Entering Study Area		Net Salt Removed from Study Area		Salt Prevented from Entering Little Ark. River
		Induced from River and Bedrock	Recharge[3] Wells	Tons	Percent[2]	
2c Plume Interception 4,000 gpm	174,400	57,500		116,900	33	1,900
3c Plume Control 4,000 gpm	131,600	132,100		(500)	100	1,500
5a Equus Beds Recharge Project	-	(70,700)	4,200	66,500		(1,400)
6c Equus Beds Recharge Project with Plume Control, 4,000 gpm	133,900	9,000	4,200	120,700	7	1,500
6d Equus Beds Recharge Project with Plume Control, 8,000 gpm	257,000	107,100	4,200	145,700	42	3,200
7a Pilot 500 gpm	31,100	10,300	-	20,800	33	400

Notes:

1. Comparison of no action and management simulations for 50 years of operation.
2. Comparison of salt entering the study area with salt removed by the management simulation.
3. Assumes 60 mg/l chloride in recharge water.

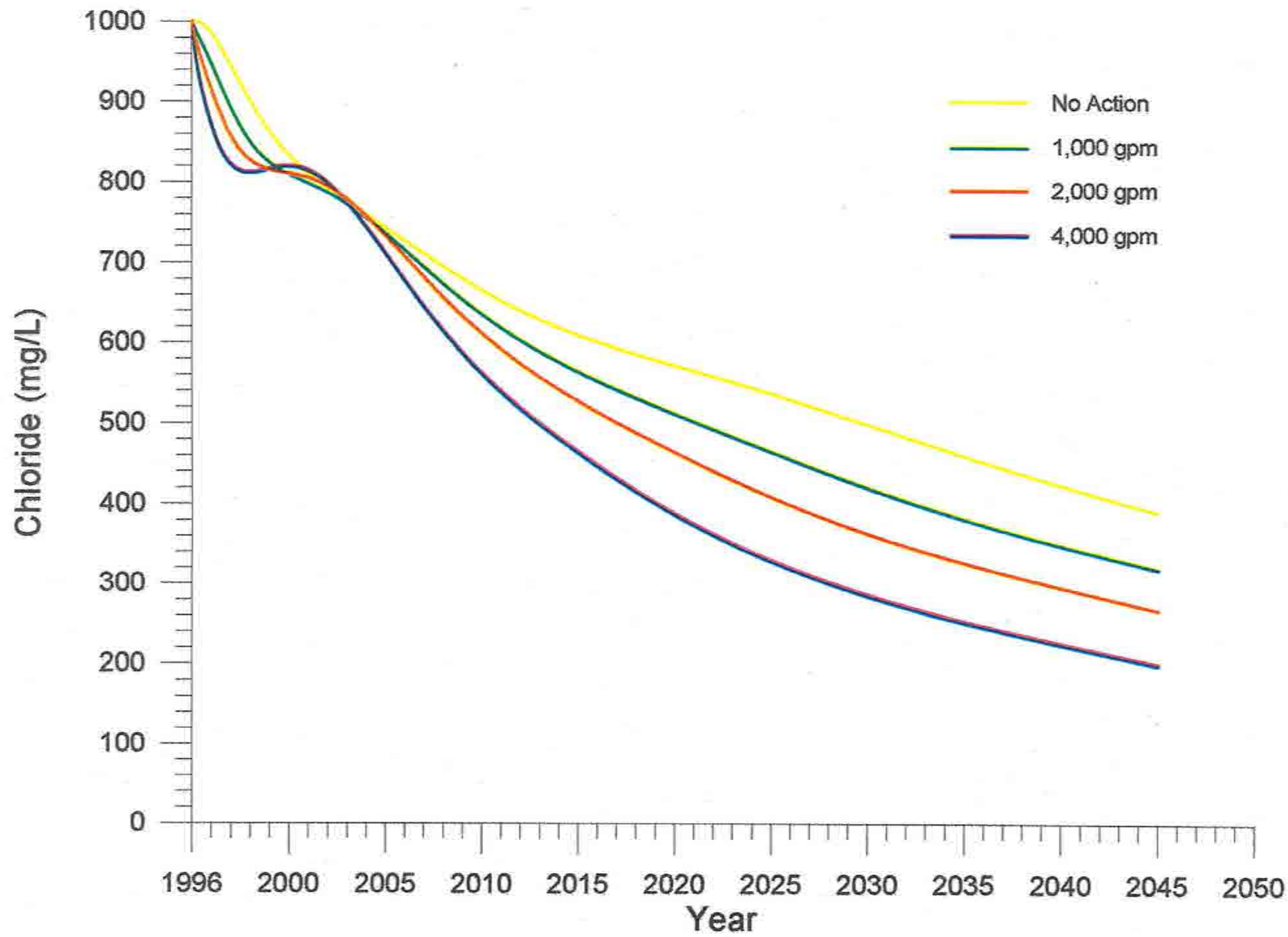

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Figure II-10

USBR INTERCEPTOR WELLS
CHLORIDE CONCENTRATIONS
WELL 5 - LEVEL P

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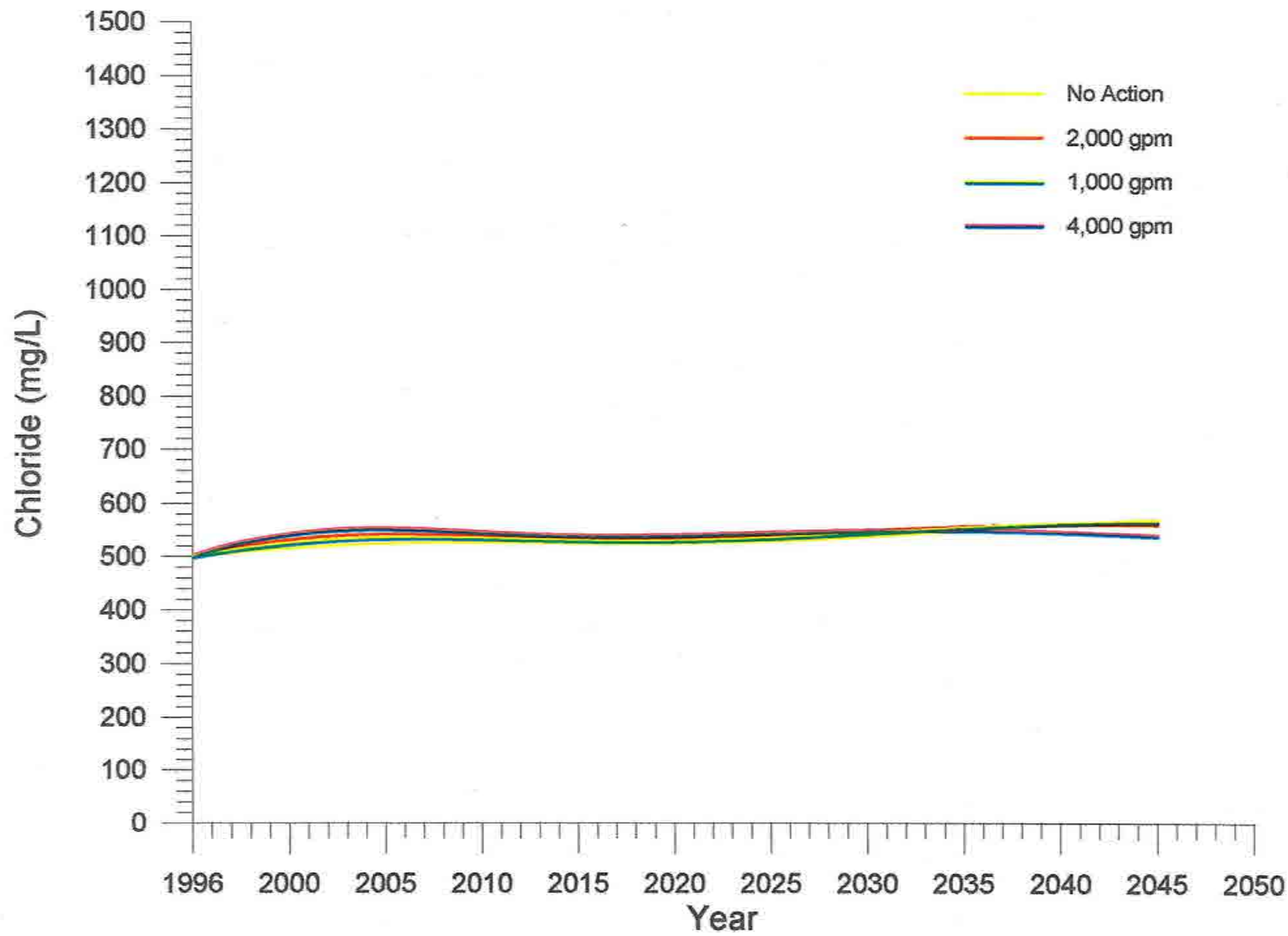

Burns & McDonnell

Figure II-11

USBR INTERCEPTOR WELLS
CHLORIDE CONCENTRATIONS
WELL 5 - LEVEL C

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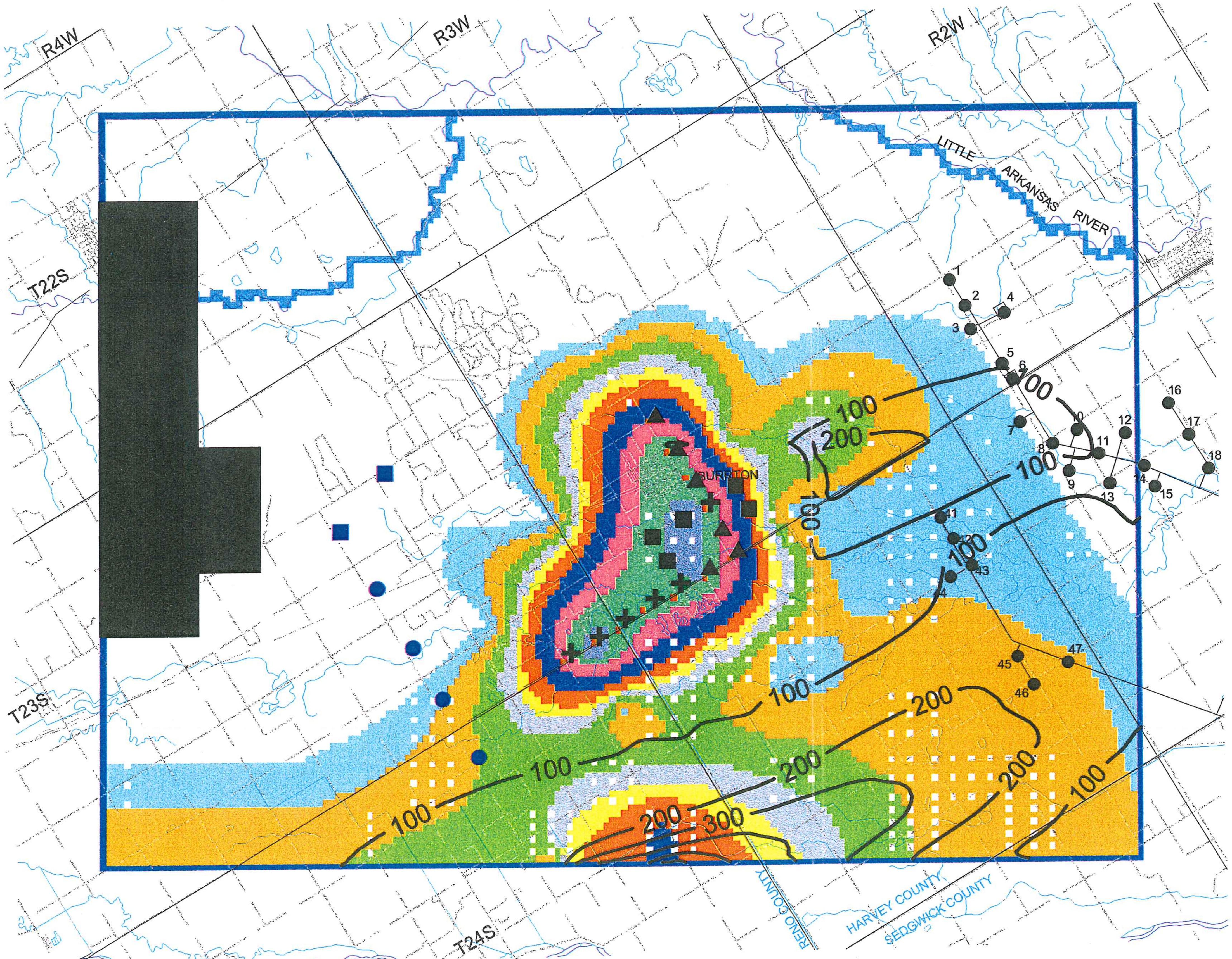
3. Plume Control (3a, 3b and 3c)

Based on the results of the no action and interceptor well scenarios, an initial withdrawal well layout was established. This layout combined the use of extraction wells located in the high concentration zones of the chloride plume and higher capacity wells upgradient of the plume. The low-capacity extraction wells are designed to remove groundwater from areas of highest chloride concentrations within hydrostratigraphic levels A and B. The gradient control wells are designed to control the local groundwater flow gradient restricting the migration of the high chloride plume within hydrostratigraphic Levels A, B and C. Plume migration control allows greater removal efficiency of high chloride concentrations by the extraction wells. The well layout is shown in Figure II-12.

Seventeen extraction wells were located in the high concentration zones of the plume with four wells screened into Level A, two wells screened into Level B, and 11 wells screened into both hydrostratigraphic units. Where feasible, wells were screened into both units to control costs by minimizing the number of wells constructed. The capacity of these wells are selected at a maximum of 87 gallons per minute (gpm) or 174 gpm, for wells screened into one or two units, respectively. The relatively low capacity of these wells was selected to prevent pulling clean water from outside of the contaminated area, thus requiring greater treatment and disposal volumes.

Six gradient control wells were placed upgradient of the main part of the plume in an area of moderate levels of chloride concentration. These wells are about two miles upgradient of the series of extraction wells to limit movement of the chloride plume. Four wells were screened in Levels B and C with two wells screened in Level B only. The maximum pumping rates for these wells were selected as 400 gpm for wells screened in two layers and 200 gpm for wells screened in a single layer.



N

10000 20000 30000 40000
SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

Burns & McDonnell

Figure II-13

LEVEL A
CHLORIDE CONCENTRATIONS
3C PLUME CONTROL
4,000 GPM FOR 50 YEARS

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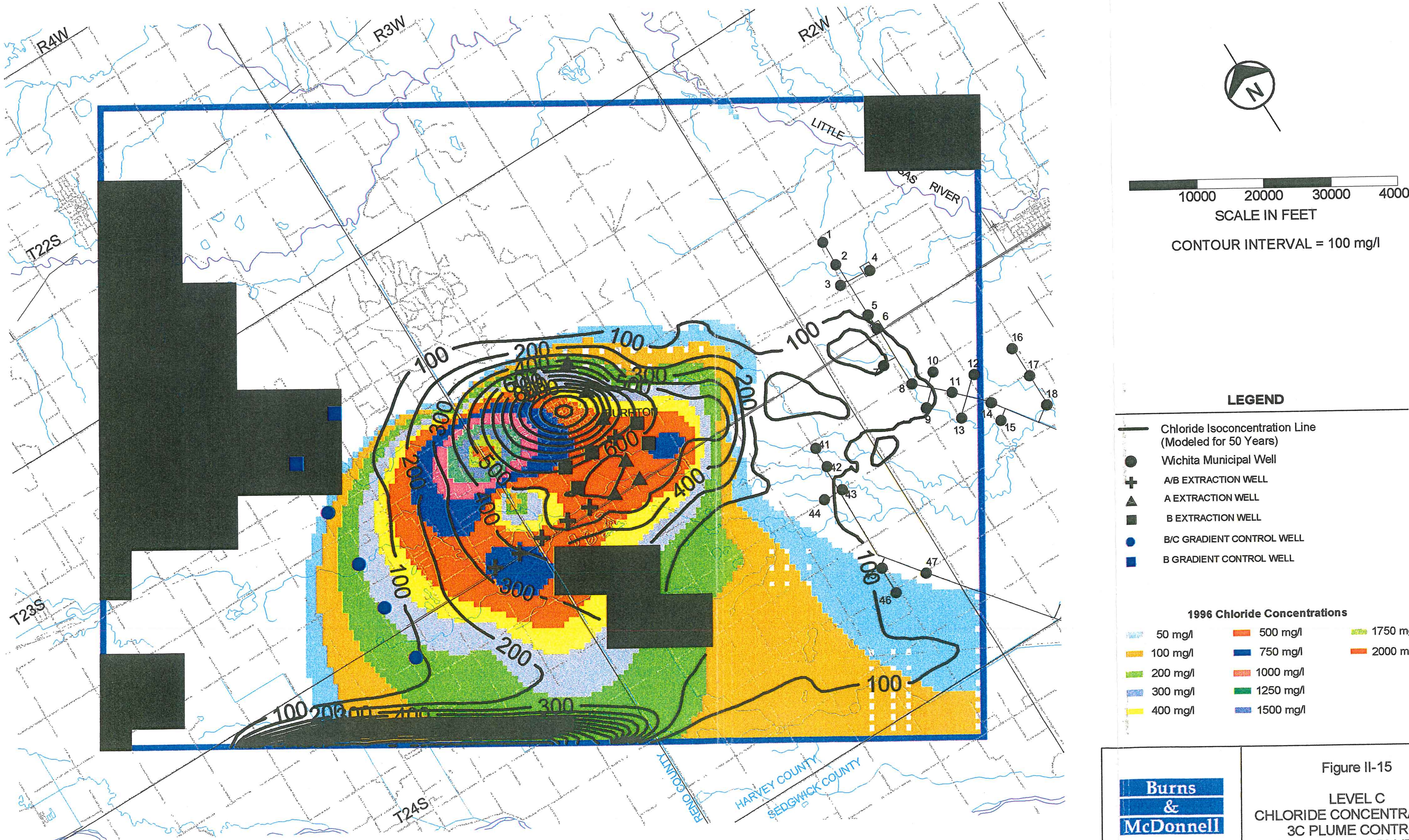

Burns & McDonnell

Figure II-15

LEVEL C
CHLORIDE CONCENTRATIONS
3C PLUME CONTROL
4000 GPM FOR 50 YEARS

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The total pumpage for the extraction wells and gradient control wells is 4000 gpm selected to reflect the pumpage modeled by the USBR (simulation 3c). Additional scenarios used total pumping rates of 2000 gpm and 1000 gpm (simulations 3c and 3b). Pumping rates for individual wells were reduced by 50% and 75% for these simulations. Figures II-13 through II-15 show the predicted chloride concentrations for management simulation 3c (4000 gpm for 50 years). Like the interceptor wells modeled by the USBR, these wells do not prevent the plume from affecting the Wichita Well Field, however, the use of this layout of wells lessens the impact of the chloride plume on the well field. As shown in Table II-2, the surface area impacted by chloride concentrations exceeding 500 mg/L is reduced to 1.7 and 7.7 square miles in levels B and C, respectively. These modeling results indicate that this scenario is less effective in level B and more effective in level C when compared to the USBR interceptor well layout pumping at equivalent rates.

Figures II-16 and II-17 shows the chloride concentrations over time for gradient simulated control well no. 3, located in the heart of the plume, as predicted by the contaminant transport model. The initial chloride concentrations range from about 100 mg/L for groundwater from Layer B to 200 mg/L for Layer C. After 50 years, the concentrations drop to about 20 mg/L and 30 mg/L, respectively. Initially, these concentrations are greater than the ambient chloride levels ranging up to 75 mg/L for the Equus Beds Aquifer. However, they are much less than the concentrations of the discharge water from the USBR interceptor wells.

The MT3D modeling estimated that approximately 132,000 tons of chlorides (as salt) are extracted over the 50 year period with management simulation 3c (4,000 gpm). However, because of the response of the hydrogeologic system, additional chloride is introduced into the model area because of the changing groundwater gradient. With this management simulation, there is an equivalent amount of

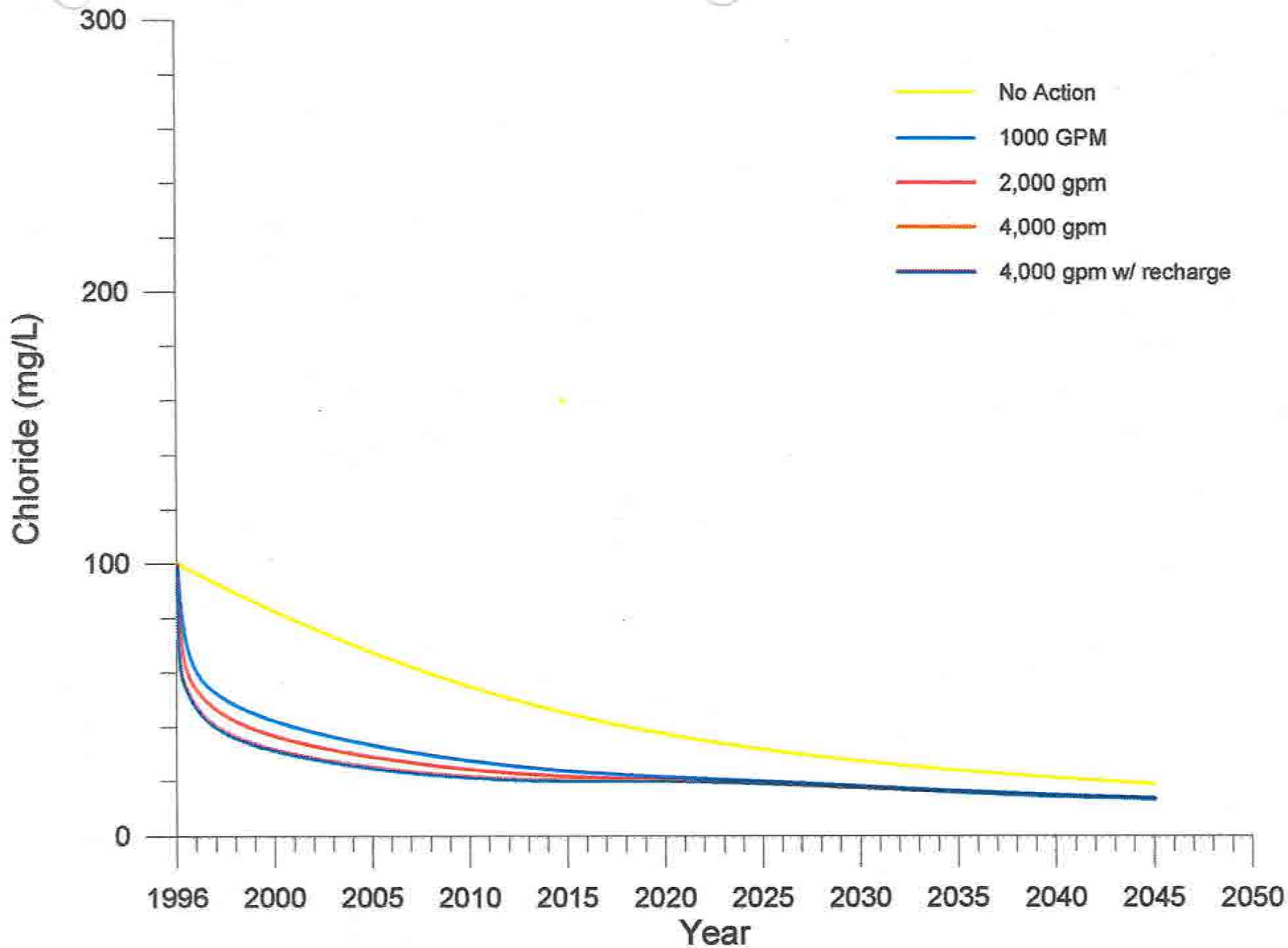

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Figure II-16

CHLORIDE CONCENTRATIONS
GRADIENT CONTROL WELL 3
LEVEL B

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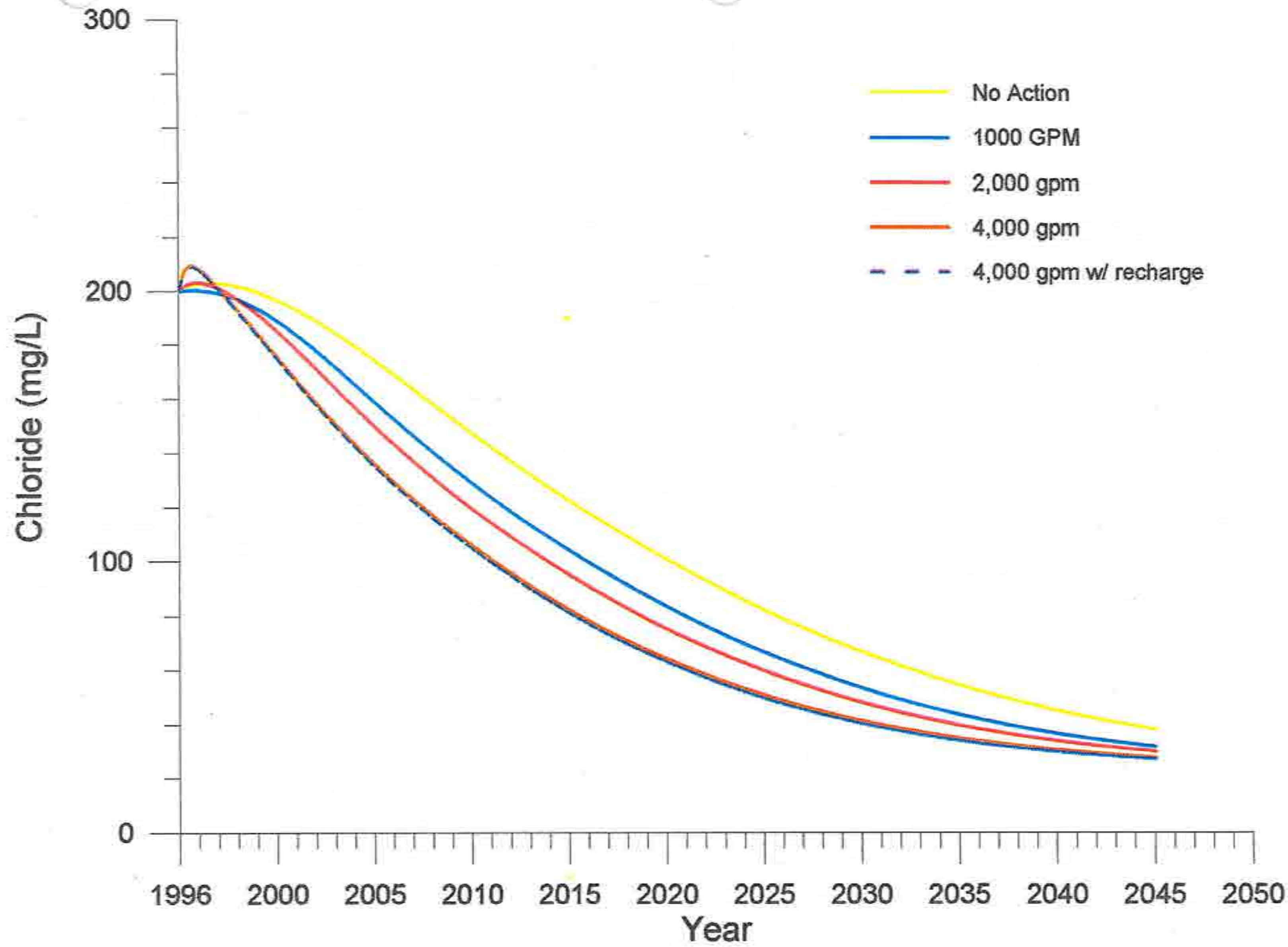

Burns & McDonnell

Figure II-17

CHLORIDE CONCENTRATIONS
GRADIENT CONTROL WELL 3
LEVEL C

J:\KCCBURTN\971834\ENGINEER\MODELS\GWVSUB\CONC\GC3GC.GRF chlorides pulled into the model area from the south as was extracted with the remediation wells resulting in very little net improvement in the study area.

4. Plume Control with Recharge Basins (4a)

Both treated water and pumpage from the gradient control wells can potentially be recharged into the Equus Beds Aquifer downgradient of the extraction well system. A simulation was conducted using management simulation 3c (4000 gpm) combined with four recharge basins (Figure II-18). The total recharge rate was selected to be 75 percent of the water pumped with the extraction wells, assuming that 25 percent is lost with reverse osmosis water treatment. The recharge water was assumed to possess a maximum chloride level of 150 mg/L and be recharged at a rate of 375 gpm in each of four basins. The recharge basins were located along the downgradient 200 mg/L chloride concentration contour. Figures II-19 through II-21 show the chloride concentrations after 50 years for this simulation. The addition of the recharge basins provides some additional control over the plume, however, like the previous scenarios, the impacts at this pumping rate are limited.

5. Plume Control Impacts of the Equus Beds Groundwater Recharge Project (5a, 5b and 5c)

The City of Wichita has undertaken a project to recharge the city's Equus Beds Well Field with above-base flow water from the Little Arkansas River. Management simulations were conducted to allow for the effects from the Equus Beds Recharge Project. Modeling scenarios were set up with the assumptions of the water levels rising five (5a) and ten feet (5b) along the eastern portion of the study area. Scenarios assuming that no action was taken to control or remediate the plume were conducted to assess the natural movement of the plume under large scale recharge conditions.

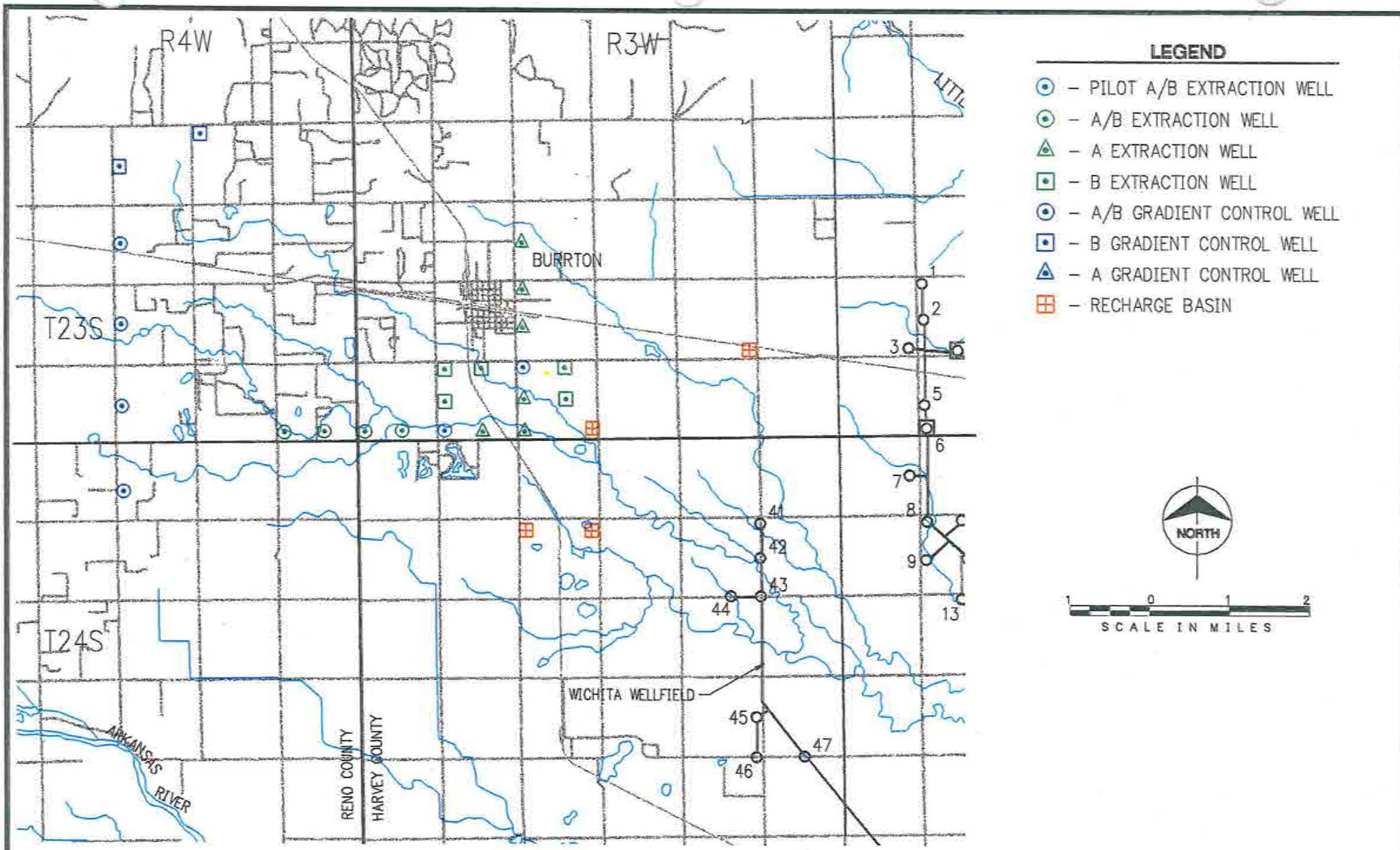

Figure II – 18

3 PLUME CONTROL
WITH RECHARGE BASINS
WELL AND BASIN LOCATIONS

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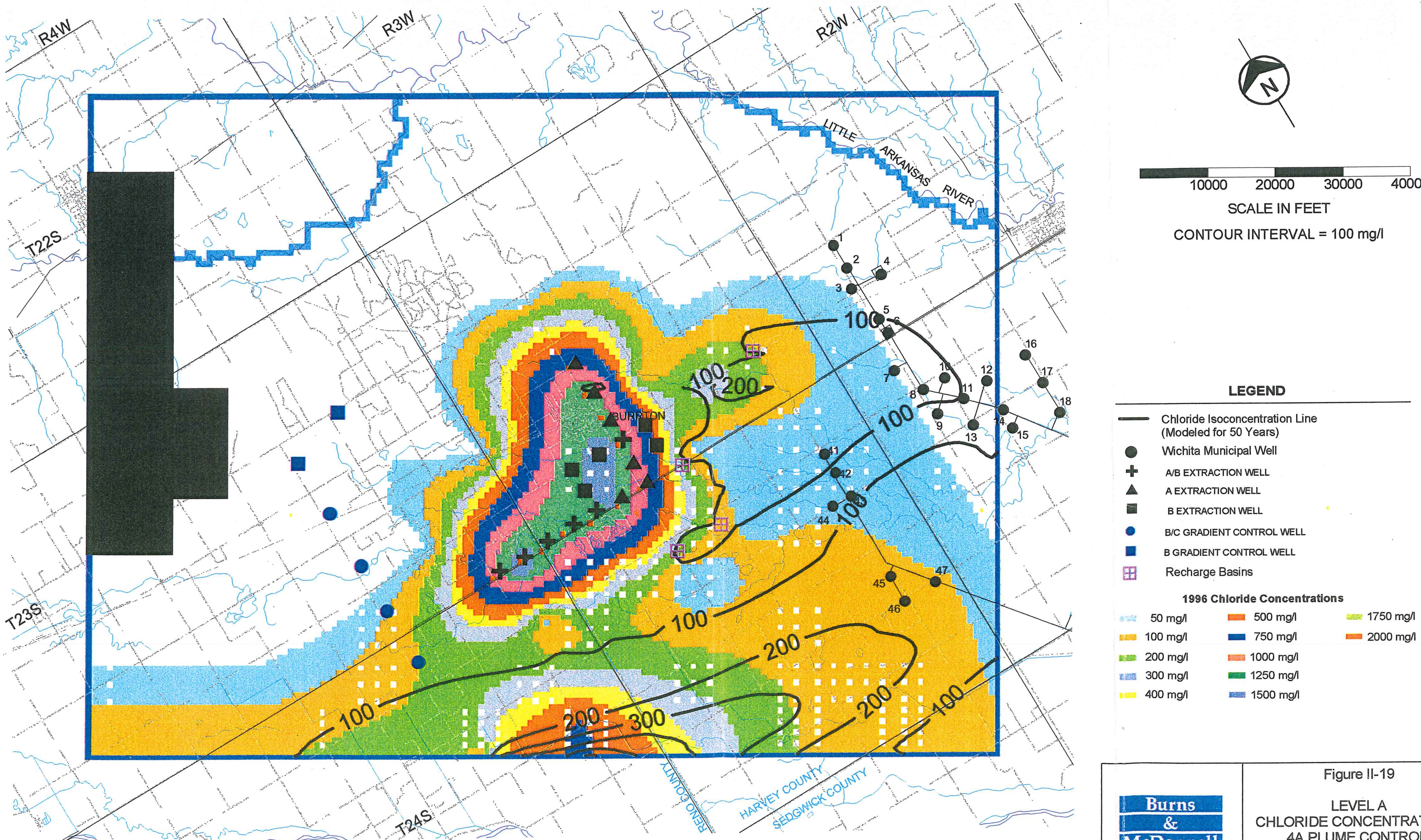

Burns & McDonnell

Figure II-19

LEVEL A
CHLORIDE CONCENTRATIONS
4A PLUME CONTROL
WITH RECHARGE BASINS
4,000 GPM FOR 50 YEARS

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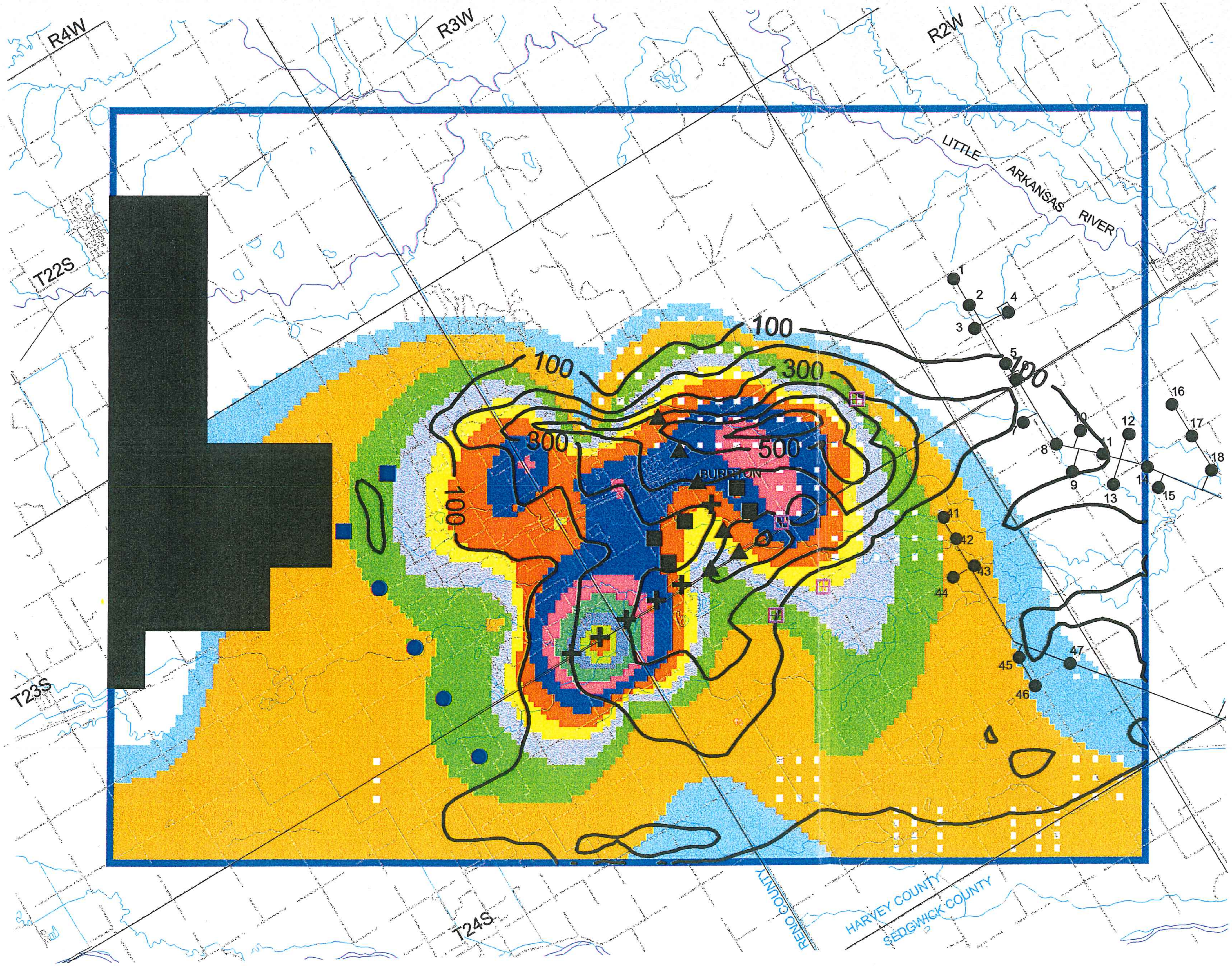

10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- Recharge Basins

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-20

LEVEL B
CHLORIDE CONCENTRATIONS
4A PLUME CONTROL
WITH RECHARGE BASINS
4000 GPM FOR 50 YEARS

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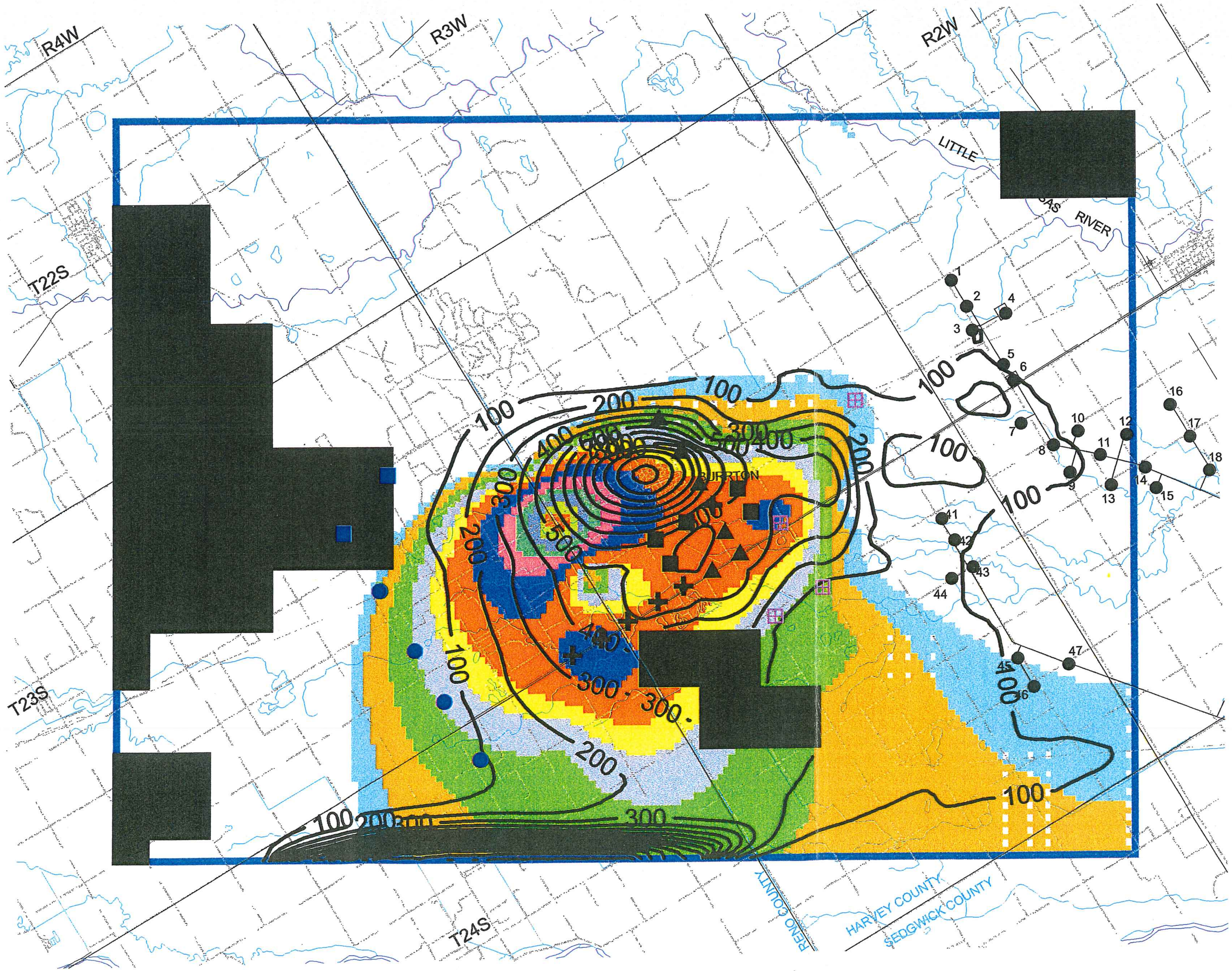

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- Recharge Basins

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-21

LEVEL C
CHLORIDE CONCENTRATIONS
4A PLUME CONTROL
WITH RECHARGE BASINS
4000 GPM FOR 50 YEARS

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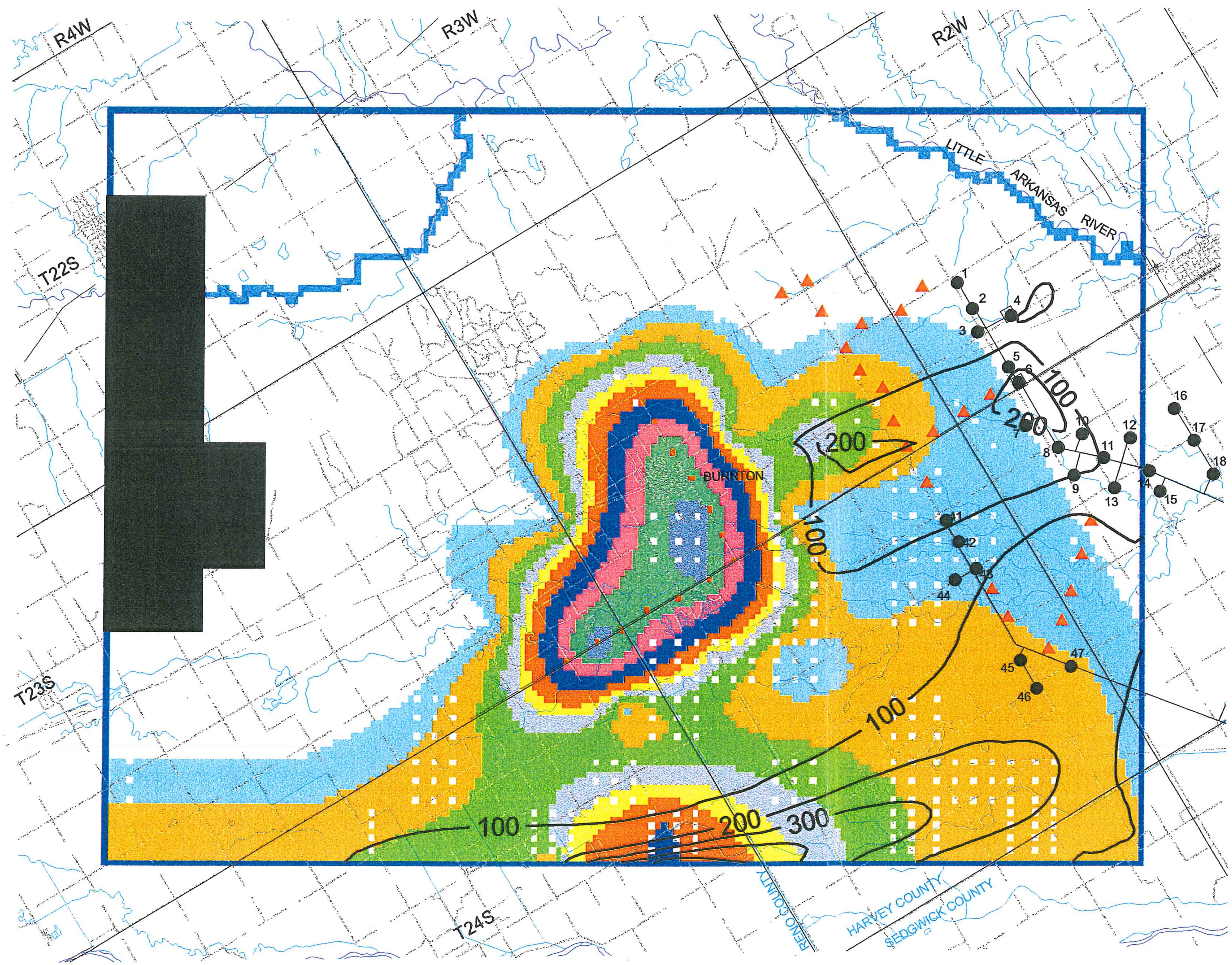
After reviewing the results from these scenarios, an additional scenario (5c) was performed using recharge wells as conceptually proposed for the Equus Beds Aquifer recharge operation. Simulation 5c assumes 22 wells, representing Equus Beds recharge wells, were operating at an average rate of 8,530 gpm. This rate is based on the anticipated average annual volume of water that is expected to be recharged during full scale operations. This rate is estimated to be approximately 627 acre-feet/year in the model area. The recharge wells were assumed to he screened and injecting water in level C.

This scenario was modeled for 50 years assuming no action was taken to control or remediate the Burrton chloride plume. Figures II-22 through II-24 show the chloride concentrations for this simulation. The results of modeling the full scale Equus Beds Recharge Project show a significant effect on the migration of the chloride plume. The surface area affected by chloride concentrations exceeding 500 mg/L is 3.8 and 11.7 square miles in levels B and C, respectively (Table II-2). The modeling demonstrates that the injection of fresh water into level C causes the plume to widen to the north and south as the downgradient migration is slowed.

6. <u>Equus Beds Recharge with Plume Control (6a, 6b. 6c and 6d)</u>

The above Equus Beds recharge simulations were combined with plume control pumping (3c with rates of 4,000 gpm) to analyze the combined impacts of the two operations. A fourth management simulation (6d) was performed to analyze the impacts of plume control pumping at rates of 8,000 gpm.

The effects of the plume control wells at a pumping rate of 4000 gpm (simulation 3c) to this scenario is shown in Figures II-25 through II-27. The reduced flow gradients in the aquifer allows the remediation scenario to have a greater impact on the control and removal of the plume. The surface area impacted by chloride



LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- Proposed Equus Beds Recharge Well

1996 Chloride Concentrations

- 50 mg/l
- 100 mg/l
- 200 mg/l
- 300 mg/l
- 400 mg/l
- 500 mg/l
- 750 mg/l
- 1000 mg/l
- 1250 mg/l
- 1500 mg/l
- 1750 mg/l
- 2000 mg/l

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Figure II-22

LEVEL A
CHLORIDE CONCENTRATIONS
5C EQUUS BEDS
RECHARGE PROJECT
8530 GPM FOR 50 YEARS

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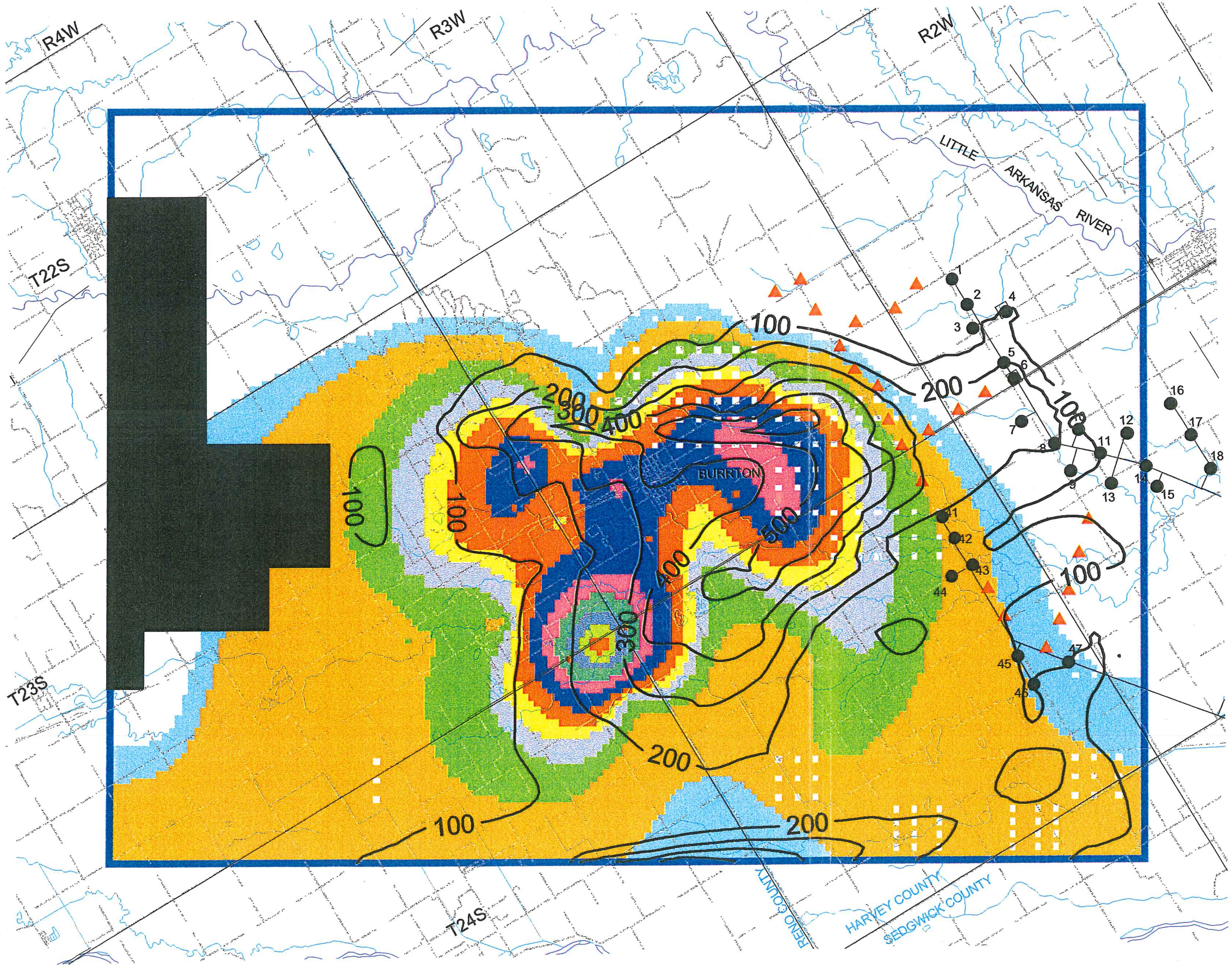

10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- Proposed Equus Beds Recharge Well

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-23

LEVEL B
CHLORIDE CONCENTRATIONS
5C EQUUS BEDS
RECHARGE PROJECT
8530 GPM FOR 50 YEARS

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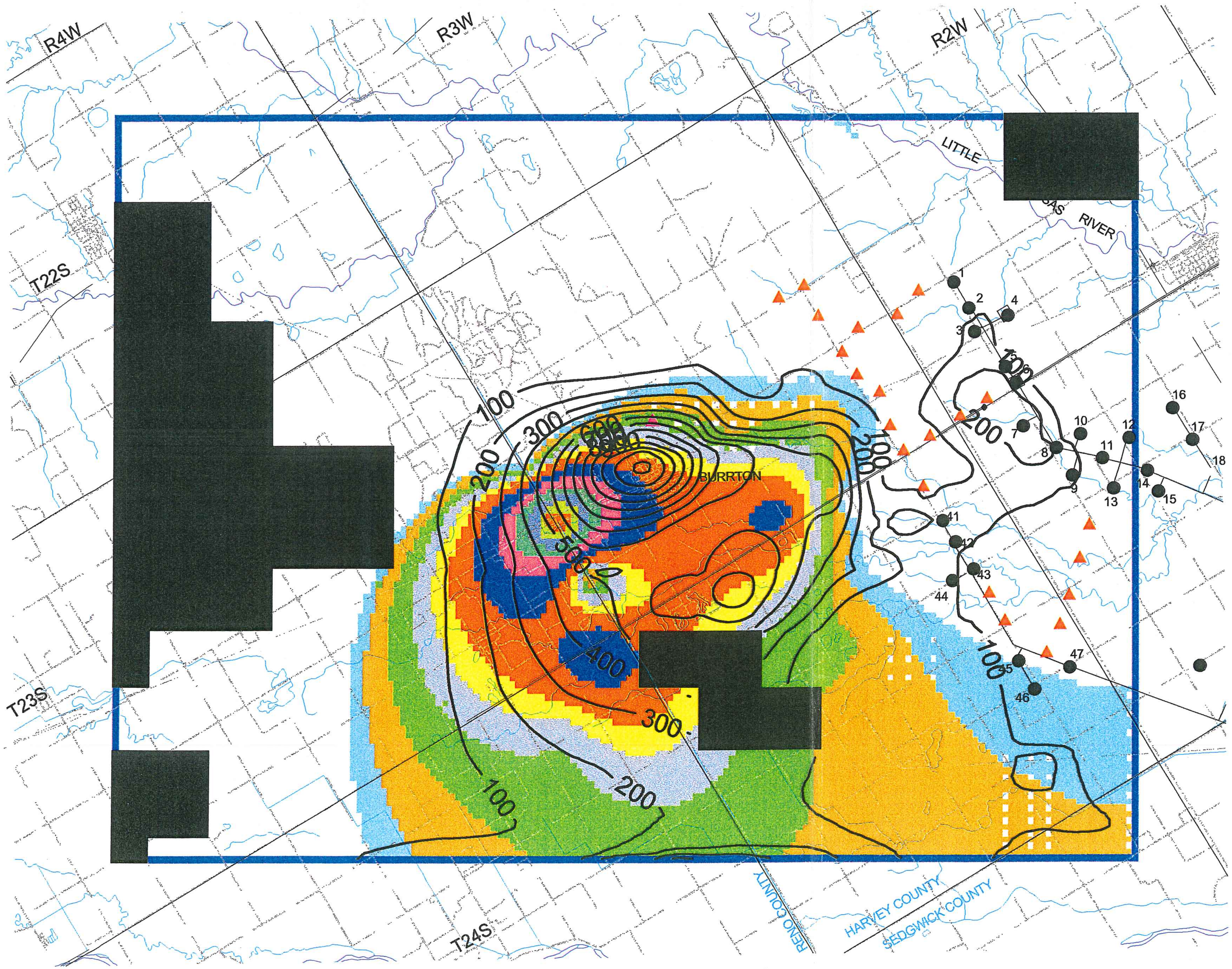

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

Chloride Isoconcentration Line (Modeled for 50 Years)

Wichita Municipal Well

Proposed Equus Beds Recharge Well

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-24

LEVEL C
CHLORIDE CONCENTRATIONS
5C EQUUS BEDS
RECHARGE PROJECT
8530 GPM FOR 50 YEARS

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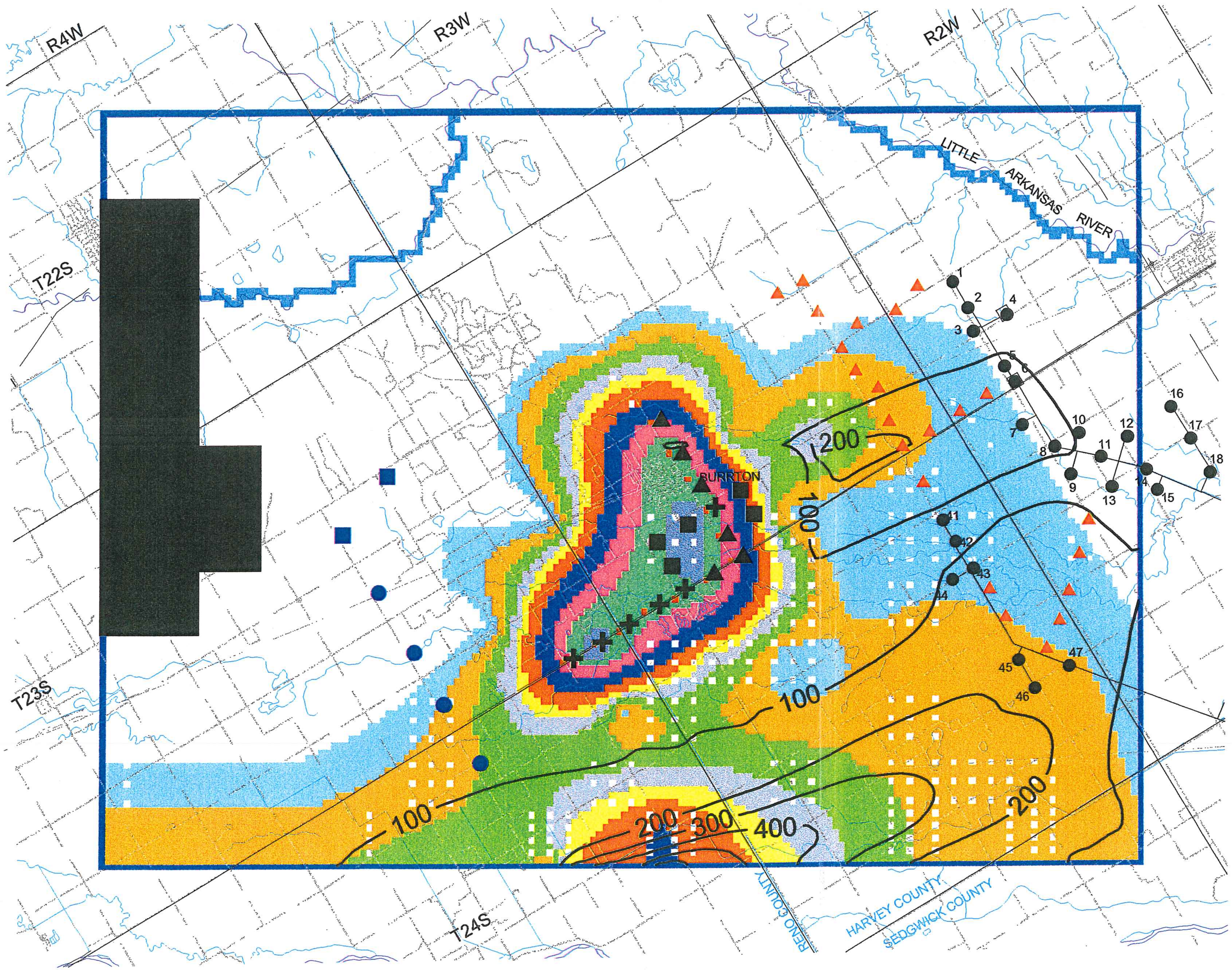

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- PROPOSED EQUUS BEDS RECHARGE WELL

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-25

LEVEL A
CHLORIDE CONCENTRATIONS
6C EQUUS BEDS RECHARGE
PROJECT WITH PLUME CONTROL
4000 GPM FOR 50 YEARS

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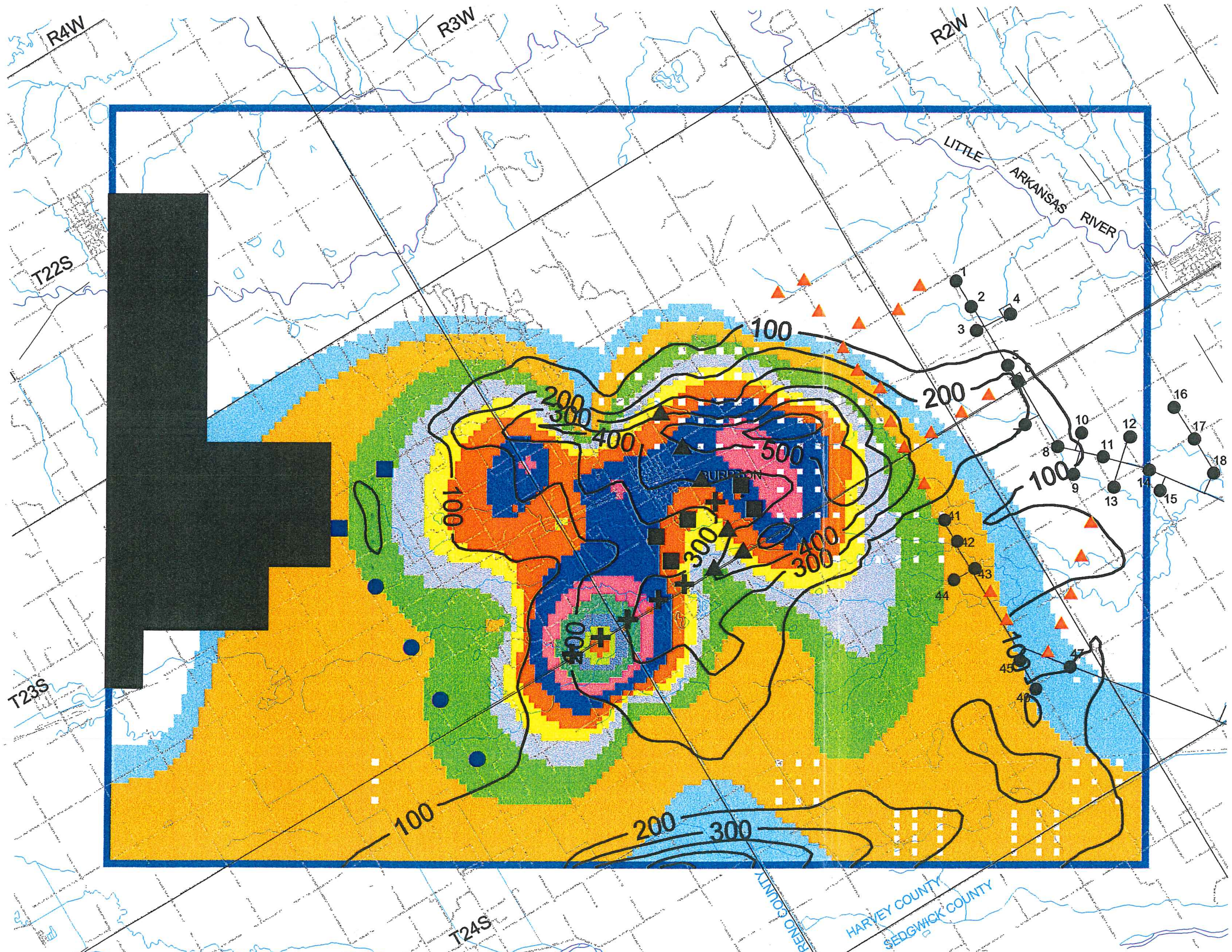

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- PROPOSED EQUUS BEDS RECHARGE WELL

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-26

LEVEL B
CHLORIDE CONCENTRATIONS
6C EQUUS BEDS RECHARGE
PROJECT WITH PLUME CONTROL
4000 GPM FOR 50 YEARS

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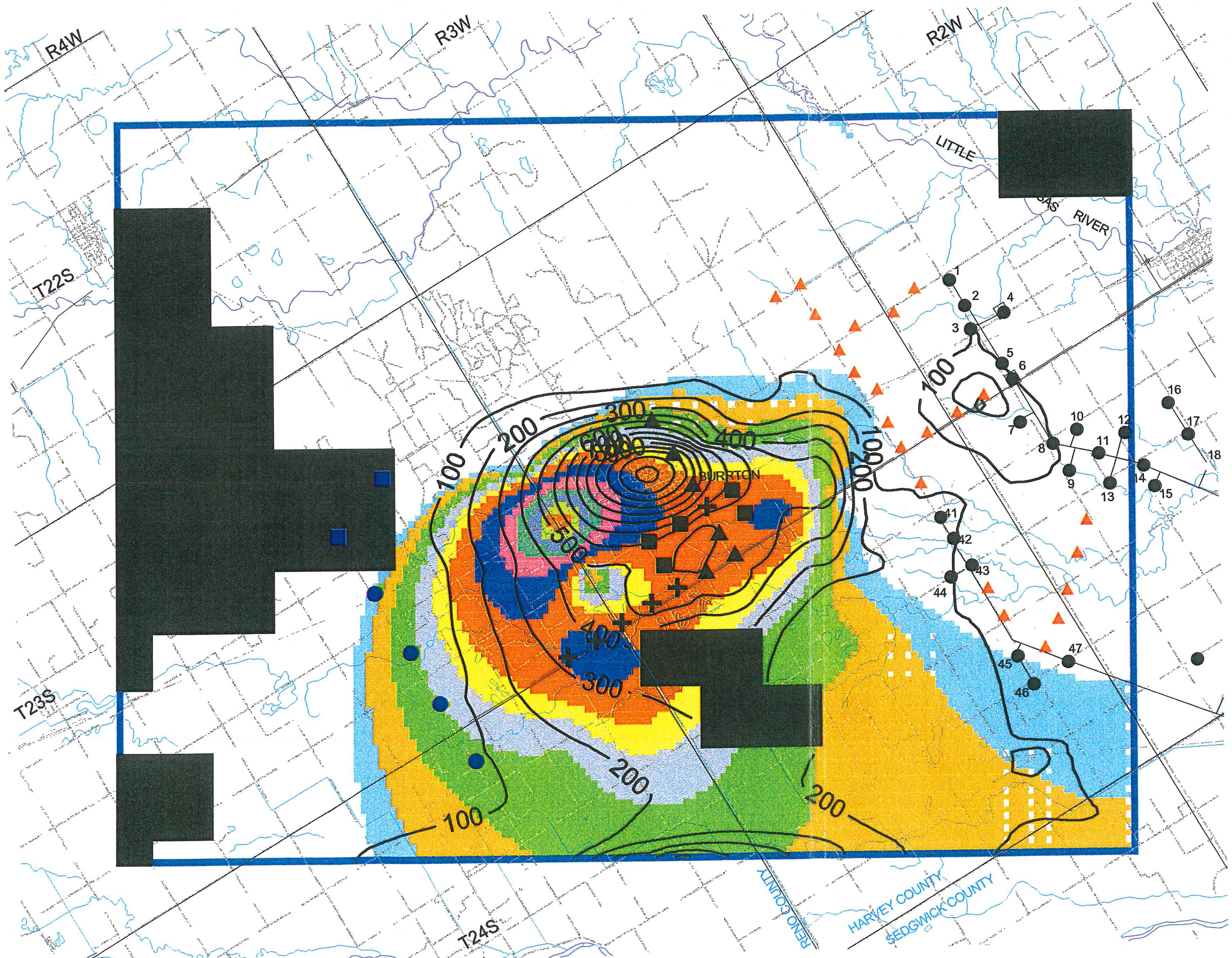

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B Extraction Well
- A Extracton Well
- B Extraction Well
- B/C Gradient Control Well
- B Gradient Control Well
- Proposed Equus Beds Recharge Well

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-27

LEVEL C
CHLORIDE CONCENTRATIONS
6C EQUUS BEDS RECHARGE
PROJECT WITH PLUME CONTROL
4000 GPM FOR 50 YEARS

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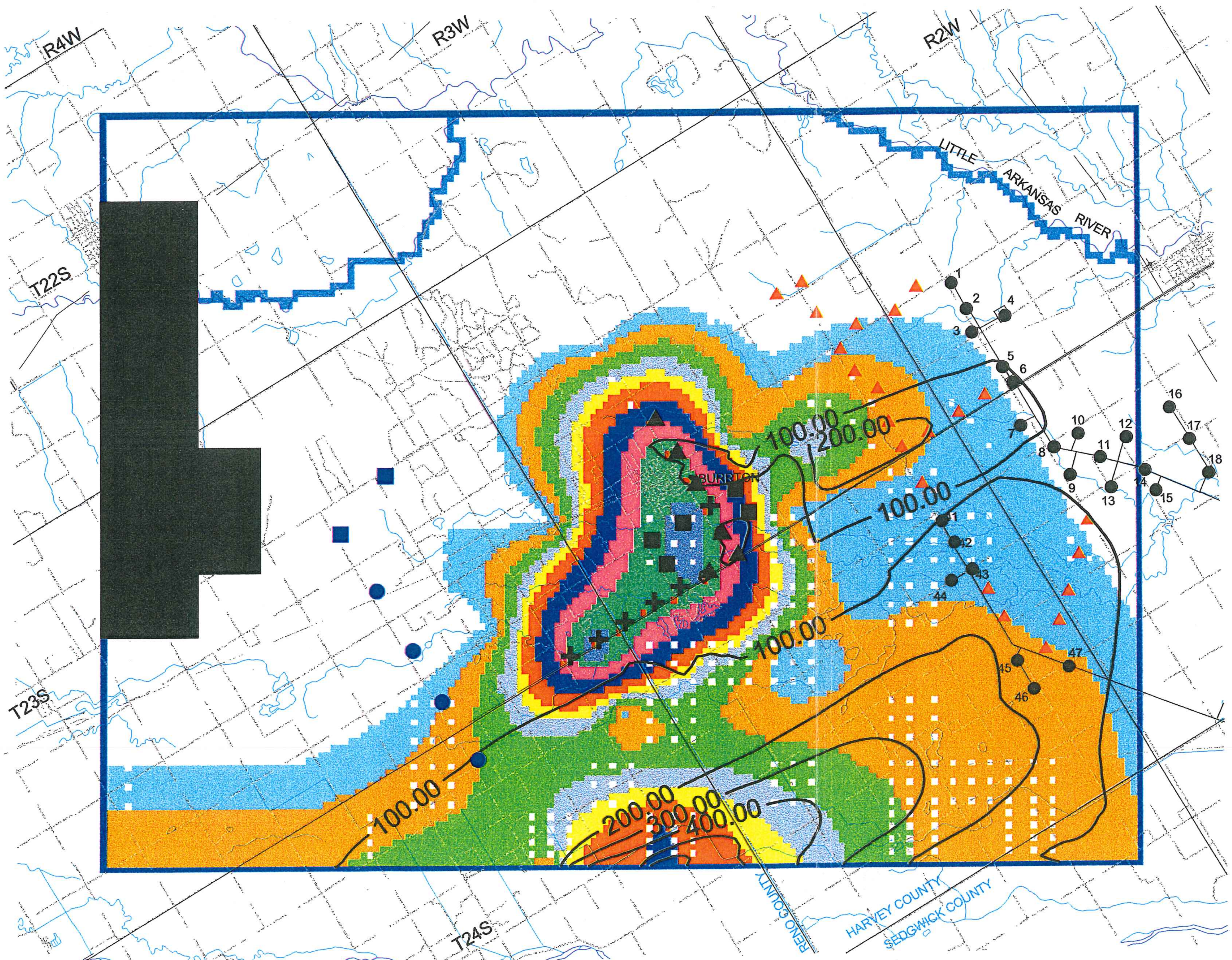
concentrations in the plume exceeding 500 mg/L is reduced to 2.0 and 8.5 square miles in levels B and C, respectively (Table II-2). Over the 50 year period, approximately 121,000 tons of salt are removed from the Equus Beds Aquifer with this management simulation. Additionally, with this scenario only 9,000 tons are pulled into the model area in response to the remediation pumping which is a significant improvement over the other management simulations. Only 7% of the salt removed from the plume is replaced with salt from outside of the model area (Table II-3).

When the plume control pumping rate was raised to 8,000 gpm (simulation 6d), the plume reduction was even more significant as shown in Figures II-28 through II-30. The impacted surface area for chloride concentrations exceeding 500 mg/L is reduced to 1.3 and 6.3 square miles for levels B and C, respectively (Table II-2). The modeling shows that approximately 257,000 tons of salt are removed. However, over 100,000 tons of salt is induced from the south. The net reduction of salt in the study area is 146,000 tons. The percentage of the salt pulled into the model area is 42 percent of the salt removed from the plume area (Table II-3).

These results shows that discharge pumping rate will have to be carefully established in order to obtain opium salt removal efficiency for the entire study area. Also, timing with other pumping stresses or recharge activities will have significant impacts or benefits to the remediation plan.

7. <u>Pilot Study (7a)</u>

A pilot scenario with a total pumping rate of 500 gpm was modeled using the 1996 conditions of the aquifer. Two wells were selected from plume control pumping simulation and pumped at 250 gpm each (Figure II-12). Simulation of the pilot study shows the approximately 21,000 tons of salt would be removed from the system over a 50 year time period.



10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- PROPOSED EQUUS BEDS RECHARGE WELL

Initial Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-28

LEVEL A
CHLORIDE CONCENTRATIONS
6D EQUUS BEDS RECHARGE
PROJECT WITH PLUME CONTROL
8000 GPM FOR 50 YEARS

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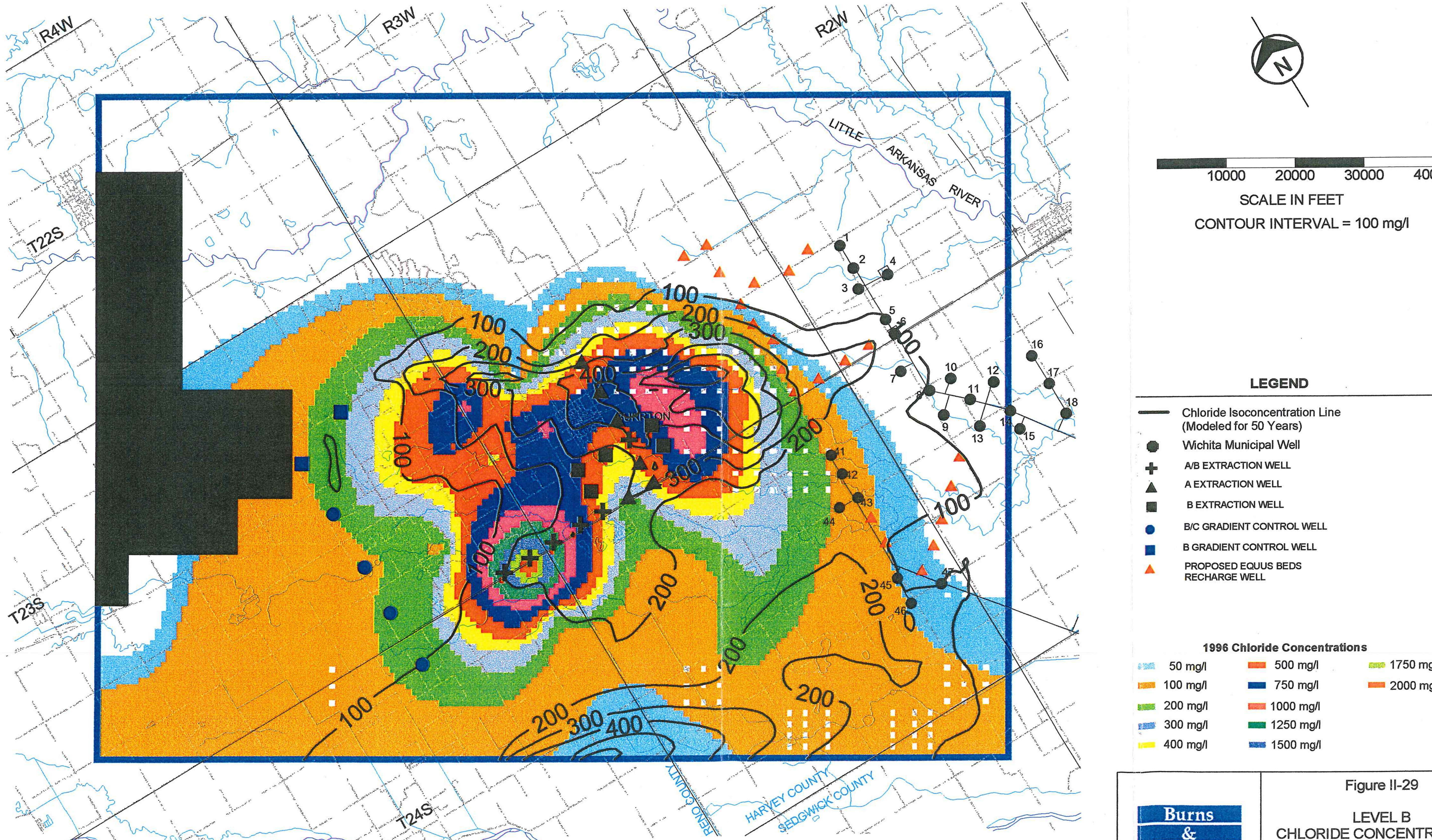

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Figure II-29

LEVEL B
CHLORIDE CONCENTRATIONS
6D EQUUS BEDS RECHARGE
PROJECT WITH PLUME CONTROL
8000 GPM FOR 50 YEARS

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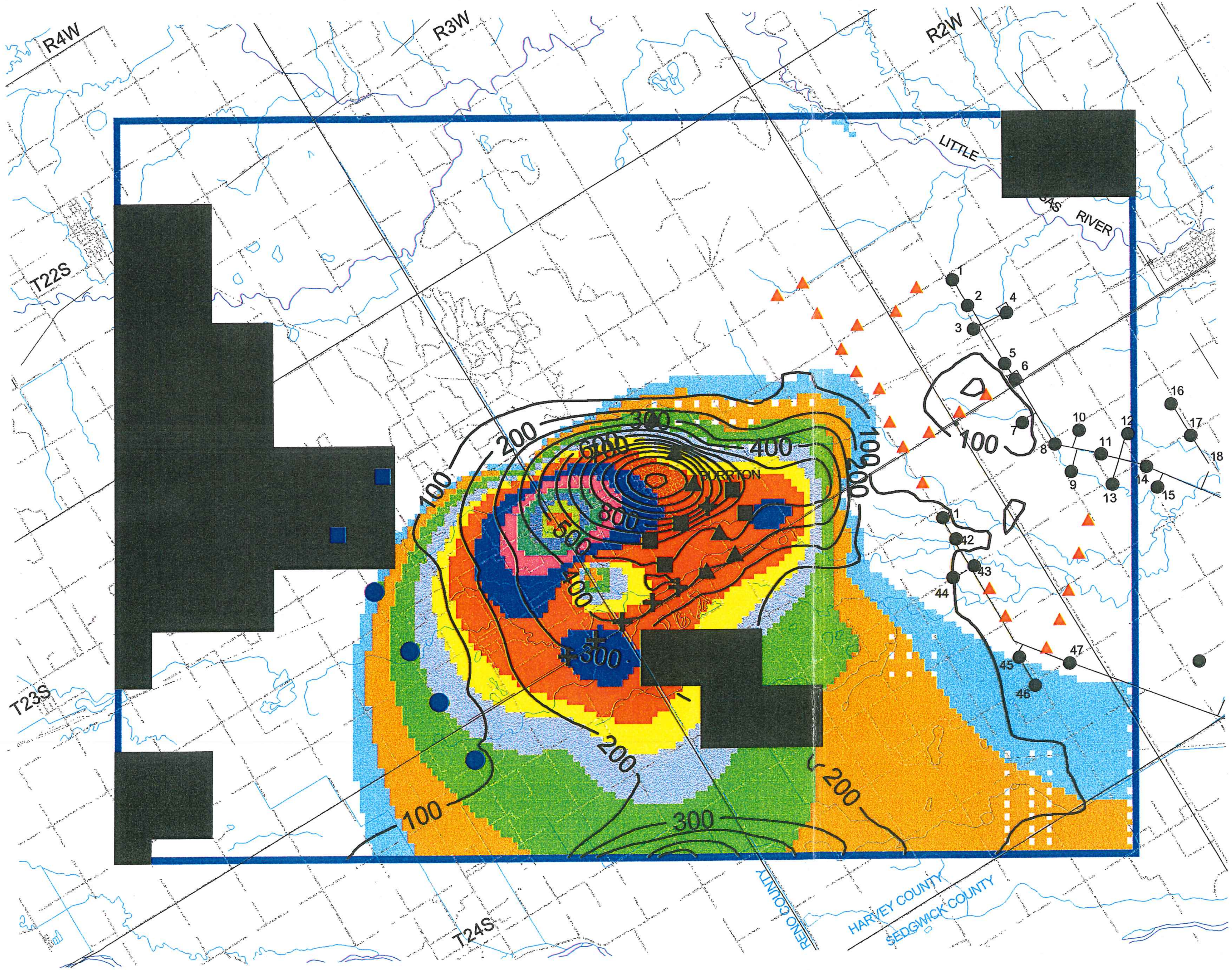

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10000 20000 30000 40000

SCALE IN FEET

CONTOUR INTERVAL = 100 mg/l

LEGEND

- Chloride Isoconcentration Line (Modeled for 50 Years)
- Wichita Municipal Well
- A/B EXTRACTION WELL
- A EXTRACTION WELL
- B EXTRACTION WELL
- B/C GRADIENT CONTROL WELL
- B GRADIENT CONTROL WELL
- PROPOSED EQUUS BEDS RECHARGE WELL

1996 Chloride Concentrations

50 mg/l	500 mg/l	1750 mg/l
100 mg/l	750 mg/l	2000 mg/l
200 mg/l	1000 mg/l	
300 mg/l	1250 mg/l	
400 mg/l	1500 mg/l	

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Figure II-30

LEVEL C
CHLORIDE CONCENTRATIONS
6D EQUUS BEDS RECHARGE
PROJECT WITH PLLUME CONTROL
8000 GPM FOR 50 YEARS

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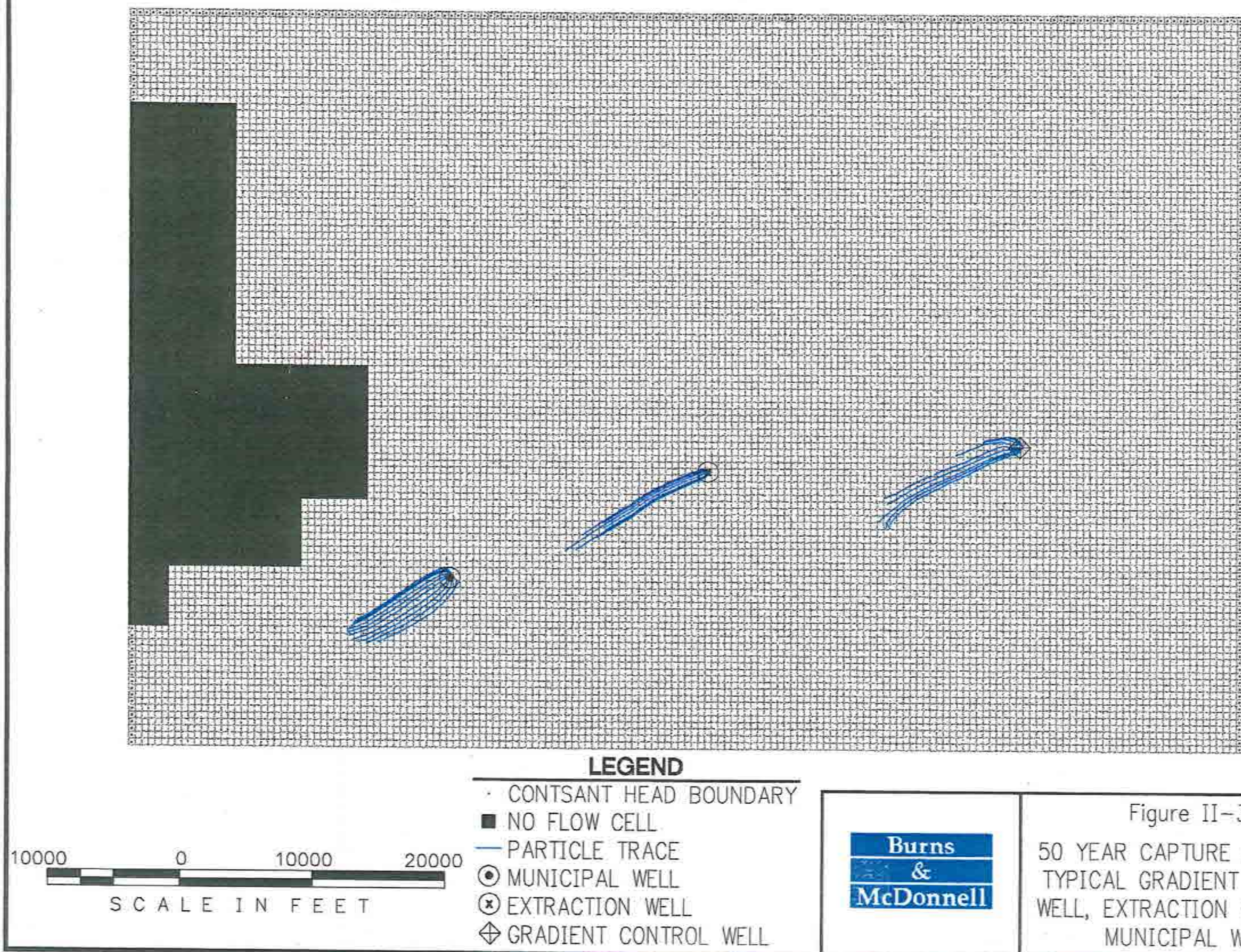

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Figure II-31

50 YEAR CAPTURE AREA FOR TYPICAL GRADIENT CONTROL WELL, EXTRACTION WELL, AND MUNICIPAL WELL

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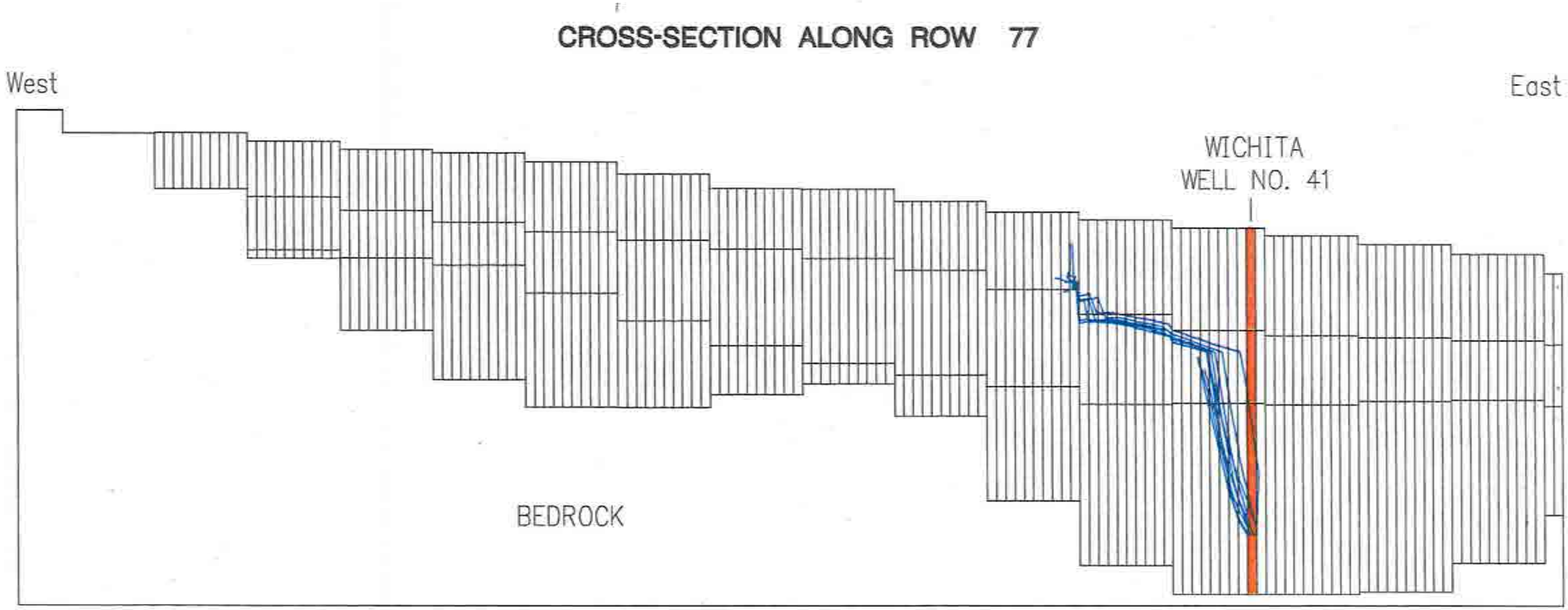

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Figure II-32

MIGRATION OF WATER PARTICLES FROM LEVEL A TO LEVEL C WICHITA WELL NO. 41

Part III - Remediation Technologies

PART III

REMEDIATION TECHNOLOGIES

A. GENERAL

Remediation techniques for the beneficial reuse or disposal of water pumped from plume control wells are evaluated herein. The development of these remedial technologies involve a comprehensive review of existing water quality, establishment of water quality goals for each reuse or disposal technique, and analyses of modeling results to balance water volume and chloride concentration. These reuse or disposal technologies are based on pumping strategies similar to those modeled in the management simulations described in Part II.

B. MEMBRANE TREATMENT PROCESSES

A primary purpose behind the development and continued research of membrane technologies has been desalination of municipal drinking water supplies. Electrodialysis (ED) and reverse osmosis (RO) have been used since the early 1960's in competition with distillation processes. Distillation processes include multistage-flash evaporation, multiple-effect evaporation, and vapor compression. Distillation technologies currently account for approximately 60 percent of the world's desalting capacity, whereas RO and ED respectively account for about 35 and 6 percent.

Over the last 10 years RO treatment capacity worldwide has increased by about 20 to 30 percent, while distillation capacity has decreased about 10 percent. Although there is greater total worldwide desalting capacity by distillation, there is a greater number of treatment plants employing membrane processes. This is due to increased interest in microbial removal, technological improvements and reductions in RO costs relative to distillation. Therefore, only membrane processes are considered further for this evaluation to control costs and maximize water quality and potential beneficial uses.

Several pressure driven membrane processes are applicable for current water treatment standards. These processes include RO, nanofiltration (NF), ultrafiltration (UF) and microfiltration (MF) and are distinguished by varying operating pressures and membrane pore size. Larger pore size results in lower operating pressures. The processes can be ranked by decreasing operating pressures and increasing pore sizes as follows: RO, NF, UF and MF. Only the dense RO membranes are specifically tailored to retain salts and low-molecular-weight solutes as found in groundwater in the Burrton area.

Another membrane technology, electrodialysis (ED), is technically applicahle to this project. Electrodialysis is an operation by which ions are driven through ion-selective membranes under the influence of an electrical potential. By alternating cation- and anion-selective membranes in a stacked arrangement with thin channels between them, it is possible to produce alternating channels of fluid that are respectively enriched and depleted in ions. A typical ED application is large-scale production of potable water from brackish water fails to remove significant levels of organics.

The RO process is the preferred over ED for the remediation of chlorides from the groundwater in the study area for the following reasons:

- RO process has superior treatment effectiveness for saline feed water relative to other pressure-driven membrane processes.
- RO has greater organic removal efficiencies compared to ED.
- RO requires less land area and less capital expenditure than ED, for water quality and capacities considered in this report.

1. Reverse Osmosis

The size and cost of a RO system designed to treat water from the extraction wells to meet current drinking water standards depends on the quality and quantity of water to be treated. The assumed water input rate (feed rate) is based on the total pumpage from the extraction well network. Feed water quality is assumed to be the average of available analytical data from 1990 to 1996 from a selected monitoring wells in the immediate vicinity of the extraction well network. The water quality data used for the cost estimates is summarized in Table III-1.

A schematic for an RO treatment system applicable to this project is shown in Figure III-1. Each treatment train of the RO facility would consist of a chemical pre-treatment system, including greensand filters and cartridge filters to remove iron, manganese, and solids, booster pumps to pressurize the feed water, RO permeators, a degasifier, chemical post-treatment units, and various appurtenances. The RO membranes permit only water (permeate), and not dissolved ions, to pass through its pores. Contaminants are left behind in a brine solution or concentrate. The concentrate contains a significantly higher level of dissolved solids than the feed solution, thus requiring additional treatment or disposal.

The water to be treated contains significant levels of iron and manganese and will require pretreatment to prevent rapid RO membrane fouling. Pretreatment will also extend the life of the membranes and reduce maintenance costs. A typical pretreatment process to remove iron, manganese and solids, consists of aeration with greensand filters and 5 to 10 micron cartridge filters followed by acidification.

Table III - 1

RO FEED WATER QUALITY CHARACTERISTICS

Temperature	50 deg. F
Calcium	240 mg/L
Magnesium	105 mg/L
Sodium	500 mg/L
Potassium	7 mg/L
Barium	0.05 mg/L
Iron	30 mg/L
Sulfate	32 mg/L
Chloride	1280 mg/L
Nitrate	0.05 mg/L
Fluoride	0.28 mg/L
TDS	3020 mg/L
pH	7.45
Total Hardness	820 mg/L as $CaCO_3$
Total Alkalinity	150 mg/L as $CaCO_3$
Turbidity (NTU)	42
Specific Conductance	4380

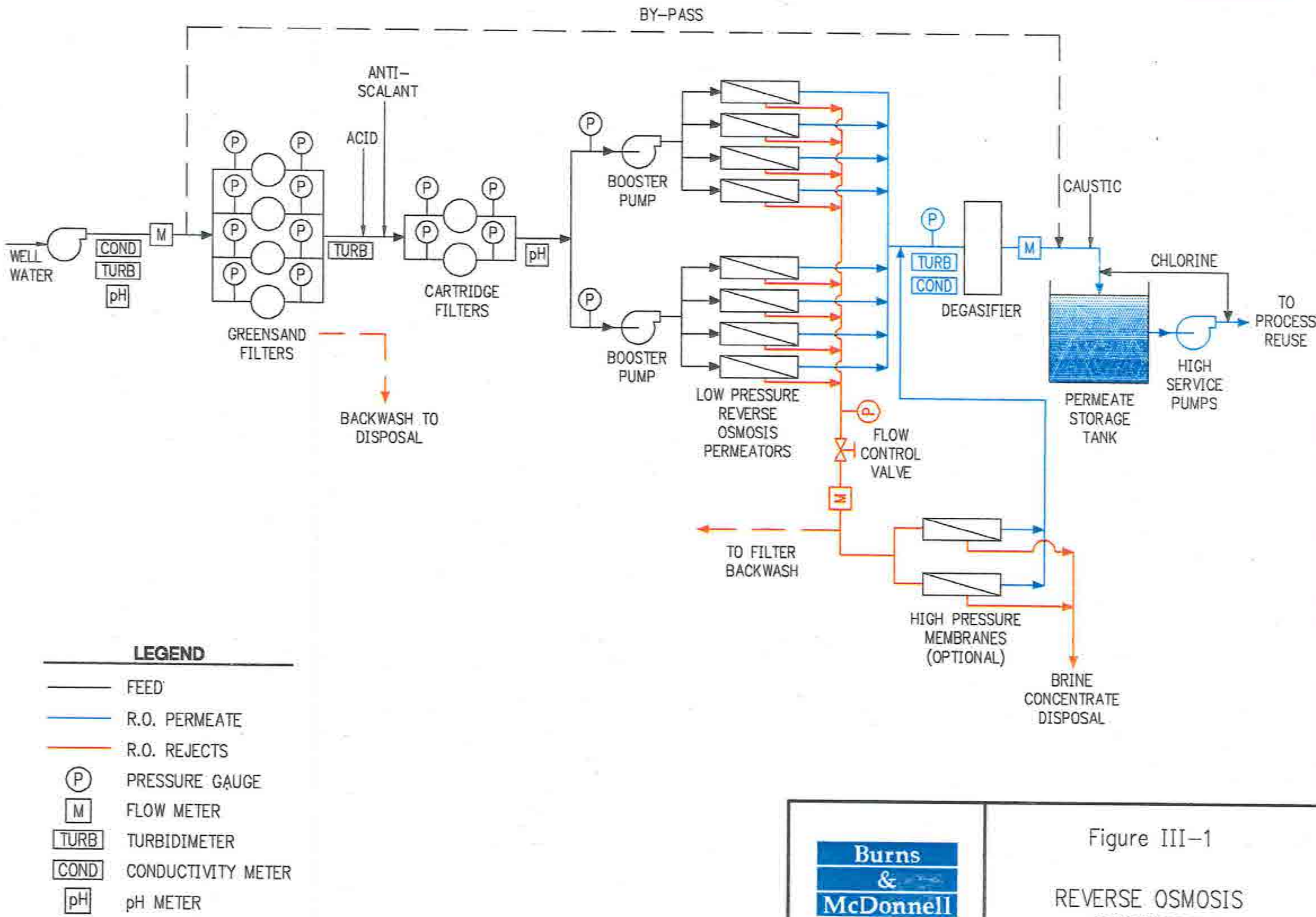

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Figure III-1

REVERSE OSMOSIS SCHEMATIC

a. <u>Waste Minimization (High Pressure Membranes)</u>

The RO concentrate flow rate from the low-pressure membranes is estimated to be 15 to 25 percent of the extraction wells pumping rate or about 75 to 300 gpm. One deep disposal well, discussed below, could potentially accommodate this flow. To further reduce the volume of waste, the RO concentrate could be further treated with use of high pressure RO membranes or by using wastewater evaporators.

The treatment system would still require one deep disposal well. The capital and operating costs for the addition of a high pressure membrane system or evaporator is not justified for this project at the flow rates reflected by the modeling scenarios. The use of these systems may be re-evaluated in the event higher volumes of concentrate are produced that would require an additional deep disposal well or if disposal well flow rates are less than expected.

b. <u>Reverse Osmosis Plant Pilot Study</u>

The Kansas Department of Health and Environment (KDHE) typically require a pilot plant study to verify the performance of an RO installation used for the treatment of a potable water supply. These studies serve to determine actual membrane performance, required chemical feed rates, optimal flux, ion rejection and recovery. Treatment of chloride-impacted groundwater to drinking water standards is considered a tested and proven technology. Recent discussions with KDHE indicated a willingness of the agency to accept a 1-year performance bond from the RO vendor as a substitute for a pilot study. KDHE would establish parameters indicative of success, in agreement with the recipient of the RO-treated water.

C. BENEFICIAL REUSE

The greatest potential for reuse of all or part of the water from the extraction or gradient control wells is for municipal use. Presently, there is little potential in the study area for industrial reuse of the recovered high chloride water. Additionally, potential use of the water for oil field flooding operations is limited. Therefore, blending of permeate from the RO process or pumpage from the gradient control wells with an existing municipal water supply was evaluated for beneficial reuse. The use of this flow by the City of Wichita and smaller nearby communities may be considered, depending on future water needs and requirements.

1. Public Water District

Several communities within the vicinity of the study area, including Sedgwick, Halstead and Newton, are currently considering a consolidation of water rights into a public water district. This consolidation would be centered at the Mission treatment facility, located about eight miles southwest of Newton. Transmission mains would interconnect the facility to nearby communities and the north Newton well field. The total average day water demand for Sedgwick, Halstead and Newton is currently estimated to be less than 5 million gallons per day (MGD) and is not expected to grow significantly during the next decade.

2. City of Wichita

The City of Wichita is performing studies to implement their Integrated Local Water Supply Plan (Plan). The Plan was adopted by Wichita in 1993 and has been slightly modified based on ongoing studies. Current studies indicate that the City will need additional water sources to meet a projected demand of 98.5 MGD in Year 2010. The Plan uses raw water from the following local sources. Associated chloride concentrations are also listed:

Water Source	Chloride Level (mg/L)
Cheney Reservoir	160 to 190
Equus Beds Aquifer	50 to 75
Local Well Field Expansion	Being studied
Local (E&S) Well Field	200
Bentley Reserve Well Field	350 to 800
Little Arkansas River	5 to 350

Under the City's Plan pumpage from the Equus Beds and Bentley Reserve Well Fields would be blended at a 3 to 1 ratio to maintain a chloride concentration of less than 200 mg/L in the finished water. The City's maximum desired finished water chloride concentration is 200 mg/L and their goal is 125 to 150 mg/L.

Recharge of the Equus Beds Aquifer in the City's well field area is a key component of the Plan. Above-base flow from the Little Arkansas River will be recharged and stored in the aquifer for recovery during dry periods. It is recognized by the City that the recharge project will also help reduce the migration of salt water from the Arkansas River and the Burrton area. Recharge activities, as currently planned, include recharge of treated above-base flow surface water through recharge basins and recharge of induced river infiltration through recharge basins and recharge wells. It is anticipated that the transmission and distribution of the recharge water will be through the existing well field pipeline system.

Potential addition of high chloride water from the Burrton plume management pumping to the Wichita aqueduct for City use will have to be carefully evaluated for the final design. The Equus Beds Well Field must continue to operate as a water supply source during recharge activities and water used for recharge must maintain a relatively low chloride content. The addition of high chloride water into the well field piping near the northwest section of the system could create control problems during recharge activities. To minimize impact to the Equus

Beds Aquifer, the high chloride water would require treatment to reduce chlorides to about 50 mg/L while recharge operations are in progress.

Chloride levels of pumpage from the extraction and/or gradient control well system could be allowed to exceed 50 mg/L if the flow could be directed to Wichita without being recharged into the aquifer or interfering with well field operation. The impacts of using water with chlorides of 125 to 1,500 mg/L are listed in Table III-2 and are based on the City's desired chloride concentration limit of 150 and the maximum limit of 200 mg/L for the blended finished water. Assumptions of the analysis are as follows:

- Wichita service area average day demand in Year 2010 is 98.5 MGD.
- Cheney Reservoir provides about 60 percent of demand at a chloride concentration of 175 mg/L .
- Equus Beds provides about 30 percent of demand at a chloride concentration of 60 mg/L.
- Local Well Field provides about 10 percent of demand at a chloride concentration of 200 mg/L.

Review of this data shows that if the City maintains a finished water chloride target of 150 mg/L the maximum flow rate that could be accepted at an influent concentration of 500 mg/L would be about 1.4 MGD (about 970 gpm). If the City would accept a higher finished water chloride target, larger volumes of Burrton plume control water could be used.

Table III-2
Potential Usable Flow (MGD)
from
Burrton Plume Control Wells

Control Well Influent Chloride Conc. (mg/L)	Wichita Finished Water Chloride Target 150 mg/L	200 mg/L
125	-*	-*
250	2.8	22.4
500	1.4	11.2
1,000	0.7	5.6
1,500	0.5	3.7

* If chloride concentration is less than the City's finished water target the potential usable flow is limited to control well system capacity.

Additionally, water providers experience seasonal variations in water demand which will influence the amount of remediation water that can be used. During the winter, with lower system demands, the amount or chloride concentration of the remediation water may have to be lowered to not exceed finished water target limit. In the summer, with higher demands, larger volumes of water or higher concentration may be feasible without exceeding the desired water quality limits.

Management simulation 3c assumes a total discharge rate 4,000 gpm (5.8 MGD) with the flow evenly divided between gradient control wells and extraction wells. The contaminant transport model indicated that the initial average concentration for the entire flow would be about 620 mg/L. After approximately three years, the concentration would fall below 500 mg/L. The modeled chloride concentrations (mg/L) of discharges from the gradient control and extraction wells are as follows:

	Gradient Control	Extraction	Blended
Initial	175	1,100	620
1 Year	135	1,000	570
5 Year	105	790	450

D. DISPOSAL TECHNIQUES

Disposal options for extraction well pumpage include deep well injection and discharge to surface water. Only deep well injection of brine concentrate from the RO process is considered. Other disposal options for RO concentrate, including mechanical evaporators, crystallizers and spray dryers are precluded from in-depth evaluation due to high capital costs associated with systems in this capacity range. Discharge of extraction well pumpage and/or RO concentrate into a local public owned treatment works (POTW) is not a feasible disposal option because of discharge water quantity and quality.

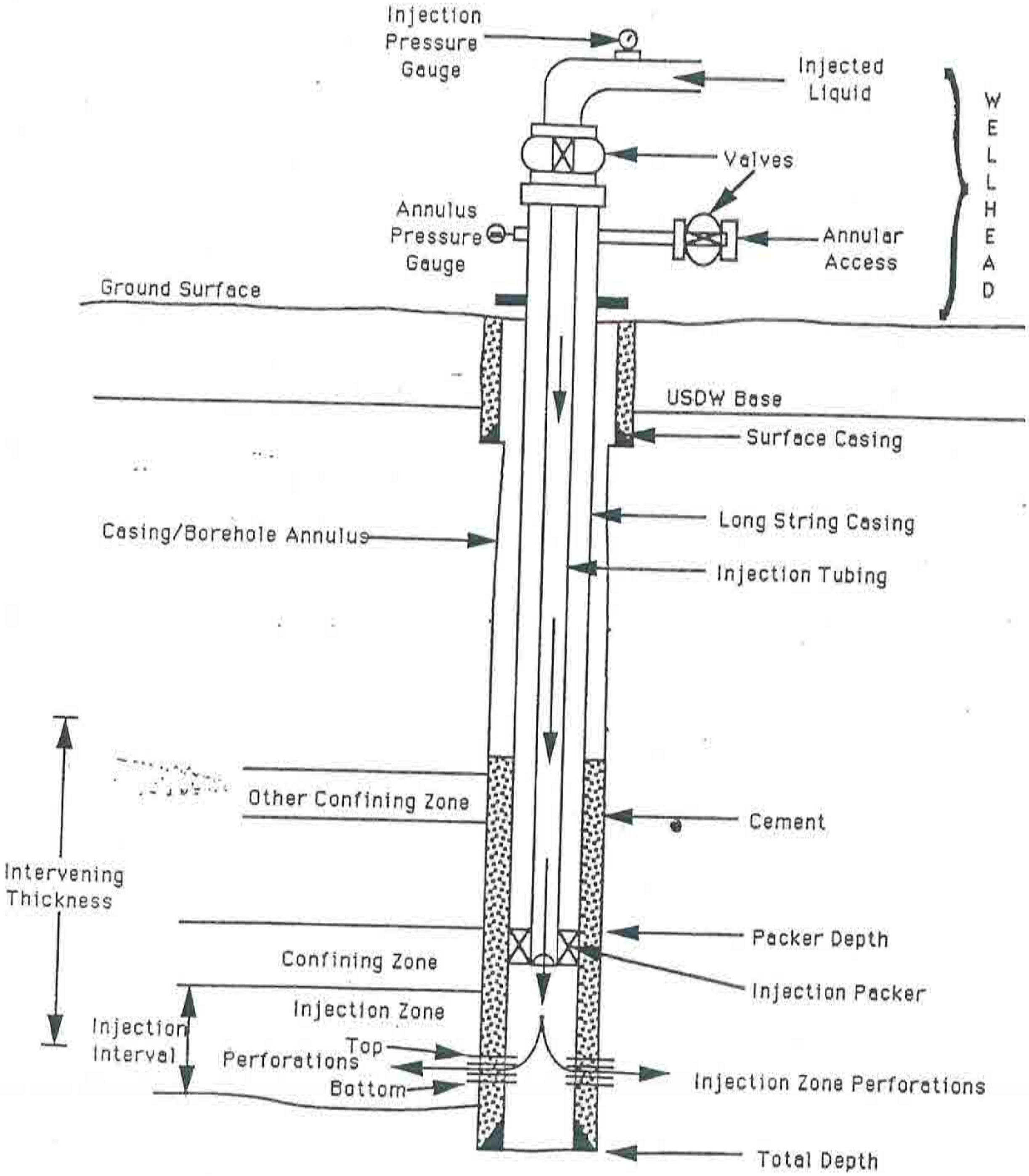
1. <u>Deep Well Injection</u>

Deep well injection is suitable for relatively small volumes of particularly high chloride concentrations, including RO concentrate and pumpage from extraction wells. The chloride levels from the extraction wells or RO concentrate may be to high for blending with an existing municipal water supply or discharge to a stream using a National Pollutant Discharge Elimination System (NPDES) permit.

Injection wells are a disposal option in which liquid wastes are injected into deep porous subsurface rock formations. Careful design and operation of injection wells is important to prevent movement of wastes into or between underground sources of drinking water.

Brine wastes related to oil field operations can be disposed of in Class II wells. These wells are regulated by Kansas Corporation Commission (KCC). A schematic of a typical Class II disposal well is shown in Figure III-2. Class II includes wells that inject fluids:

- that are brought to the surface in connection with conventional natural gas storage operations or conventional oil or natural gas production and may be commingled with wastewater from gas plants that are an integral part of production operations, unless those waters are classified as a hazardous waste at the time of injection.
- for enhanced recovery of oil or natural gas.
- for storage of hydrocarbons that are liquid at standard temperature and pressure.



SOURCE: KCC, 1997

Burns & McDonnell

FIGURE III-2

SCHEMATIC OF CLASS II INJECTION WELL

In discussions concerning this project, EPA has confirmed that the RO concentrate resulting from treating groundwater containing oil field brine, in addition to high chloride water from the extraction wells, can be disposed in Class II wells regulated by KCC.

These wells would be constructed to depth of about 4,500 feet deep into the Arbuckle Formation. Typical capacity of a Class II well ranges from about 15,000 - 20,000 barrels per day (440 - 580 gpm). No spacing limitations are known. Additional pump head is not required because adequate head is provided by the water column within the well. Surge tanks are required to provide a manageable, continuous flow rate into a disposal well.

Due to the significant capital costs of this option, the minimization of disposal volume is prudent. This minimization can be achieved by placing extraction wells only were necessary within the zones of highest chloride concentrations in the plume.

2. <u>Discharge to Surface Water</u>

The Arkansas River is the nearest significant surface water body to the extraction/gradient control well system. The median chloride concentration of the water in the river near this project area is about 630 mg/L. Because of the river's high flow volume and high chloride concentration, the river was evaluated to receive discharge from Burrton plume control and extraction wells.

A NPDES permit will be required by KDHE to discharge all or a portion of the pumpage, provided that the discharge meets all applicable requirements of the Clean Water Act (CWA) relating to effluent limitations, water quality standards and implementation plans, new source performance standards, toxic and pretreatment effluent standards, inspections, monitoring and entry provisions.

Treatment may be deemed necessary to stabilize the concentrate or to remove constituents harmful to the receiving water flora and fauna or both. The most common treatment requirement is aeration to increase the DO concentration. The chloride concentrations of the discharged water allowable under an NPDES permit are anticipated to be less than concentrations within the Arkansas River.

Pending regulatory reviews, chloride levels allowable through future NPDES permit renewals are likely to become more stringent. Discharge to the Arkansas River may currently be allowed considering the quality of the receiving water. Allowable chloride discharge standards are currently under review and are to be finalized by Year 2001. Permits issued under current standards would then be required to meet those new standards.

Chloride concentrations in the Arkansas River generally increase downstream of Great Bend Kansas, located approximately 60 miles upstream of Hutchinson. The source of elevated chloride concentrations is thought to be salt marshes on tributaries to the Arkansas River upstream of Hutchinson. Within the study area, water from the Arkansas River is classified as brackish or salty.

Chloride concentrations from samples of Arkansas River water collected near Hutchinson, Haven, Mount Hope, Bentley, and Maize the median generally ranges from 620 to 640 mg/L (Myers, 1996).

Blending of pumpages from extraction and gradient control wells can be considered to lower chloride concentrations and provide a more continuous discharge rate to surface water.

3. Evaporation Ponds

Solar evaporation is a well-established method for removing water from concentrated brines. Solar evaporation ponds have been used for centuries to recover salt (sodium chloride) from seawater. Evaporation ponds for concentrated brine disposal are appropriate primarily for regions of the United States having a relatively warm, dry climate with high evaporation rates, level terrain, and low land costs. There are several advantages associated with evaporation ponds which include simple construction, low maintenance, low energy requirements and little operator attention.

Despite the advantages of evaporation ponds, state permitting and other problems can limit their application. Evaporation ponds were not included as a remedial alternative for several reasons: (1) the study area is not characterized by particularly high year-round evaporation rates and the pumping is relatively high throughout the year, (2) KDHE is more reluctant to permit these structures because they generally fail and leak, contaminating the underlying groundwater.

* * * * *

Part IV - Remediation Alternatives

PART IV

REMEDIATION ALTERNATIVES

A. GENERAL

Five remediaton alternatives, based on the modeled management simulations, were developed to provide control and removal of the chloride plume, treatment of the pumpage (if applicable) and disposal and/or transmission for beneficial use of the water. Capital and annual costs are estimated and used to compare the alternatives. The layout of the gradient control and extraction wells are similar for each alternative. Each alternative includes a preliminary well and pipeline plan. These plans were developed using well locations correspond to management simulations with interceptor and gradient control wells used in the subregional model. The sizing of alternative components is based on a 4,000 gallons per minute (gpm) flow rate from the entire control well system. Additionally, individual component costs can be used to modify the evaluated alternatives. The alternatives are summarized as follows:

- **Alternative 1:** Pumpage from the extraction well network, 2,000 gpm, is continuously directed to four Class II disposal wells. Effluent from the gradient control wells, 2,000 gpm, are pumped to a connection at the Wichita Well Field and blended with the Wichita water supply and is shown in Figure IV-1.

- **Alternative 2:** Pumpage from the extraction well network, 2,000 gpm, is continuously directed to four Class II disposal wells. Gradient control well discharge is continuously directed to an outfall along the Arkansas River and is shown in Figure IV-2.

- **Alternative 3:** Pumpage from the extraction and gradient control networks are blended and discharged at an outfall along the Arkansas River and are shown in Figure IV-3.

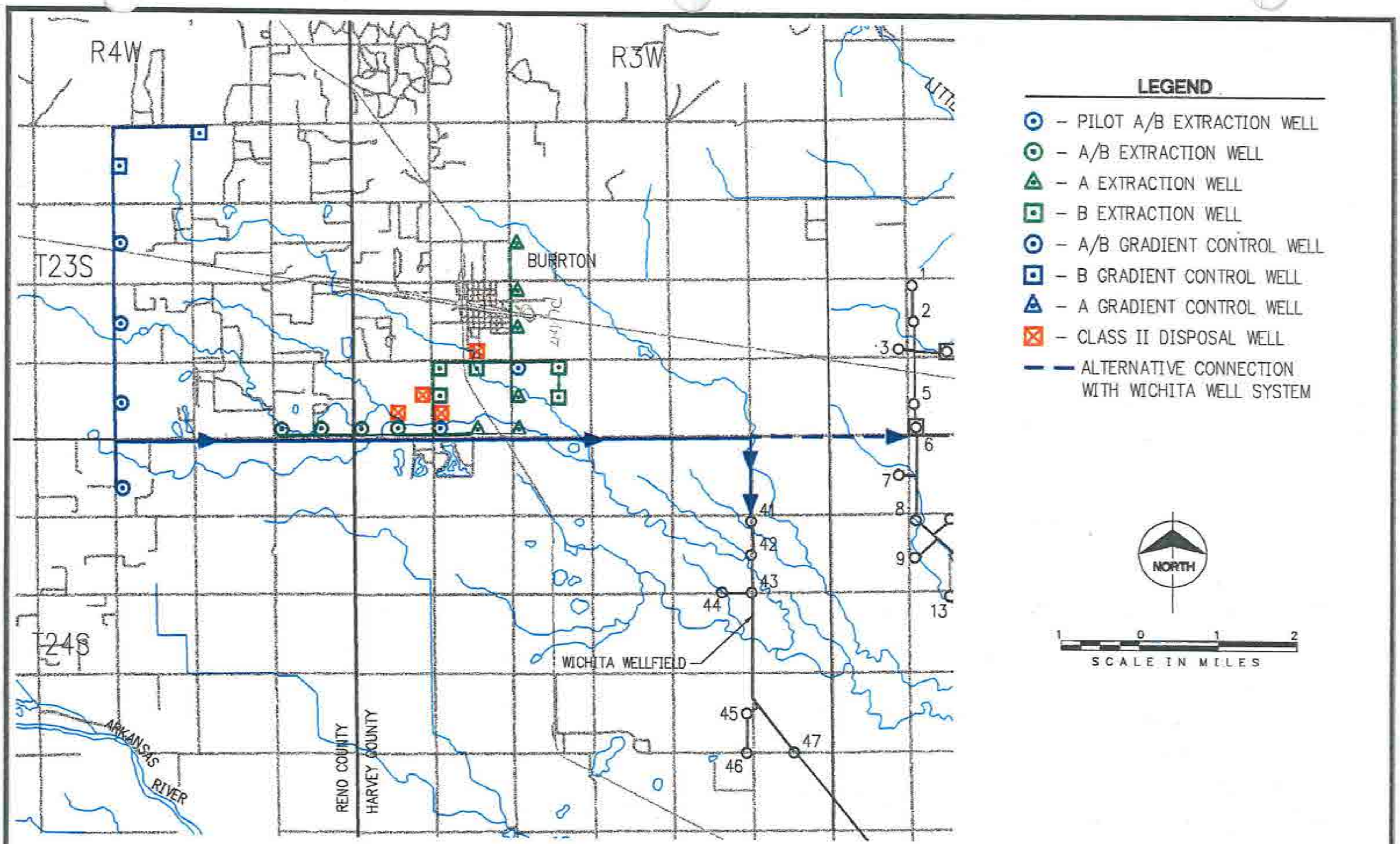


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Figure IV – 1

ALTERNATIVE 1

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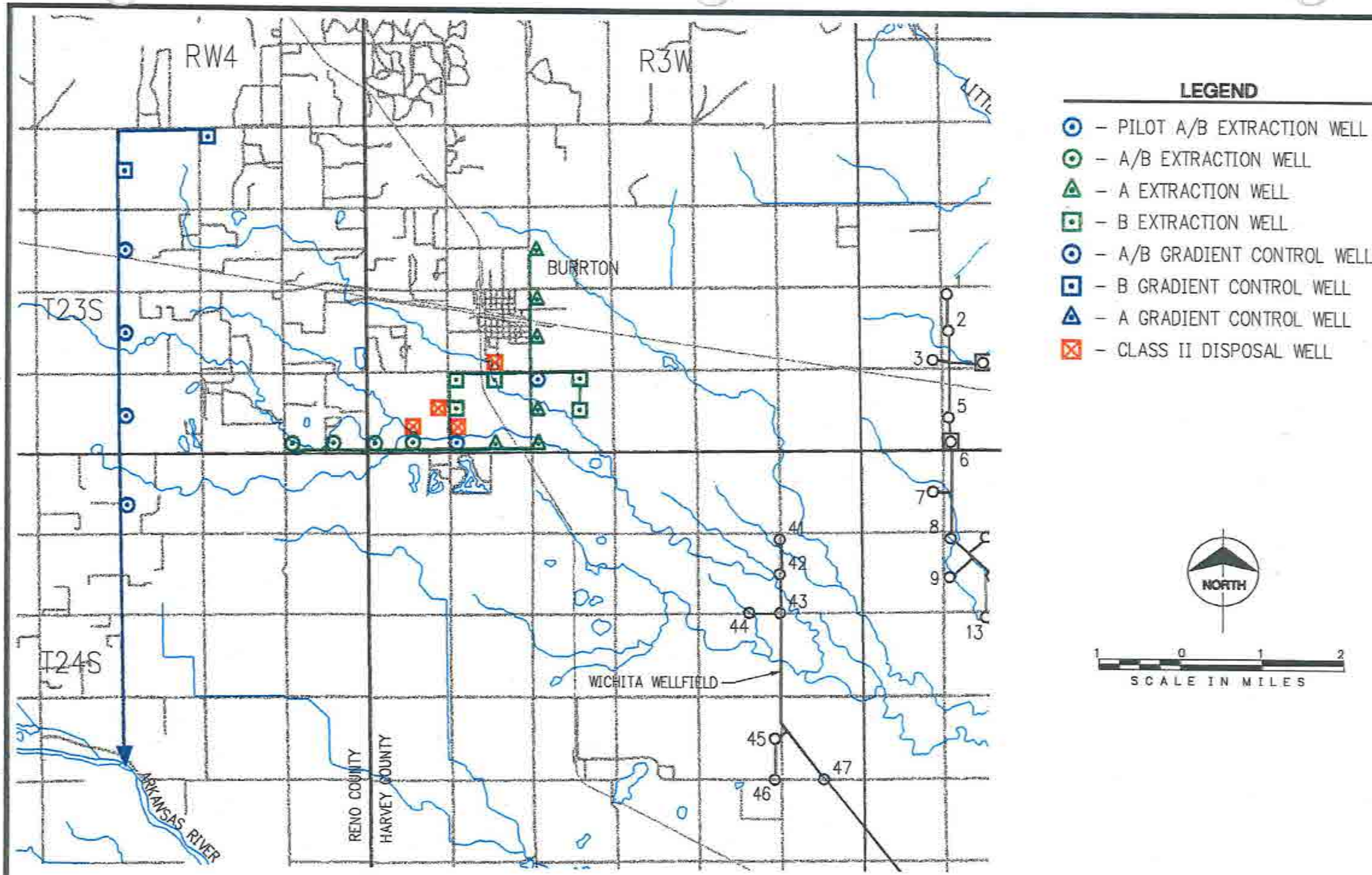

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Figure IV – 2

ALTERNATIVE 2

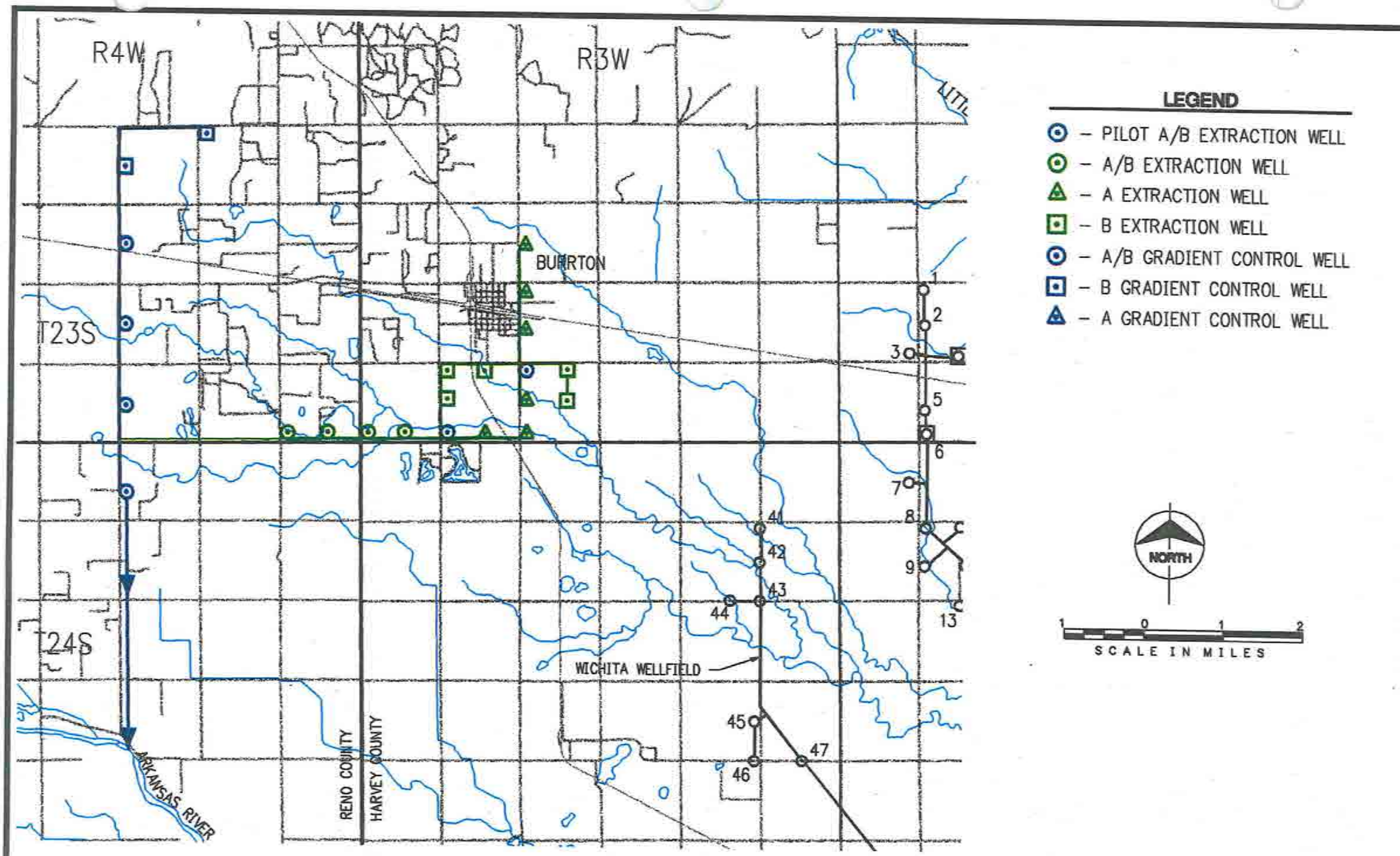

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Figure IV – 3

ALTERNATIVE 3

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- **Alternative 4:** Pumpage from the extraction well network, 2,000 gpm, is routed to a reverse osmosis (RO) treatment plant. RO concentrate is disposed via one Class II disposal well. RO permeate and gradient control pumpage is blended and discharged at a connection with the Wichita Well Field and is shown in Figure IV-4.

- **Alternative 5:** Pumpage from the extraction well network, 2,000 gpm, is routed to an RO treatment plant. RO concentrate is disposed via one Class II disposal well. RO permeate and gradient control pumpage is blended and discharged to four one-half acre recharge basins and is shown in Figure IV -5.

After initial evaluation of the alternatives, costs were developed a pilot-scale installation of two 250 gpm extraction wells with disposal of pumpage to one Class II disposal well and is shown in Figure IV-6.

Alternatives 1, 4 and 5 require a recipient for blended or RO treated water. The proximity of the potential recipient affects total pipeline costs. Two water suppliers have been identified as potential recipient of the reclaimed water. These are the proposed public water supply district and the City of Wichita. The public water district interconnect is assumed to be at Halstead which is several mile further east that a connection with the existing Wichita well field pipeline. Additionally, the demands of the public water district may not be large enough to utilize the entire volume of water reclaimed from the Burrton brine plume. The City of Wichita has the capacity and potential demand to utilize all of the projects flows on a continuous basis provided the water quality does not interfere with recharge or normal water supply operations. Therefore, the City of Wichita is considered to be the recipient of water from the gradient or extraction well networks of each applicable remedial alternative for costing purposes.

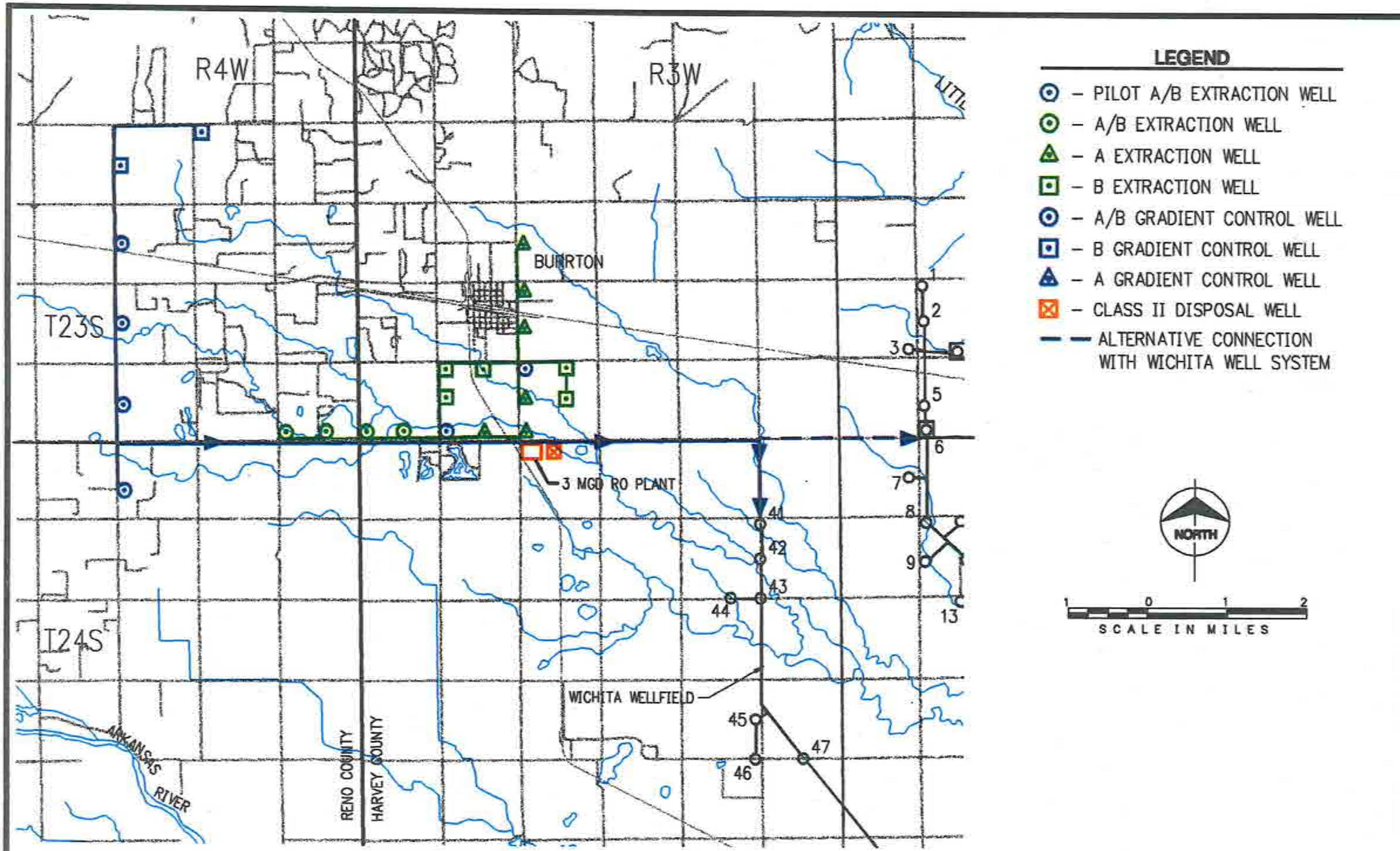

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Figure IV – 4

ALTERNATIVE 4

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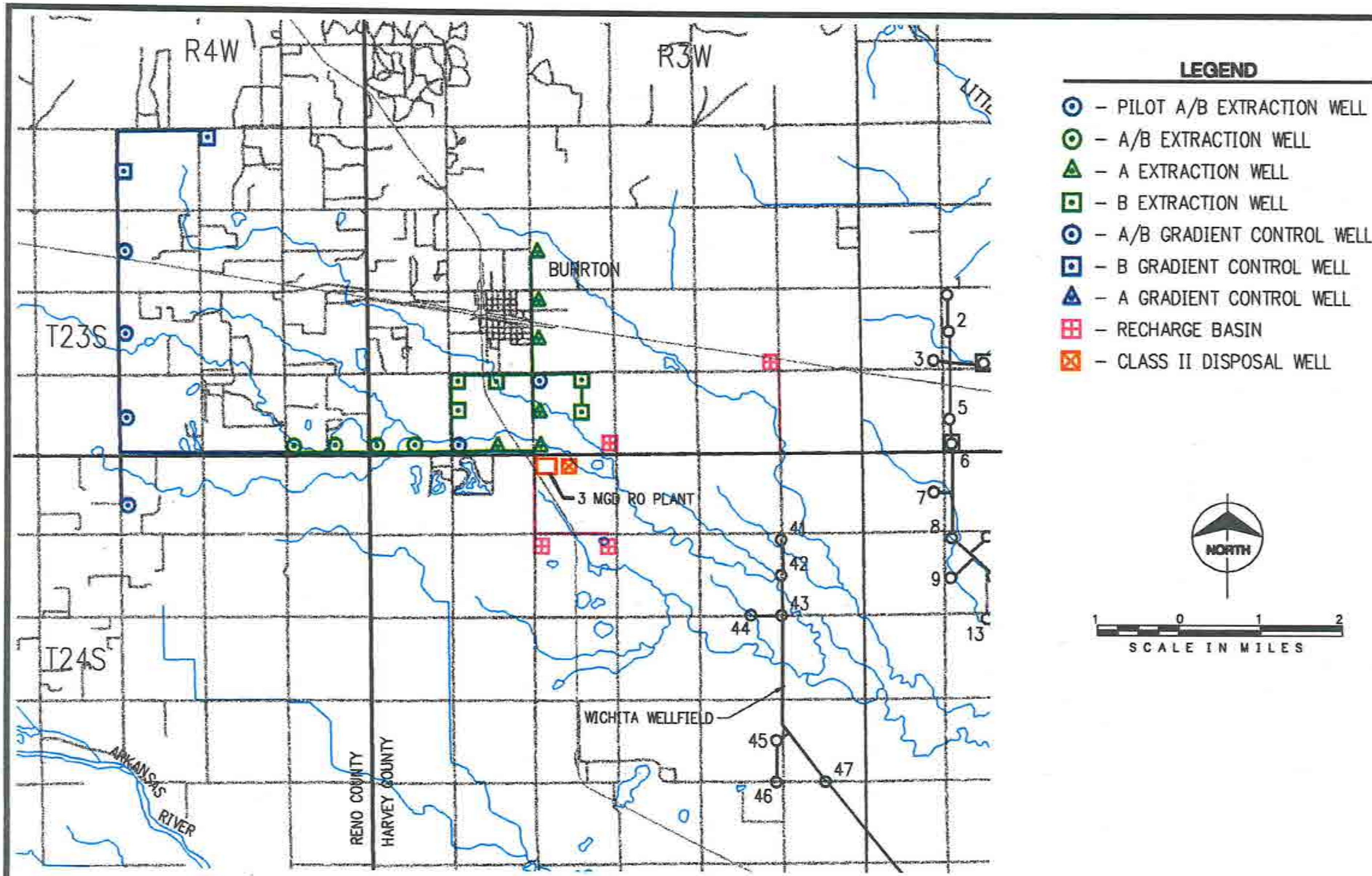

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Figure IV – 5

ALTERNATIVE 5

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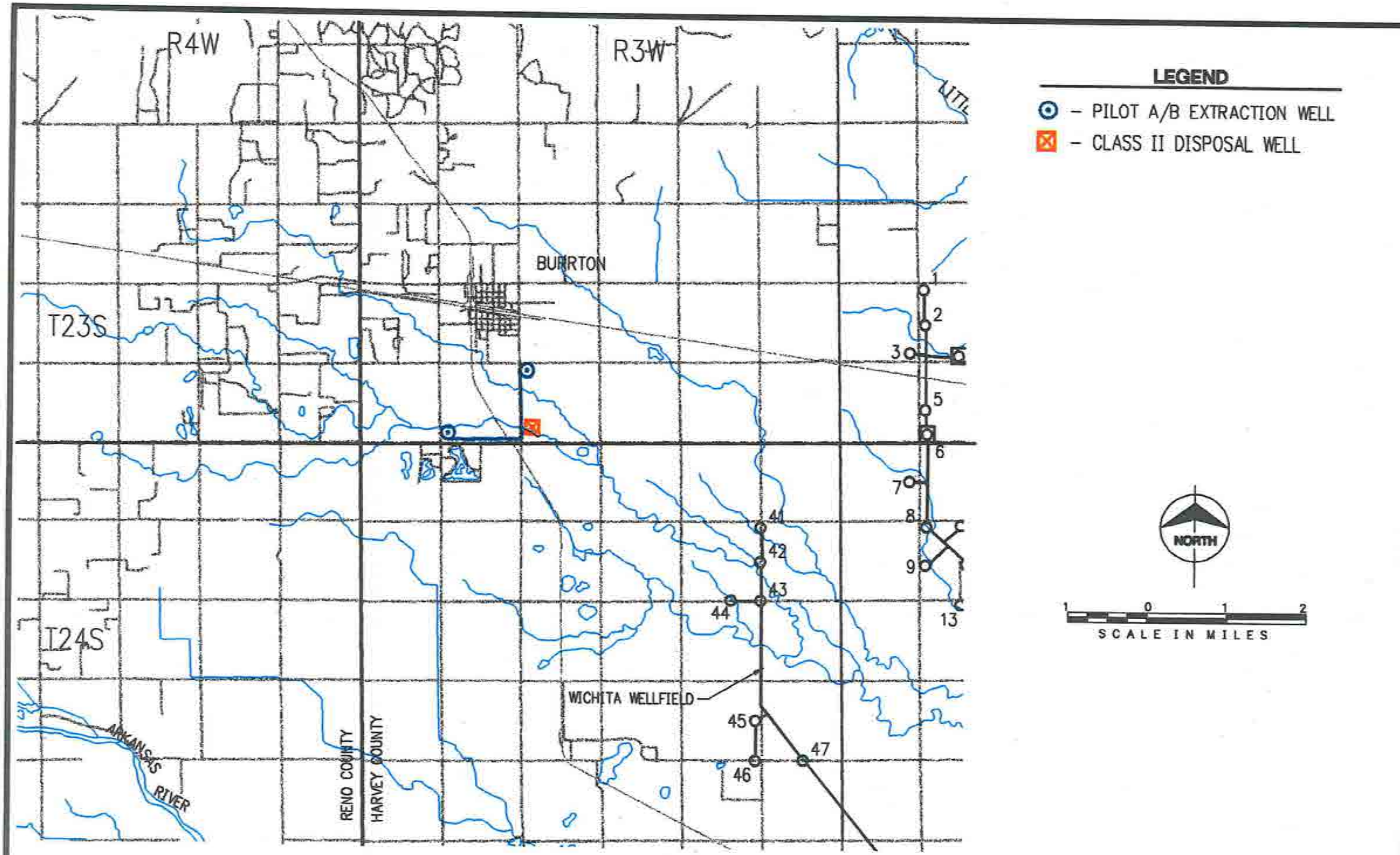

Burns & McDonnell

Figure IV – 6

ALTERNATIVE 6
PILOT PROJECT

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B. CAPITAL COSTS FOR ALTERNATIVE COMPONENTS

The alternatives listed above have many similar components including extraction wells, gradient control wells, Class II disposal wells and an RO treatment plant. Costs for these components are used to develop capital and present worth costs for each alternative. The individual components are discussed below.

1. Extraction Wells

Capital costs for the extraction wells corresponding to the management simulations using plume interceptor wells (2a, 2b and 2c) are summarized in Table IV-1. Based on the modeling simulations, six wells are screened into Level A, five wells are screened into Level B, and six wells are screened into both levels. The approximate pumping rate for wells screened in one level is 90 gpm and wells screened in both levels are estimated to pump 180 gpm from the groundwater plume. Pump motor sizing vary among the remedial alternatives because of different pumping head requirements; therefore additional motor costs to achieve required pumping heads at the wells was priced separately for each alternative.

The capital costs for the plume control interceptor wells are the same for all remedial alternatives.

2. Gradient Control Wells

Capital cost estimates for gradient control wells, screened into Level B and/or C are given in Table IV-2. The maximum pumping rate modeled in the management simulations using gradient control wells include two wells at 200 gpm screened in Level B and four wells pumping 400 gpm screened in both Levels B and C. Motor costs were adjusted for each alternative because of different pumping head requirements.

Table IV -1

CAPITAL COSTS FOR EXTRACTION WELLS:

	Quantity	Unit Cost	Total cost
90 and 180 gpm wells, 50' and 200' TD			
Construction, development and testing of 10" x 24" x 50' well,* (screened into Level A)	6	22,000	132,000
Construction, development and testing of 10" x 24" x 200' well* (screened into Level B)	5	49,000	245,000
Construction, development and testing of 12" x 30" x 200' well* (screened into Level A and B)	6	54,000	324,000
Provision and installation of 10" pitless unit, pad and piping to valve vault	17	5,500	94,000
Provision and installation of 12" pitless unit, pad and piping to valve vault	17	7,000	119,000
Provision and installation of valve vault with 4" piping including a flow meter, check valve and gate valve.	17	7,000	119,000
Provision and installation of 90 gpm at 50' TDH submersible pump, discharge column, cable, check valve, airline and control panel, motor, (50 TD).	6	5,600	34,000
Provision and installation of 90 gpm at 50' TDH submersible pump, discharge column, cable, check valve, airline and control panel, motor (200' TD).	5	6,200	31,000
Provision and installation of 180 gpm at 50' TDH submersible pump, discharge column, cable, check valve, airline and control panel, motor (200' TD).	6	7,500	45,000
SUBTOTAL:			1,143,000
Contingency (20%)			229,000
SUBTOTAL:			1,372,000
Engineering, Legal, Surveying, etc.			206,000
TOTAL CAPITAL COSTS:			1,578,000

Note: * wells designated by (casing diameter) x (bore diameter) x (total well depth)

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Table IV -2

CAPITAL COSTS FOR GRADIENT CONTROL WELLS:

	Quantity	Unit Cost	Total cost
200 and 400 gpm wells, 200' average TD			
Construction, development and testing of 12" x 30" x 200' well,* 200 gpm, (screened into Level B)	2	53,000	106,000
Construction, development and testing of 18" x 36" x 200' well,* 400 gpm, (screened into Level B and C)	4	70,000	280,000
Provision and installation of 12" pitless unit, pad and piping to valve vault	2	7,000	14,000
Provision and installation of 18" pitless unit, pad and piping to valve vault	4	15,750	63,000
Provision and installation of valve vault with 4" piping including flow meter, check valve and gate valve	2	7,000	14,000
Provision and installation of valve vault with 6" piping including flow meter, check valve and gate valve	4	7,500	30,000
Provision and installation of 200 gpm at 200' TDH submersible pump, discharge column, cable, check valve, airline and control panel, motor, (200 TD).	2	11,500	23,000
Provision and installation of 400 gpm at 200' TDH submersible pump, discharge column, cable, check valve, airline and control panel, motor, (200 TD).	4	16,500	66,000
SUBTOTAL:			**596,000**
Contingency (20%)			119,000
SUBTOTAL:			**715,000**
Engineering, Legal, Surveying, etc.			107,000
TOTAL CAPITAL COSTS:			**822,000**

Note: * wells designated by (casing diameter) x (bore diameter) x (total well depth)

Capital costs for the gradient control pipeline network are also included in Alternatives 1 through 5.

3. Class II Disposal Wells

Capital costs for a Class II disposal wells are summarized in Table IV-3. These costs, provided by KCC, are actual costs incurred for the recent construction of a Class II well in Kansas. Operation and maintenance (O & M) costs for a disposal well are minimal. Annual maintenance is estimated at about $200 for periodic leak testing and minor repairs. Operational costs are included within overall well field and pipeline operations for each alternative that uses this disposal option (Alternatives 1, 2, 4 and 5).

The costs developed in Table IV-3 are for deep well disposal of extraction well discharge water in Alternatives 1, 2 and 6 and for RO concentrate in Alternatives 4 and 5.

4. Reverse Osmosis Plant

Estimated capital costs for an RO plant to treat extraction well discharge water is listed in Table IV-4. The feed water flow rate is assumed to reflect the total pumpage of 2,000 gpm from the extraction well network of modeled in the management simulations. The RO plant is sized at a nominal 3 MGD. Assumptions regarding the costs are as follows:

- RO feed water characteristics are indicated in Table III-1 and conservatively assumed to remain constant throughout the life of the remediation.

- high pressure membranes to treat RO concentrate are not cost effective at a 3 MGD plant influent feed rate.

Table IV -3

CAPITAL COSTS FOR CLASS II DISPOSAL WELLS:

	Quantity	Unit Cost	Total cost
Class II disposal wells (4)			
Drilling costs:			
Contractor supervision	40	300	12,000
Footage (7" diameter)	15,800	6	99,000
Labor	480	185	89,000
Drill bits, etc.			20,000
Rental--slimhole drill string			42,000
Water hauling services			10,000
Excavation/earthwork			4,000
Drilling fluid disposal/waste management			
Closed mud system			20,000
Drilling fluids/technical services			19,000
Trucking			6,000
Wireline services			10,000
Cement materials and services			
Conductor string			4,000
Surface pipe string			10,000
Casing			
20" conductor string			6,000
13 3/8" surface pipe			46,000
Trucking			8,000
Completion Costs:			
Casing			
7" disposal string			134,000
4 1/2" plastic lined tubing			76,000
Misc. connections and heads			6,000
Misc. Services			8,000
Trucking			8,000
Labor for casing crew			10,000
Cement materials and services (7" disposal string)			10,000
Wireline services (cased hole)			12,000
Acid treatment			8,000
Oil field rental--misc. equipment			6,000
Supervision	20	350	7,000
Site restoration			2,000
Surge tank/misc. equipment:			
250-barrel closed top fiberglass tank (12' x 12')	12	3,350	40,000
Concrete pad/misc. plumbing fittings			6,000
SUBTOTAL:			**738,000**
Contingency (5%)			37,000
SUBTOTAL:			**775,000**
Engineering, Legal, Surveying, etc.			116,000
TOTAL CAPITAL COSTS:			**891,000**

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Table IV -4

CAPITAL COSTS FOR 3-MGD REVERSE OSMOSIS PLANT:

	Quantity	Unit Cost	Total cost
Greensand filters			270,000
R.O. membrane,cartridge filters, pressure units and booster pumps			1,400,000
RO startup and warranty			45,000
Degasifier			400,000
Cleaning system for R.O. membranes			65,000
Chlorine contact chamber/clearwell			100,000
Chemical storage/feed system			150,000
Process piping			200,000
SCADA, instrumentation and controls (RO plant only)			100,000
Building and sitework (incl. HVAC, plumbing and electrical)			200,000
Yard piping and raw water piping and connections			300,000
High service pumps with motors to municipal water supply system			50,000
SUBTOTAL:			**3,280,000**
Contingency (20%)			656,000
SUBTOTAL:			**3,936,000**
Engineering, Legal, Surveying, etc.			590,000
TOTAL CAPITAL COSTS:			**4,526,000**

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- principle components of the system are as itemized in Table IV-4.

- the permeate will meet all applicable federal and state drinking water standards and will have a maximum chloride effluent concentration of 125 mg/L.

- a pilot study will not be necessary, due to the tested and proven status of the RO process for the treatment of chloride impacted feed water.

The assumed water treatment goal of 125 mg/L was chosen so that the total pumpage from the extraction well could be accommodated by potential water suppliers in the area as a public water supply. A lower treatment goal will be required if the blended water is used for recharge or if a higher permeate flow rate is deemed necessary.

An RO plant is a component of both Alternatives 4 and 5.

5. Recharge Basins

Only Alternative 5 includes recharge basins to replenish the Equus Beds Aquifer with water derived from this plume control project. The effects of recharge basins on plume migration control were modeled, assuming 75 % of the 2,000 gpm of pumpage from the gradient control well system was recharged downgradient of the plume. Four recharge basins, each occupying one-half acre of area and receiving 375 gpm of flow were located where groundwater currently has chloride concentrations of 100 to 200 mg/L. Recharge into the proposed areas may require less stringent treatment goals, due to the higher ambient chloride concentrations in the aquifer at this location, in comparison to ambient chloride levels closer to the Wichita Well Field.

Capital costs for the four recharge basins are itemized in Table IV-5. These costs are based on information from the Equus Beds Groundwater Recharge Demonstration Project.

6. <u>Pipelines</u>

Each of the remedial alternatives includes a unique network of transmission and header pipelines, depicted in Figures IV-1 through IV-6. The primary criterion of limiting flow velocities to 5 ft/sec or less was utilized for the selection or pipeline diameters for the preliminary plans of each alternative. The preferred pipeline material for header pipes and transmission mains is PVC, for standard diameters up to 16 inches, due to superior corrosion resistance and cost effectiveness. Unit costs for common pipe sizes were determined from rates currently applicable to the study area. These unit costs are summarized below:

- 6" PVC: $8/LF
- 8" PVC: $10/LF
- 12" PVC: $15/LF
- 16" PVC: $20/LF
- 24" ductile iron pipe (DIP): $50/LF

The easement unit cost was estimated at $1/ft. Costs for air release valves were estimated under the assumption that one valve would be placed per quarter mile of pipeline.

C. REMEDIAL ALTERNATIVES COST ESTIMATES

Estimates of capital costs, operation, maintenance, energy and equipment replacement costs are itemized for each of the remedial alternatives 1 through 5 in Tables IV-6 through IV-10. These costs for the pilot system are shown in Table IV-11. Annual costs, involving energy and labor rates, were obtained from sources local to the study area.

Table IV -5

CAPITAL COSTS FOR RECHARGE BASINS:

	Quantity	Unit Cost	Total cost
Earthwork and materials for recharge basins (4 @ 1/2 acre/basin)	4	70,000	280,000
Slope protection	4	50,000	200,000
Control building (at each basin)	4	20,000	80,000
Control building piping and valves	4	15,000	60,000
Yard piping	4	10,000	40,000
Misc. site work	4	10,000	40,000
Monitoring well (6 wells, 70' deep)	6	2,500	15,000
Piezometers (15 piezometers, 70' deep)	15	700	11,000
Fencing	4	1,800	7,000
SCADA	4	25,000	100,000
SUBTOTAL:			**833,000**
Contingency (20%)			167,000
SUBTOTAL:			**1,000,000**
Engineering, Legal, Surveying, etc.			150,000
TOTAL CAPITAL COSTS:			**1,150,000**

COSTS.WK4

Table I V -6

ALTERNATIVE COST SUMMARY

ALTERNATIVE 1:

Extraction wells to Class II disposal wells
Gradient control wells to Wichita well field

Capital costs:	Quantity	Unit Cost	Total cost
Extraction wells (17)			1,143,000
Class II disposal wells (4)			738,000
Gradient control wells (6)			596,000
Add for required total dynamic pump head (TDH)			59,000
Transmission and header pipelines:			
4" diameter pipe and fittings	10,600	7	74,000
6" diameter pipe and fittings	21,200	8	170,000
8" diameter pipe and fittings	21,200	10	212,000
12" diameter pipe and fittings	18,600	15	279,000
16" diameter pipe and fittings	48,000	20	960,000
4" connections from wells to header (50' per well)	550	7	4,000
6" connections from wells to header (50' per well)	1,150	8	9,000
Easement	121,300	1	121,000
Air/Vacuum valves	92	2,000	184,000
Interconnect/meter vault			35,000
Operations building (incl. HVAC, electrical, plumbing, site work)			100,000
SCADA, instrumentation and controls			730,000
SUBTOTAL:			**5,414,000**
Contingency (20%)			1,083,000
SUBTOTAL:			**6,497,000**
Engineering, Legal, Surveying, etc.			975,000
TOTAL CAPITAL COSTS:			**7,472,000**
Annual O & M Costs:			
Well field and pipeline operations (1 FT supervisor, 2 FT assistants)			225,000
Well field and pipeline maintenance (by contractors)			30,000
Energy costs ($0.10/kwh)			275,000
Lab fees			8,000
TOTAL O & M COST ($/yr.)			**538,000**

COSTS.WK4

Table IV - 7

ALTERNATIVE 2:

Extraction wells to Class II disposal wells
Gradient control wells to Arkansas River

Capital costs:	Quantity	Unit Cost	Total cost
Extraction wells (17)			1,143,000
Class II disposal wells (4)			738,000
Gradient control wells (6)			596,000
Add for required total dynamic pump head (TDH)			30,000
Transmission and header pipelines:			
4" diameter pipe and fittings	10,600	7	74,000
6" diameter pipe and fittings	21,200	8	170,000
8" diameter pipe and fittings	18,600	10	186,000
12" diameter pipe and fittings	21,200	15	318,000
16" diameter pipe and fittings	17,500	20	350,000
4" connections from wells to header (50' per well)	550	7	4,000
6" connections from wells to header (50' per well)	1,150	8	9,000
Easement	90,800	1	91,000
Air/Vacuum valves	69	2,000	138,000
Outfall			25,000
Operations building (incl. HVAC, electrical, plumbing, site work)			100,000
SCADA, instrumentation and controls			730,000
SUBTOTAL:			**4,702,000**
Contingency (20%)			940,000
SUBTOTAL:			**5,642,000**
Engineering, Legal, Surveying, etc.			846,000
TOTAL CAPITAL COSTS:			**6,488,000**
Annual O & M Costs:			
Well field and pipeline operations (1 FT supervisor, 2 FT assistants)			225,000
Well field and pipeline maintenance (by contractors)			30,000
Energy costs ($0.10/kwh)			190,000
Lab fees			8,000
TOTAL O & M COST ($/yr.)			**453,000**

Table IV - 8

ALTERNATIVE 3:

Extraction and gradient control wells to Arkansas River

Capital costs:	Quantity	Unit Cost	Total cost
Extraction wells (17)			1,143,000
Gradient control wells (6)			596,000
Add for required total dynamic pump head (TDH)			37,000
Transmission and header pipelines:			
4" diameter pipe and fittings	10,600	7	74,000
6" diameter pipe and fittings	18,600	8	149,000
8" diameter pipe and fittings	10,600	10	106,000
12" diameter pipe and fittings	18,600	15	279,000
16" diameter pipe and fittings	21,200	20	424,000
24" diameter pipe and fittings	20,000	50	1,000,000
4" connections from wells to header (50' per well)	550	7	4,000
6" connections from wells to header (50' per well)	1,150	8	9,000
Easement	101,300	1	101,000
Air/Vacuum valves	77	2,000	154,000
Outfall			25,000
Operations building (incl. HVAC, electrical, plumbing, site work)			100,000
SCADA, instrumentation and controls			730,000
SUBTOTAL:			**4,931,000**
Contingency (20%)			986,000
SUBTOTAL:			**5,917,000**
Engineering, Legal, Surveying, etc.			888,000
TOTAL CAPITAL COSTS:			**6,805,000**
Annual O & M Costs:			
Well field and pipeline operations (1 FT supervisor, 2 FT assistants)			225,000
Well field and pipeline maintenance (by contractors)			30,000
Energy costs ($0.10/kwh)			240,000
Lab fees			8,000
TOTAL O & M COST ($/yr.)			**503,000**

Table IV - 9

ALTERNATIVE 4:

Extraction and gradient control wells to RO (with bypass) to Wichita
RO concentrate to disposal well

Capital costs:	Quantity	Unit Cost	Total cost
Extraction wells (17)			1,143,000
Class II disposal well (1)			185,000
Gradient control wells (6)			596,000
Add for required total dynamic pump head (TDH)			35,000
3-MGD R.O. plant with bypass			3,280,000
Transmission and header pipelines:			
4" diameter pipe and fittings	8,000	7	56,000
6" diameter pipe and fittings	16,000	8	128,000
8" diameter pipe and fittings	13,300	10	133,000
12" diameter pipe and fittings	18,600	15	279,000
16" diameter pipe and fittings	21,200	20	424,000
24" diameter pipe and fittings	26,500	50	1,325,000
4" connections from wells to header (50' per well)	550	7	4,000
6" connections from wells to header (50' per well)	1,150	8	9,000
Easement	105,300	1	105,000
Air/Vacuum valves	80	2,000	160,000
Interconnect/meter vault			35,000
SCADA, instrumentation and controls			730,000
SUBTOTAL:			**8,627,000**
Contingency (20%)			1,725,000
SUBTOTAL:			**10,352,000**
Engineering, Legal, Surveying, etc.			1,553,000
TOTAL CAPITAL COSTS:			**11,905,000**
Annual O & M Costs:			
Well field and pipeline O & M			705,000
Well field and pipeline maintenance (by contractors)			30,000
Energy costs ($0.10/kwh)			240,000
Lab fees			8,000
R.O. plant O & M			660,000
TOTAL O & M COST ($/yr.)			**1,643,000**

Table IV - 10

ALTERNATIVE 5:

Extraction and gradient control wells to RO (with bypass) to recharge
RO concentrate to disposal well

Capital costs:	Quantity	Unit Cost	Total cost
Extraction wells (17)	17		1,143,000
Class II disposal well (1)	1		165,000
Gradient control wells (6)	6		596,000
Add for required total dynamic pump head (TDH)			35,000
3-MGD R.O. plant with bypass			3,280,000
Recharge basins (4)	4		280,000
Transmission and header pipelines:			
4" diameter pipe and fittings	8,000	7	56,000
6" diameter pipe and fittings	16,000	8	128,000
8" diameter pipe and fittings	29,200	10	292,000
12" diameter pipe and fittings	29,200	15	438,000
16" diameter pipe and fittings	21,200	20	424,000
24" diameter pipe and fittings	5,300	50	265,000
4" connections from wells to header (50' per well)	550	7	4,000
6" connections from wells to header (50' per well)	1,150	8	9,000
Easement	110,600	1	111,000
Air/Vacuum valves	84	2,000	168,000
Interconnect/meter vault			35,000
SCADA, instrumentation and controls			855,000
SUBTOTAL:			**8,304,000**
Contingency (20%)			1,661,000
SUBTOTAL:			**9,965,000**
Engineering, Legal, Surveying, etc.			1,495,000
TOTAL CAPITAL COSTS:			**11,460,000**
Annual O & M Costs:			
Well field, pipeline, and recharge basin O & M			870,000
Well field and pipeline maintenance (by contractors)			30,000
Energy costs ($0.10/kwh)			260,000
Lab fees			8,000
R.O. plant O & M			660,000
TOTAL O & M COST ($/yr.)			**1,828,000**

COSTS.WK4

Part V - Conclusions and Recommendations

PART V

CONCLUSIONS AND RECOMMENDATIONS

A. CONCLUSIONS

Brine contamination of the Equus Beds Aquifer in the Burrton Intensive Groundwater Use Control Area (IGUCA) covers a very large area. Approximately 22 square miles in the vicinity of Burrton is underlain by water with chloride levels greater than the MCL of 250 mg/L. Review of previous investigations, analysis of computer modeling for this study, discussions from the first project meeting with the local water authorities provide the basis for the following conclusions.

1. The water quality of the Equus Beds Aquifer is generally very good with chloride levels ranging from 10 to 55 mg/L.

2. Several natural and man-caused sources of salt threaten the water quality of the Equus Beds Aquifer. Overdevelopment and pumping in excess of the aquifer's safe yield has increased migration of existing high chloride water and has induced movement of outside sources of chlorides into the aquifer.

3. The leading edge of the Burrton chloride plume has reached several of the municipal wells operated by the City of Wichita. Wichita Well Nos. 43 and 44 have chloride levels above 100 mg/L.

4. If no action is taken, the plume will continue to migrate to the east and deeper into the aquifer in response to the groundwater flow and aquifer pumping stresses from irrigation and municipal demands. The peak chloride concentrations will decrease as the plume is diluted. However, modeling shows that large areas, in excess of 11 square miles, will continue to have groundwater with chloride levels in excess of 500 mg/L.

5. Even with the evaluated plume management scenarios, a large chloride plume with concentrations of 100 to 1000 mg/L will reach downgradient irrigation wells and more of the City of Wichita's wells in the next 50 years; however, the size of the impacted area will be reduced and the peak concentration will be reduced. The use of interceptor wells as determined by the US Bureau of Reclamation (USBR) only slows the impact of the plume on the City of Wichita Well Field. This is also the case with many of the management simulations modeled for this study which combines the use of gradient control wells and extraction wells.

   The management simulations using recharge basins downgradient of the present location of the plume provide little additional control over the chloride plume at the pumping rates evaluated because of large downgradient pumping stresses.

6. The maximum salt removal rate modeled by the USBR interceptor well layout (simulation 2c) and the management simulation 4c from this study (a combination of gradient control wells and extraction wells) is about 130,000 to 170,00 tons of salt over the 50 year simulation period. This represents approximately 7 to 9 percent of the salt that is estimated to have entered the aquifer from past oil field operations. Even with these simulations, significant migration of the plume toward the Wichita Well Field will continue to occur.

7. Higher pumping rates will remove greater amounts of salt; however, increased groundwater gradients will induce high chloride water flow from the Arkansas River or additional subsurface brine from the deeper bedrock valley. In some modeling scenarios, the amount of salt removed was completely replaced by salt entering the model area from the Arkansas River or subsurface sources with no net chloride improvement in the study area.

8. Previous modeling studies by the Kansas Geological Survey (KGS) concluded that pumping rates of 90 cubic feet per second (cfs) for three years followed by pumping rates of 55 cfs thereafter was required to completely control the Burrton oil field brine (Heidari, 1987). Pumping and disposing of this volume of contaminated groundwater was considered in the KGS report as not practical and would induce substantial flow of high chloride water from the Arkansas River and bedrock valleys.

9. The aquifer recharge program being investigated by the City of Wichita with state and federal agencies is found to be a significant factor in the management of the Burrton salt plume. Higher water levels provided through such a recharge project reduces the groundwater gradient which will slow the migration of the salt plume. Slower migration of the plume allows extraction wells to be more efficient. The higher water levels will also reduce the amount of salt entering the system from the Arkansas River during remediation pumping.

10. Downgradient pumping stresses appears to have significant impacts on the migration of the Burrton salt plume. Management or reduction of downgradient pumping may be a second significant factor in the management of the Burrton chloride plume, although specific scenarios were not evaluated as part of this investigation. Replacement of some downgradient municipal with upgradient diversion from the gradient control well is expected to have additional significant impacts in slowing the plume.

11. The gradient control wells used in modeled management simulations withdraw groundwater with relatively low chloride concentrations. Initially, the chlorides are approximately 300 to 500 mg/L and decline to ambient levels. This water requires little, if any treatment prior to beneficial use. Extraction wells may have

initial concentrations of near 1,500 mg/L. Use of this water would require treatment or dilution with larger amounts of potable water prior to use.

12. The conclusions from the groundwater modeling for this study are based on the model flow assumptions which use relatively wide-spread data and are based the updated USBR model. Additional geologic data, aquifer information, and chloride concentration information would allow the model to be modified to more accurately represent the study area for detailed design of remediation systems.

13. Reverse osmosis (RO) is the preferred treatment technology for removing chlorides from the groundwater in the study area. The low concentration permeate from the RO process could be used beneficially by blending with a municipal water supply or as recharge into the aquifer to act as a barrier to the migration of the plume. The RO concentrate could be disposed of through deep well injection.

14. The treated water from the RO process and/or dilute brine from gradient control or extractions wells could be blended into a municipal water system for beneficial use. The City of Wichita is a near by municipal water supply system with flows large enough to dilute moderate amounts of high chloride water for potable use. The City of Wichita has a desired finished-water chloride limit of 150 mg/L. Up to 1.4 MGD of plume control water at a chloride concentration of 500 mg/L could be used and not exceed the desired maximum finished water chloride limit. Larger amounts of plume control water could be used if the average chloride content is reduced by blending or treatment.

15. Recharge of the RO permeate into the aquifer would act to raise water levels in the aquifer and act as a barrier to slow migration of the plume. Water recharged into areas along the edge of the plume may require less treatment due to the

higher ambient chloride concentrations in these areas resulting in lower treatment costs than if the water were used for a public water supply. Recharge at locations close to the plume would have a greater effect on local groundwater; however, only recharging treated high-chloride water or gradient control water will not have a significant impact in controlling the plume.

16. Five remediation alternatives were considered to develop comparative cost estimates. The capital costs for the five evaluated alternatives ranged from $6,486,000 to $11,903,000. The costs for these alternatives were based on the 4,000 gpm total pumping rate for the management of the plume.

17. A pilot study would provide additional information about the aquifer and the chloride concentrations both in the plume and in the extracted groundwater. In addition, the pilot study would provide information to develop a larger scale RO treatment system, including membrane performance, chemical feed rates, optimal flux, and ion rejection and recovery rates. Siting of the pilot study facilities will require test hole drilling to determine final well location and design.

B. RECOMMENDATIONS

The following recommendations are based on the results of the modeling analysis of the Burrton oil brine area that was derived form the USBR's initial contaminant transport model of the region and updated to currently available information (1996); information provided in other reports concerning the proposed Equus Beds Groundwater Recharge Demonstration Project; and cost estimates developed for selected management simulations.

1. Because of the magnitude of the Burrton salt plume and the delicate balance of pumping with the inflow of salt from outside sources, phased implementation of

remedial measures is recommended so that hydrogeologic system response can be evaluated before larger and more expensive systems are installed.

2. Initial analysis indicates that relatively costly remediation measures would remove 8 to 10 percent of the salt contamination. This evaluation is based on modeling assumptions and data that widely space across the study area. In order to develop additional data and operation experience to determine more precisely how the aquifer will react to clean up measures, a pilot remediation program is recommended.

   A pilot study with two recovery wells, one injection well, and six to eight additional monitoring wells is highly recommended. The additional monitoring wells are to evaluate aquifer response to the cleanup efforts and determine the suitability of using the recovered water in a public water supply system. The new monitoring wells should be constructed to standards that allow for organic water quality analyses. Additionally, detailed water quality analysis of the recovered water from the pilot operation will allow a more detailed evaluation of treatment required and costs for expanded remediation systems.

2. After a period of operation and data collection, the model should be refined for a more detailed evaluation, siting, and design of expanded remediation systems.

3. Additional modeling studies should be conducted to evaluate supplemental plume management strategies involving alternate operation of the City of Wichita Well Field and surrounding irrigators to reduce the impacts of the salt plume. Options include, but are not limited to the following:
   - reduced Wichita Well Field pumping near the plume and obtaining greater amounts of water for their demands at more distant well field locations.

- replacing larger volumes of Wichita Well Field water with plume management water (and reduced well field pumping).
- Seasonal (or long time period cyclic) pumping of the Wichita wells near the plume to allow long periods of water level recovery.

- Reduced pumping for all irrigators and municipal supplies to the aquifers "safe yield" capacity as defined by the GMD2.

4. Additional water quality analysis is recommended. Complete safe drinking water analyses are needed to confirm the water will meet EPA drinking water standards and could be used in municipal systems. It is recommended that six to ten "environmental quality" monitoring wells be constructed throughout the Burrton salt plume and water quality samples collected for the complete EPA drinking water quality analysis.

5. Begin discussions with the federal, state and local agencies to explore the regulatory constraints of using the plume control water for municipal use. Specific issues include:
   - water rights for the plume control wells and impacts to the existing municipal water rights.
   - funding of the facilities.
   - water quality monitoring requirements.

* * * * *

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