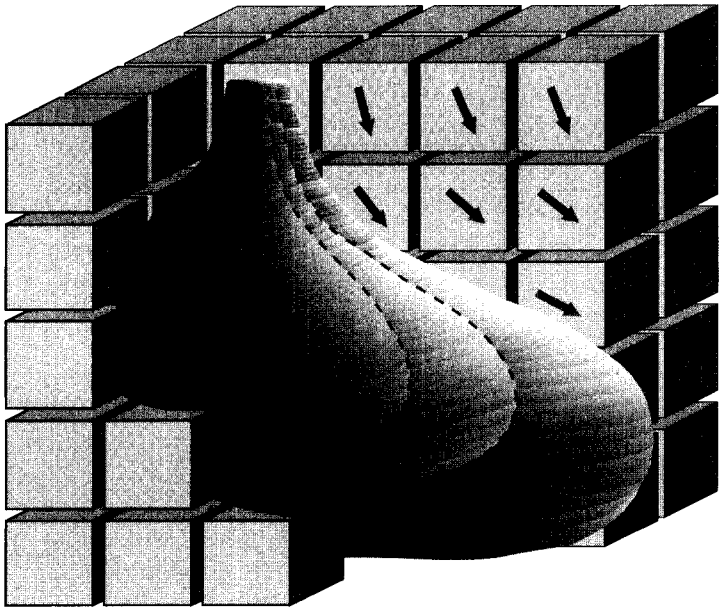
Numerical simulation of the movement of sulfate in ground water in southwestern Salt Lake Valley, Utah

Prepared by the U.S. Geological Survey



Technical Publication No. 110-D
State of Utah
DEPARTMENT OF NATURAL RESOURCES
1996

This report was prepared as a part of the Statewide cooperative water-resource investigation program administered jointly by the Utah Department of Natural Resources, Division of Water Rights and the U.S. Geological Survey. The program is conducted to meet the water administration and water-resource data needs of the State as well as the water information needs of many units of government and the general public

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NUMERICAL SIMULATION OF THE MOVEMENT OF SULFATE IN GROUND WATER IN SOUTHWESTERN SALT LAKE VALLEY, UTAH

By P.M. Lambert
U.S. Geological Survey

Prepared by the
United States Geological Survey
in cooperation with the
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CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATED PARAMETER UNITS

Multiply	By	To obtain
acre-foot (acre-ft)	4,047	square meter
acre-foot per year (acre-ft/yr)	1,233	cubic meter
foot (ft)	0.3048	meter
gallon (gal)	3.785	liter
mile (mi)	1.609	kilometer
square mile (mi^2)	2.59	square kilometer

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.

The generic units of mass (M), length (L), and time (T) are used to define the units of variables in some equations presented in this report. To be distinguished from the generic unit abbreviation for length (L), the standard U.S. Geological Survey abbreviation for liter "L" is not used in this report and has been replaced by "*l*."

Dissolved-phase chemical concentration is reported only in metric units, in milligrams per liter (mg/*l*). Milligrams per liter is a unit expressing the solute per unit volume (liter) of water. One thousand micrograms per liter is equivalent to 1 milligram per liter. For concentrations less than 7,000 milligrams per liter, the numerical value is about the same as for concentrations in parts per million. Solid-phase chemical concentration is reported in milligrams per kilogram (mg/kg). Density is reported in grams per cubic centimeter (g/cm^3). The Freundlich isotherm constant (K_f) is reported in liters squared per milligram squared (l^2/mg^2).

Numerical Simulation of the Movement of Sulfate in Ground Water in Southwestern Salt Lake Valley, Utah

By P.M. Lambert
U.S. Geological Survey

ABSTRACT

Contaminated ground water characterized by high concentrations of dissolved solids and dissolved sulfate, and in areas, by low pH and elevated concentrations of metals, is present near public-supply wells in southwestern Salt Lake Valley. To provide State officials and water users with information concerning the potential movement of contaminated ground water to points of withdrawal in the area, an analysis of solute transport using computer models was done by the U.S. Geological Survey in cooperation with the Utah Department of Natural Resources, Division of Water Rights, and local municipalities and water users.

A three-dimensional solute-transport model was developed and coupled with an existing ground-water flow model of Salt Lake Valley to simulate the movement of dissolved sulfate in ground water in southwestern Salt Lake Valley. Development and calibration of the transport model focused mainly on sulfate movement downgradient from the Bingham Creek Reservoirs and the South Jordan evaporation ponds east of the mouth of Bingham Canyon. Estimates of transport parameters were adjusted during a calibration simulation representing conditions during 1965-93. After calibration, the transport model was used to simulate future sulfate movement for 1994-2043.

Because of uncertainty in estimated transport-parameter values, three projection transport simulations incorporating a range of probable parameter values were done to evaluate future sulfate movement and changes in sulfate concentrations at selected public-supply wells. These projection simulations produced a possible range of computed transport rates and patterns. In general, the projection simulations indicated movement of the sulfate plume east of the Bingham Creek Reservoirs toward public-supply wells northeast of the reservoirs and then eastward toward the Jordan River. Ground water with high concentrations of sulfate east of the South Jordan evaporation ponds is simulated as moving west to east under the Jordan River toward public-supply wells during the final 25 years of the simulation period. An increase in sulfate concentration from 200 mg/*l* in 2006 to 4,100 mg/*l* in 2022 was the largest simulated increase at public-supply wells northeast of the reservoirs. An increase in sulfate concentration from 150 mg/*l* in 2024 to 340 mg/*l* in 2043 was the largest simulated increase at public-supply wells in south-central Salt Lake Valley just east of the Jordan River.

INTRODUCTION

Salt Lake Valley is the main population and industrial center in the State of Utah (fig. 1). Maintenance of an adequate supply of water suitable for domestic use is one of the most important factors in sustaining the current population and industrial activity and in allowing for continued economic growth. State officials are in need of detailed information concerning the occurrence and potential movement of poor-quality ground water with high dissolved-solids concentration to anticipate and prevent migration of this water to points of withdrawal and thus to better manage development of the ground-water system. In July 1990, the U.S. Geological Survey, in cooperation with the Utah Department of Natural Resources, Division of Water Rights, and the Utah Department of Environmental Quality, Division of Water Quality, began a study of ground-water flow and solute transport in Salt Lake Valley. Local municipalities and water users also participated in the study, including the Salt Lake City Corporation, The Salt Lake County Water Conservancy District, the Taylorsville-Bennion Improvement District, the Granger-Hunter Improvement District, Murray City, the City of South Salt Lake, Kearns Improvement District, and Kennecott Utah Copper.

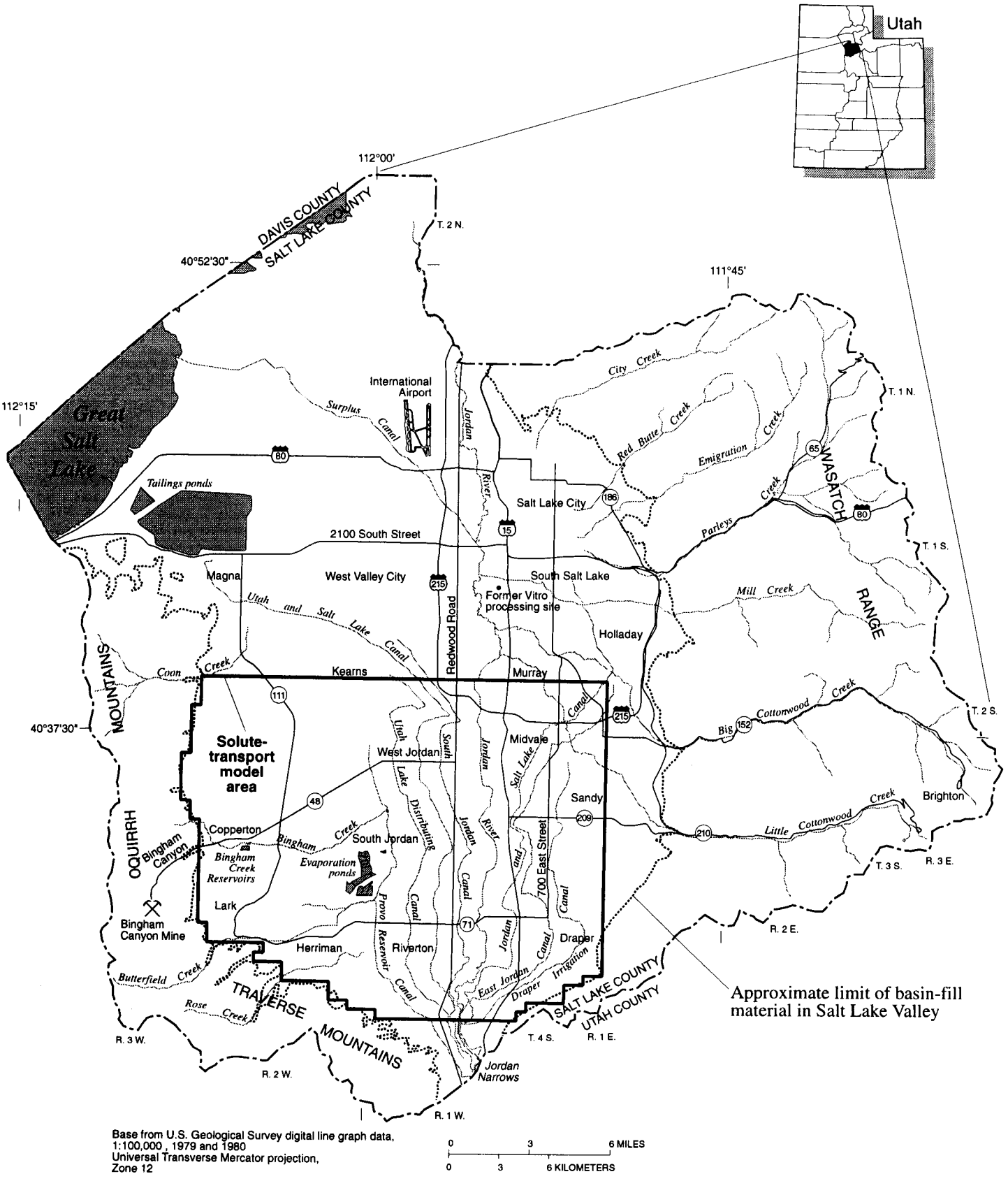

Figure 1. Location of Salt Lake Valley, Utah, and solute-transport model area.

The study was done to better define the ground-water flow system in Salt Lake Valley and to evaluate the movement of poor-quality ground water in areas of withdrawal for public supply. The approach to these objectives included the development of ground-water flow and solute-transport computer simulations to enable planners to better understand the direction and rate at which ground water and solutes move within the system under different stress conditions created by ground-water pumping.

Southwestern Salt Lake Valley is the site of the Bingham Canyon Mine, one of the world's largest open-pit copper mines (Kennecott Utah Copper, 1992, p. 7). The mine currently is operated by Kennecott Utah Copper Corporation, which, together with its predecessors, has been mining copper and other metals from Bingham Canyon since the early 1900s (Kennecott Utah Copper, 1992, p. 7). A wastewater and leachate-collection system that includes evaporation ponds and containment reservoirs east of the canyon mouth has received mine drainage and waters from ore-leaching facilities since 1935. As a result of seepage from these structures and other smaller sources, ground water with high dissolved-solids and high sulfate concentration, and in some areas, low-pH and elevated metal concentration, is present in the principal aquifer of the ground-water flow system between the canyon mouth and the Jordan River. Analyses of ground-water flow done previously during this study (Lambert, 1995b) indicate that the quality of water withdrawn at some wells in the area may be affected by future movement of contaminated ground water east of the canyon mouth.

In the component of the study documented in this report, a three-dimensional solute-transport model coupled with a ground-water flow model of Salt Lake Valley (Lambert, 1995a) was used to simulate the movement of dissolved sulfate in the basin-fill ground-water flow system in southwestern Salt Lake Valley (fig. 1). The simulation analysis was done to provide more information on the future movement of contaminated ground water relative to points of withdrawal in southwestern Salt Lake Valley. A secondary objective of the analysis was to evaluate the limitations of the applied modeling approach and to identify areas where additional data will be needed to improve future modeling analyses.

Purpose and Scope

This report documents the simulation of sulfate movement in the basin-fill ground-water flow system of southwestern Salt Lake Valley. A three-dimensional solute-transport model of southwestern Salt Lake Valley was used in conjunction with a regional, ground-water flow model of Salt Lake Valley (Lambert, 1995a). The report describes the development and calibration of the transport model and the use of the model to investigate potential future changes in sulfate concentration in ground water in southwestern Salt Lake Valley.

The scope of analysis was limited to the simulation of sulfate movement in the basin-fill ground-water flow system in southwestern Salt Lake Valley. Sulfate was selected as the modeled solute because sulfate is a principal constituent of the primary contaminant sources in the area and is the focus of ongoing remediation and feasibility studies by Kennecott Utah Copper (1995). Also, a substantial amount of data is available that defines sulfate concentration in ground water in the area.

The transport model documented in this report represents an area in southwestern Salt Lake Valley that extends east from the foot of the Oquirrh Mountains to about 5 mi east of the Jordan River (fig. 1). Transport-model parameters estimated on the basis of data collected previously in the area were evaluated in a calibration simulation representing conditions during 1965-93. Future sulfate movement in the area was simulated for 1994-2043 on the basis of projected increased withdrawals for public supply in Salt Lake Valley.

As with all computer simulations, the modeling approach used in this analysis is based on limiting assumptions. This report discusses some of the inherent difficulties in the simulation of solute transport using a computer model of a complex ground-water flow system. Limitations of the analysis, the significance of simulation results, and the need for future work are discussed at the end of the report.

Previous Work

Previous work done during this study includes the quantification of chemical properties of water and hydrologic properties of the basin-fill ground-water flow system in Salt Lake Valley (Thiros, 1992; Thiros,

1995), and the development and calibration of a three-dimensional, finite-difference, numerical model to simulate ground-water flow in that system (Lambert, 1995a). The ground-water flow model of Salt Lake Valley (Lambert, 1995a) was developed as a tool for planning and water management throughout the valley, and for use in ground-water flow and solute-transport analyses in selected subregions of the valley.

The ground-water flow model of Salt Lake Valley (Lambert, 1995a) was applied previously during this study to simulate future hydrologic conditions on the basis of projected increases in withdrawals for public supply (Lambert, 1995b). These simulations were used in conjunction with a particle-tracking program to identify areas of the valley where the future movement of poor-quality ground water may affect the quality of water withdrawn from public-supply wells. The results of the analysis indicated that zones of the aquifer that supply water to some public-supply wells in southwestern and south-central Salt Lake Valley may contain contaminated ground water associated with historic mining activities in Bingham Canyon.

The particle-tracking method used in that analysis (Lambert, 1995b) allowed for the definition of pathways from areas of poor-quality ground water in southwestern Salt Lake Valley to points of discharge at nearby public-supply wells. The analysis did not, however, provide information quantifying the solute concentration in ground water that may be withdrawn from the wells in the future, or the effects of mixing, dispersion, and chemical reactions on future solute movement relative to those wells.

Description of Transport-Model Area

Hydrogeologic Setting

The area represented by the transport model (fig. 1) is part of the regional ground-water flow system of Salt Lake Valley. The boundaries of the transport-model area were selected for the purpose of simulating sulfate movement and do not represent hydrologic boundaries. Thus, the hydrogeologic setting for the transport-model area is discussed as part of the regional setting of Salt Lake Valley.

Salt Lake Valley is a structural graben filled with semiconsolidated and unconsolidated basin-fill material. The valley is surrounded by the Oquirrh Mountains on the west, the Traverse Mountains on the south, the Wasatch Range on the east and northeast, and Great Salt Lake on the northwest (fig. 1). The surrounding mountains are composed of consolidated rock with negligible primary porosity but with substantial secondary porosity in the form of fractures and solution openings (Hely and others, 1971, p. 10). Geophysical data indicate that the consolidated-rock base of the valley is an irregular surface formed by the tops of fault blocks (Cook and Berg, 1961, p. 81), with inner-valley grabens containing, in some places, more than 4,000 ft of unconsolidated and semiconsolidated basin-fill material (Mattick, 1970, fig. 6).

The basin-fill material in Salt Lake Valley consists mostly of sediments of Tertiary and Quaternary age and is made up of clay, silt, sand, gravel, tuff, and lava. The history and sequence of deposition of these sediments were described by Marine and Price (1964) as "extremely complex" as a result of the different mechanisms of deposition and erosion working in the valley and in the adjacent mountains throughout time. The valley received lake deposits in areas that were inundated by a series of ancient lakes, the most extensive of which was Lake Bonneville. As lakes dried, lake sediments were subjected to stream erosion, and previously inundated areas received stream-channel and flood-plain deposits. Alluvial fans formed along the mountain fronts at canyon mouths; glacial and mudrock flow deposits also were laid down at the margins of the valley. As lakes reappeared and filled the valley, lacustrine deposition again predominated. The changes in depositional environments in the valley as lakes formed, dried up, and reappeared has resulted in the interlayered lacustrine, alluvial, and glacial sediments that make up most of the basin fill today, with coarse-grained sediments common near the mountains and finer-grained sediments in the low-lying areas of the central and northern parts of the valley.

The basin-fill ground-water flow system in Salt Lake Valley (fig. 2) has been described by Hely and others (1971, p. 107) as consisting of (1) a confined (artesian) aquifer, (2) a deep unconfined aquifer between the confined aquifer and the mountains, (3) a shallow unconfined aquifer overlying the confined aquifer, and (4) local unconfined perched aquifers. The confined aquifer is in the central and northern parts of the valley and consists of deposits of clay, silt, sand, and gravel. In the confined aquifer, beds and lenses of fine-grained material of slight to moderate permeability tend to confine water in beds of sand and gravel. Other than in the northern part of the valley, the beds and lenses are relatively thin and discontinuous; therefore,

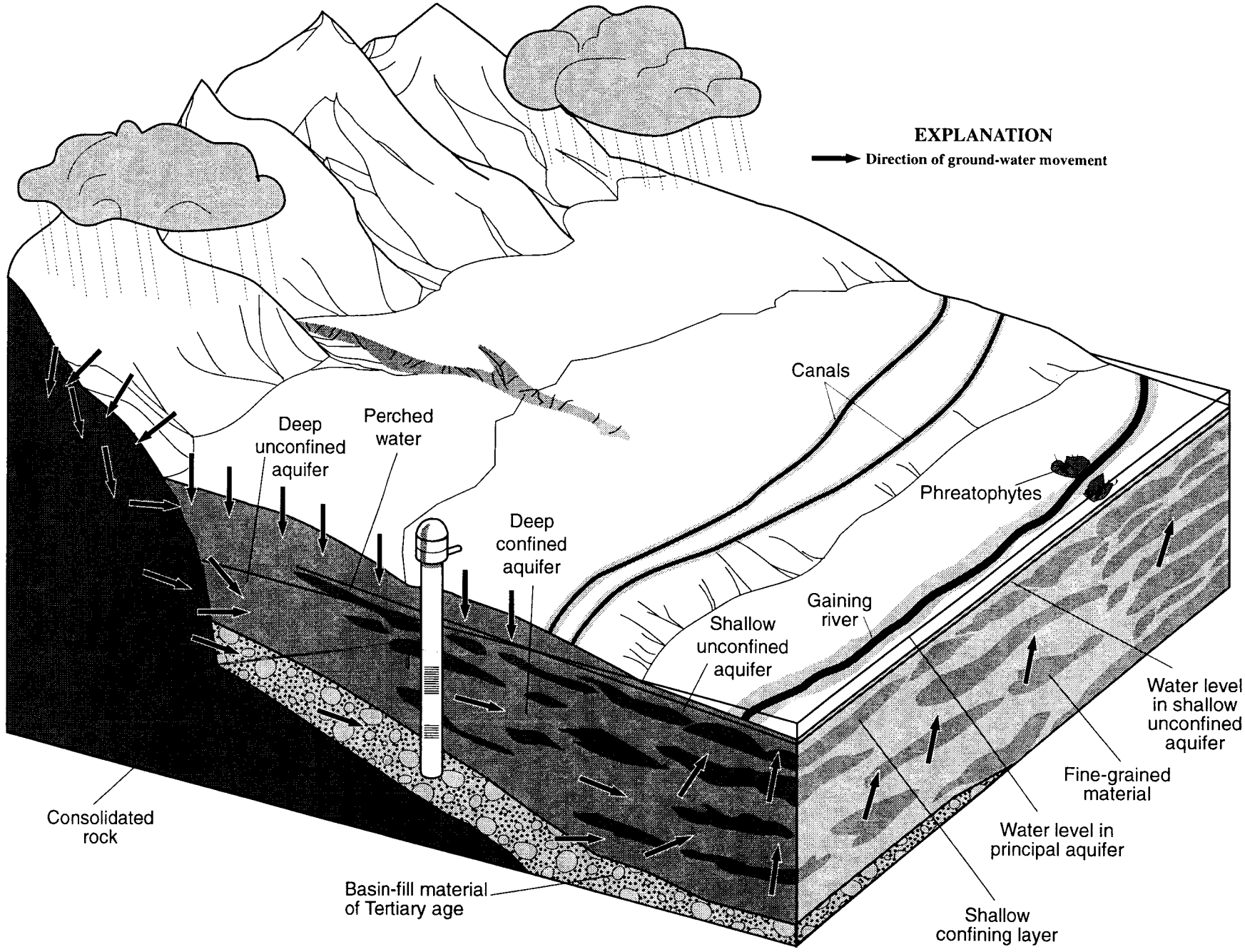

Figure 2. Generalized block diagram showing the basin-fill ground-water flow system in Salt Lake Valley, Utah. Modified from Hely and others, (1971).

there is appreciable movement of water between more permeable beds of sand and gravel. Overlying the confined aquifer are sediments of relatively low permeability that consist of interfingering and overlapping layers and lenses of clay, silt, and fine-grained sand of Quaternary age. Although the continuity of these sediments varies, in many areas they act as a single bed, the top of which generally lies within 100 ft of land surface. For the purpose of discussion, this bed of fine-grained sediments is referred to in this report as the shallow confining layer. Near the margins of the valley where confining sediments are absent, ground water is unconfined. The confined zone beneath the shallow confining layer and the unconfined zones near the margins of the valley make up the principal aquifer of the ground-water flow system, which is the main source of most of the ground water withdrawn from wells in the valley (Hely and others, 1971, p. 109; Waddell and others, 1987a, p. 5).

The thickness of basin-fill material of Quaternary age that makes up most of the principal aquifer in Salt Lake Valley ranges from 0 to 2,000 ft. Quaternary-age basin fill of the principal aquifer generally overlies less permeable, semiconsolidated sediments of Tertiary and pre-Tertiary age (Arnow and others, 1970, p. D257). In some areas of the valley, including the southwestern part of the valley, more permeable Tertiary-age basin fill yields water to wells. Water-yielding zones in Tertiary-age basin fill are included with the principal aquifer in this study.

In southwestern Salt Lake Valley, the deep unconfined aquifer occurs mainly in alluvial-fan deposits that extend from the foot of the Oquirrh Mountains to the South Jordan evaporation ponds about 4.5 mi to the east. These sediments rest unconformably on volcanic rocks of Tertiary age and locally on bedrock of Paleozoic age (Kennecott Utah Copper, 1992). Results of test drilling by Kennecott Utah Copper (1992, p. 59) indicate that the alluvial-fan deposits consist of two types of material, one consisting mainly of Paleozoic quartzite and sandstone clasts and the other made up mostly of volcanic detritus. The quartzite/sandstone deposits are found throughout the southwestern area west of the evaporation ponds and generally overlie volcanic fan deposits where they exist. Observations made during test drilling in the area suggest that the volcanic fan deposits are, in general, substantially less permeable than the quartzite/sandstone deposits (Kennecott Utah Copper, 1995, p. 17).

The deep unconfined aquifer in southwestern Salt Lake Valley extends laterally into the confined aquifer east of the evaporation ponds where fan and other alluvial deposits grade into finer-grained lacustrine deposits. The shallow unconfined aquifer overlies the confined aquifer and is separated from it by the shallow confining layer. The shallow unconfined aquifer, which is composed mainly of fine-grained sediments, is the source of water for local irrigation but is seldom used to supply water for domestic or industrial purposes because it yields water slowly and generally contains water of poor quality (Seiler and Waddell, 1984, p. 2). Ground water is perched in areas where the water level in the principal aquifer is below the bottom of the shallow confining layer; thus, an unsaturated zone exists between the water table in the principal aquifer and the bodies of perched water above it. Areas of perched water in southwestern Salt Lake Valley are known to occur between Herriman and Riverton.

Recharge to the basin-fill ground-water flow system in Salt Lake Valley is primarily from inflow from consolidated rock at the margins of the valley, seepage from streams and canals with a water-surface altitude higher than that of the water table, infiltration of precipitation on the valley floor, and infiltration of unconsumed irrigation water from fields, lawns, and gardens. Ground water moves from primary recharge areas near the margins of the valley to the central and northern parts of the valley (fig. 2). As ground water moves from the deep unconfined aquifer beneath recharge areas into the confined aquifer, an upward vertical gradient is established. From the axial part of the valley, ground water moves upward in the confined aquifer, into and through the overlying confining layer, and into the shallow unconfined aquifer, where it is discharged mainly into the Jordan River or to drains, is used by riparian vegetation, or evaporates at land surface. Most of the discharge from the basin-fill ground-water flow system in southwestern Salt Lake Valley, other than discharge from wells, is via the shallow unconfined aquifer.

Present-day hydrology in Salt Lake Valley is greatly affected by municipal and industrial use of ground water. A summary of annual ground-water withdrawals from wells during 1931-68 (Hely and others, 1971, fig. 66) indicates a range from 38,000 acre-ft in 1931 to 118,000 acre-ft in 1966. The rate of withdrawals began to level off about 1964 and averaged 107,000 acre-ft/yr during 1964-68 (Hely and others, 1971, p. 140). Increases in ground-water withdrawals during 1987-91 combined with less-than-average

recharge to the ground-water flow system have resulted in water-level declines in the southeastern part of the valley of up to 26 ft for that period (Batty and others, 1993, fig. 11). In southwestern Salt Lake Valley, recharge to the ground-water system has historically included seepage from wastewater and leachate-collection and containment systems, and evaporation ponds east of the mouth of Bingham Canyon. Elimination or modifications to these structures during 1990-94 has eliminated recharge from these sources. The decline in recharge from these sources coupled with increases in withdrawals for public supply in the area has resulted in declines in water levels of up to 22 ft during 1991-93 in the southwestern part of the valley.

History and Pattern of Contamination

Mine waste from the Bingham Canyon Mine has been artificially leached in the area since the early 1930s. The waste rock is located along the east side of the Oquirrh Mountains (fig. 3) and covers a 4.5-mi^2 area that extends south from the town of Copperton (Kennecott Utah Copper, 1992, fig. 2). Untreated wastewater from the leaching process along with mine drainage was discharged into Bingham Creek and the Jordan River until 1937. The wastewater was generally acidic, with high sulfate and metal concentrations. Evaporation ponds were constructed in 1935 about 4.5 mi east of the mouth of Bingham Canyon (the South Jordan evaporation ponds) to store and evaporate waste-rock leach-process water and Bingham Canyon watershed water (Kennecott Utah Copper, 1992, p. 14 and 19) and to prevent discharge of those waters to the Jordan River. Between 1937 and 1965, most if not all surface water discharging from the Bingham Canyon area was disposed of in the evaporation ponds (Dames and Moore, 1988, p. 9). The ponds were initially unlined and constructed on top of permeable material, and much of the water seeped into the ground-water system (Dames and Moore, 1988, p. 9). On the basis of hydrologic studies of the upper Bingham Creek basin, discharge to these ponds was estimated to be about 900 acre-ft/yr prior to 1965 (Dames and Moore, 1988, p. 9). Sulfate concentrations of water in the ponds were estimated by Dames and Moore (1989, table 4) to be as much as 40,000 mg/*l*.

A 20-million-gal reservoir (Small Bingham Creek Reservoir) and a 520-million-gal reservoir (Large Bingham Creek Reservoir) were constructed along the Bingham Creek channel east of Copperton during 1962-65 to store water previously diverted to the South Jordan evaporation ponds. From 1965 to 1988, all the drainage from Bingham Canyon was diverted through the Large Bingham Creek Reservoir. Overflow from the reservoirs in high-flow years was diverted to the evaporation ponds. Seepage from the reservoirs during that time was estimated to range from 1,400 to 2,400 acre-ft/yr (Dames and Moore, 1989, table 4). Typical water quality of the reservoir, reported by Dames and Moore (1988, table 1), indicates a dissolved-solids concentration of 46,600 mg/*l* and a sulfate concentration of 37,800 mg/*l*. A small pond (Cemetery Pond) was constructed in 1984 south of the Small Bingham Creek Reservoir to lime-treat mildly contaminated waters. Losses from the pond occurred from 1984 to 1990; however, the sulfate concentration of the treated water in the pond was estimated to be about 350 mg/*l* (Dames and Moore, 1989, table 3) and seepage from the pond is not considered to have been a major contaminant source.

An additional 1,270 acre-ft of evaporation capacity was constructed during 1983-84 (Dames and Moore, 1988, p. 9) at the South Jordan evaporation ponds for the storage and evaporation of the excess runoff from greater-than-average snowpack that winter. Water was diverted to the evaporation ponds after the construction of the Bingham Creek Reservoirs only during 1972-76 and 1982-86. Discharge to the ponds during those periods consisted of a mixture of untreated and lime-treated runoff and wastewater and was estimated to have an average sulfate concentration of about 3,500 mg/*l* (Dames and Moore, 1989, table 3). Seepage from the evaporation ponds during wet seasons was estimated to average about 250 acre-ft/yr during 1972-76 and about 1,800 acre-ft/yr during 1982-86 (Dames and Moore, 1989, table 4).

Seepage from the Bingham Creek Reservoirs and the South Jordan evaporation ponds along with unquantified losses from a leachate-collection system along the base of the waste-rock dumps (east-side collection system) have been identified to be the most substantial sources of ground-water contamination from mine-waste surface water (Kennecott Utah Copper, 1992, p. 13). There have been numerous smaller mining-related sources of contaminated water in southwestern Salt Lake Valley, including leachate from smaller mine tailings or waste-rock dumps east of the Bingham Mine waste dumps, abandoned mine tunnels, underflow of degraded ground water at the mouth of Bingham Canyon, and direct seepage of degraded water from stream channels.

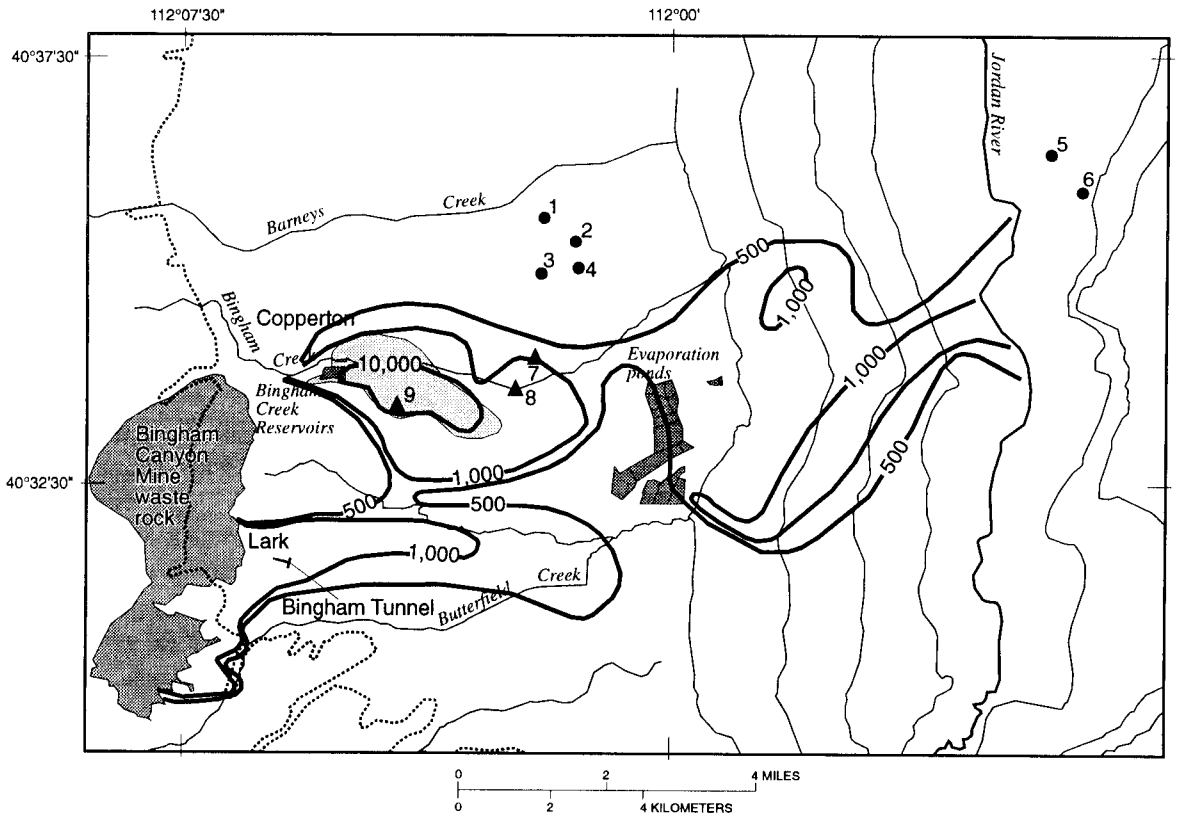

Figure 3. Sulfate concentration in ground water in southwestern Salt Lake Valley, Utah, 1993. Contoured data from Kennecott Utah Copper (written commun., 1994).

Since 1990, Kennecott Utah Copper has established source-control measures to limit the introduction of additional leachate into the ground-water system. In 1990, Small Bingham Creek Reservoir was reconstructed with a double liner and leak-detection system (Kennecott Utah Copper, 1993, p. 3-3). The use of Large Bingham Creek Reservoir was discontinued in 1991 and the reservoir sludge was removed (Kennecott Utah Copper, 1992, p. 16). The reservoir was divided into two zones and lined. The reservoirs are currently in service for flood control. Kennecott also upgraded their systems for collection of leachate and stormwater from waste-rock disposal areas and moved waste rock located outside the collection system to inside the system.

Nonmining-related sources of dissolved sulfate include agricultural-related surface waters. Most of the water used to irrigate fields in southwestern Salt Lake Valley comes from Utah Lake via the Jordan River and a network of canals. Losses from major canals diverting water from the Jordan River are estimated to be about 21,000 acre-ft/yr within the transport-model area west of the Jordan River. Sulfate concentration of canal water is assumed to be similar to that of the Jordan River near the Jordan Narrows, which has been reported to be about 350 mg/l (Hely and others, 1971, table 13). Prior to 1987, some of the irrigation water used in the Herriman area was diverted from Butterfield Creek and the Bingham Tunnel and contained water affected by mining operations. Recharge from unconsumed irrigation water in the Herriman area prior to

1987 was estimated by Dames and Moore (1989, table 3) to be less than 400 acre-ft annually and to have a sulfate concentration of about 1,200 mg/*l*.

As a result of seepage, mainly from the Bingham Creek Reservoirs, the South Jordan evaporation ponds, and the leachate-collection system at the base of the waste-rock dump, ground water in parts of the principal aquifer between the mouth of Bingham Canyon and the Jordan River about 10 mi to the east has been contaminated (Waddell and others, 1987b, p. 18). Three zones, or plumes, of affected ground water defined by ground water with sulfate concentration greater than 500 mg/*l* (a Utah primary drinking-water standard for sulfate) can be identified from 1993 data interpreted by Kennecott Utah Copper (written commun., 1994) (fig. 3): (1) a plume east of the Bingham Creek Reservoirs; (2) a plume east of the South Jordan evaporation ponds; and (3) a plume east of the waste-rock dump near Lark. The sulfate plume east of the Bingham Creek Reservoirs contains a zone of low-pH ground water (fig. 3) characterized by sulfate concentrations that typically exceed 5,000 mg/*l*, and in some areas, by elevated concentrations of trace metals (Kennecott Utah Copper, 1995, p. 24). Sulfate concentrations in ground water measured in 1993 (fig. 3) indicate that the plume originating at the reservoirs has migrated southeast and east toward the evaporation ponds. Sulfate concentrations east of the evaporation ponds and in the Lark area do not generally exceed 1,500 mg/*l* and generally do not contain elevated levels of dissolved metals.

The geochemical environment of southwestern Salt Lake Valley, including the chemical characteristics of contaminated ground water and the mineralogy of the basin-fill aquifer, may have substantially affected the movement of dissolved sulfate in some areas. On the basis of recent work in the area, Kennecott Utah Copper and entities associated with that company have developed a conceptual model describing reactions between low-pH, high-sulfate ground water and the aquifer material that are driven by disequilibrium between water and rock (Kennecott Utah Copper, 1993, p. 1-24 to 1-29 and A2-4 to A2-8). This conceptual model states, in general, that as low-pH, high-sulfate water enters the aquifer, the low-pH water is neutralized by calcite in the aquifer material and calcium is released to solution. Calcium also is released to solution as magnesium in the leachate is exchanged for calcium at exchange sites. Because of the high sulfate concentration of the ground water, the solution becomes oversaturated with gypsum and gypsum precipitates, removing sulfate from solution. Under near-neutral pH conditions where calcite is present, the conceptual model indicates that the solubility of gypsum limits the concentration of dissolved sulfate to 2,000-3,500 mg/*l*. Under these conditions, which prevail throughout most of the transport-model area, sulfate is considered to be conservative and its movement is not assumed to be affected by chemical reactions.

The conceptual model indicates that as the low-pH zone of the plume east of the Bingham Creek Reservoirs (fig. 3) migrates downgradient, it leaves a zone of precipitated gypsum behind. As fresher water moves into the gypsum zone, gypsum is dissolved. In these areas, water quality will likely continue to be affected by the influx of sulfate derived from the dissolution of gypsum.

The conceptual model briefly described above is only a general concept based on limited data. The details of the complex combination of chemical reactions taking place within the sulfate plume east of the Bingham Creek Reservoirs and the effect of those reactions on the migration of the plume are not yet completely understood.

The Movement of Solute in Ground Water

Solutes such as sulfate occurring in ground water as ions or molecules undergo transport by processes termed advection and dispersion (Domenico and Schwartz, 1990, p. 358-370). With advective transport, the direction and rate of solute movement are the same as those of the ground water in which it is dissolved. Solutes, however, tend to spread out from the path predicted by advection. This spreading phenomenon is called dispersion and occurs because of mechanical mixing of fluids and because of molecular diffusion (Freeze and Cherry, 1979, p. 75). Also, a solute may undergo chemical, nuclear, or biological reactions that affect the distribution of solute mass among ions or between the dissolved and solid phases within the aquifer.

In a completely advective system, the rate of solute transport is equal to the average linear velocity of ground water (v), which can be defined by the following form of the Darcy equation:

$$v = -\frac{K}{\theta}\frac{\partial h}{\partial x} \qquad (1)$$

where

K is hydraulic conductivity (L/T),

θ is effective porosity of the aquifer (dimensionless), and

$\partial h/\partial x$ is hydraulic gradient (dimensionless).

The direction of solute movement in a completely advective system is defined by the pathlines of ground-water flow. The basic features of advective transport are shown by the flow net in figure 4a. In an advective system, when solute mass is added to a stream tube defined by ground-water flow lines, it will remain within the stream tube throughout transport (fig. 4a).

The effects of dispersion on the transport of solute added to the stream tube would be to move solute mass into mixing zones ahead of the advective front and outside of the flow tube where the solute was released (fig. 4b). The two-dimensional spreading, both ahead of the advective front and laterally into adjacent flow tubes, is referred to as longitudinal and transverse dispersion, respectively. Dispersion in three dimensions involves spreading in two transverse directions as well as longitudinally.

Dispersion occurs as a result of two different physical processes termed mechanical dispersion and molecular diffusion. Mechanical dispersion is the mixing that occurs as a result of local variations in velocity around some mean velocity of flow (Domenico and Schwartz, 1990, p. 369). The main causes of these variations in velocity are heterogeneities within the aquifer material. Heterogeneities that cause mechanical dispersion may occur at all scales from microscopic variations in pore spacing or geometry to megascopic changes in aquifer materials that have different hydraulic-conductivity values (fig. 5). Molecular diffusion occurs because of mixing caused by molecular motions driven by the thermal kinetic energy of the solute (Domenico and Schwartz, 1990, p. 367). Generally, the effect of molecular diffusion becomes important only in environments where ground-water velocity is very slow and in most field problems is negligible compared to the effect of mechanical dispersion.

Changes in solute concentration in ground water may occur because of chemical reactions that take place entirely within the dissolved phase or because of the transfer of solute mass to or from the solid or gas phases (Freeze and Cherry, 1979, p. 402). The transfer of solute mass by chemical processes from the dissolved phase while flow occurs causes the advance rate of the solute front to be retarded as illustrated in the flow net (fig. 4c). The figure depicts the results of the release of two solutes in a flow tube. One solute is not

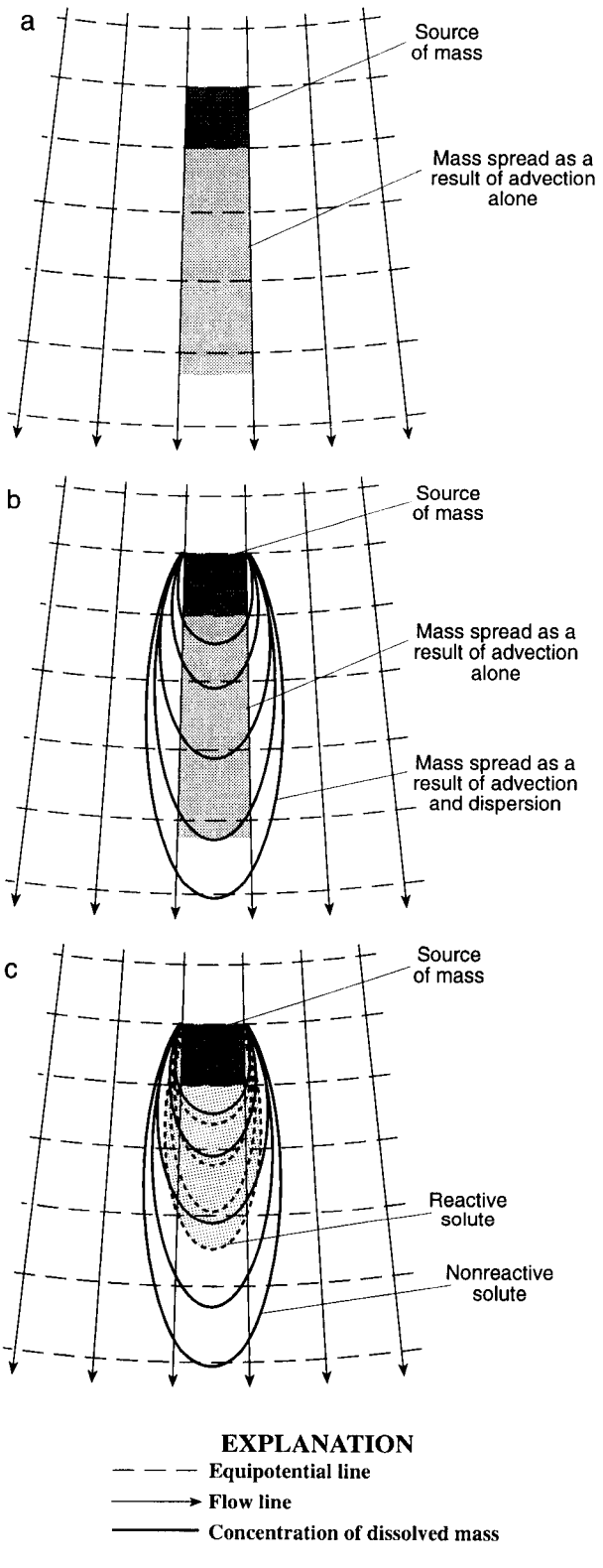

Figure 4. Comparison of mass distribution in an aquifer as a result of (a) advection alone, (b) advection and dispersion, and (c) advection and dispersion, and retardation resulting from chemical reactions. Adapted from Domenico and Schwartz (1990, fig. 10.6).

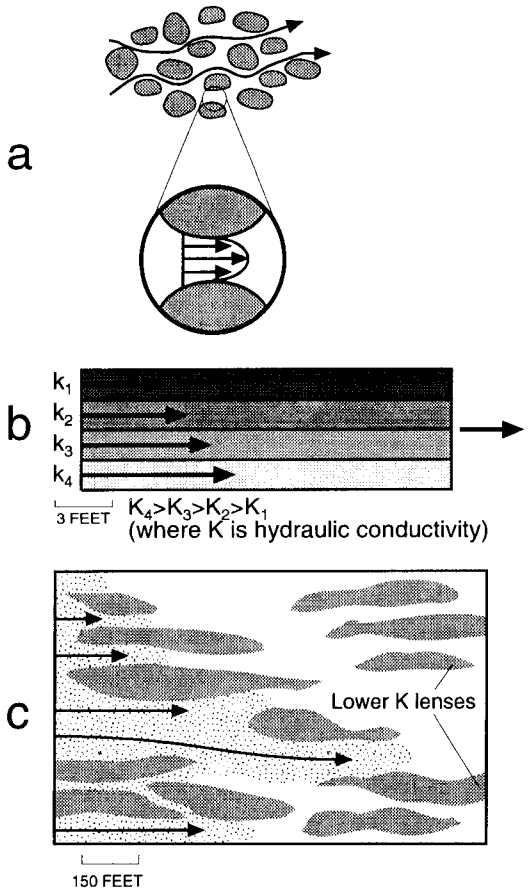

Figure 5. Examples of heterogeneities at different scales within a basin-fill aquifer that give rise to mechanical dispersion. (a) Microscale heterogeneities—variation in pore size and geometry, (b) macroscale heterogeneities—simple layering, and (c) macro to megascale heterogeneities—complex structure of interfingering and overlapping beds and lenses. Adapted from Domenico and Schwartz (1990, fig. 10.9).

affected by chemical processes and moves with water by advection and dispersion. The other undergoes a chemical process that transfers solute mass from the dissolved phase to the solid aquifer material. The water mass represented by the reactive solute travels behind the front of the nonreactive solute. The reactive solute is therefore said to be retarded.

The combined transport processes described above define the movement of solute mass along a flow path and the spreading out of that mass from that flow path. These processes can be represented by using mathematical equations that describe solute transport in a given model domain. Such equations are incorporated in the approach to simulating the movement of dissolved sulfate that is used in this analysis and are discussed in the following sections.

MODELING APPROACH

The modular three-dimensional transport model MT3D (Zheng, 1992) was used in conjunction with a MODFLOW (McDonald and Harbaugh, 1988) ground-water flow model of Salt Lake Valley developed by Lambert (1995a) to simulate sulfate movement in ground water in southwestern Salt Lake Valley. The transport model functions in conjunction with the ground-water flow model by retrieving necessary model-computed hydraulic heads and intercell flows, and sink/source terms from flow-model output. Specified values for transport-model parameters along with flow-model output are incorporated into the transport model to simulate the advection, dispersion of, and chemical reactions of dissolved sulfate within the transport-model area (fig. 1).

Development and calibration of the transport model focused mainly on the simulation of sulfate movement downgradient from the Bingham Creek Reservoirs and the South Jordan evaporation ponds. Data defining seepage of high-sulfate waters from other sources are sparse, and sulfate influx other than from the Bingham Creek Reservoirs, South Jordan evaporation ponds, and natural sources were excluded. Available data defining sources of dissolved sulfate and the approach to model calibration are discussed in detail later in the report.

Three transport-model simulations were made during the analysis: (1) a preliminary transport simulation representing conditions prior to 1965, (2) a calibration transport simulation representing conditions during 1965-93, and (3) a projection transport simulation representing projected conditions for 1994-2043. The transport simulations incorporated output from flow-model simulations representing flow conditions for the same periods.

The preliminary transport simulation was made to estimate the distribution of sulfate in ground water in 1965 for use as starting conditions for the calibration simulation. In the preliminary transport simulation, influx of sulfate from natural sources was simulated for more than 300 years to estimate the distribution of dissolved sulfate in ground water prior to anthropogenic influences on water quality. This simulation also incorporated seepage from the South Jordan evaporation ponds during 1935-64.

Specified transport-model parameters were adjusted during the calibration simulation to match model-computed distributions of dissolved sulfate with distributions defined from field measurements at

selected times during 1965-93. After model calibration was completed, the transport model was used to simulate future sulfate movement for 1994-2043 based on projected increases in municipal pumpage in Salt Lake Valley.

Simulation of Ground-Water Flow

A three-dimensional, finite-difference, numerical model of ground-water flow in the basin-fill groundwater system in Salt Lake Valley (Lambert, 1995a) provided hydraulic-head and intercell-flow data for the solute-transport model in southwestern Salt Lake Valley. Areally, the model grid of the Salt Lake Valley flow model is 94 rows by 62 columns (fig. 6), with each model cell 0.35 mile on a side. Vertically, the aquifer system is divided into seven layers. The shallow unconfined aquifer and the underlying shallow confining layer are represented by one model layer each (model layers 1 and 2, respectively) (fig. 6). The thickness of model layers 1 and 2 is variable, roughly imitating the estimated depth and thickness of the shallow unconfined aquifer and the underlying shallow confining layer. The principal aquifer is divided into five layers (model layers 3 to 7) (fig. 6). Model layers 3 to 5 are each 150 ft thick; the simulated saturated thickness of model layer 3 may vary during problem solution. Model layer 6 is 200 ft thick. Model layer 7 ranges in thickness from 200 ft to more than 1,500 ft. The thickness of each of model layers 4 to 7 is not explicitly incorporated into the flow model but is implicitly incorporated in model input that defines the aquifer properties of those model layers. Active cells in model layers 3 to 7 represent basin-fill material of Quaternary age in the principal aquifer and in some areas include the upper part of the underlying Tertiary unit.

Specified-flux boundaries are used in the flow model to simulate recharge entering the ground-water system as (1) inflow from consolidated rock in areas at the margins of the valley, (2) seepage from streams and major canals, (3) infiltration of precipitation and unconsumed irrigation water, (4) seepage from the Bingham Creek Reservoirs and the South Jordan evaporation ponds, and (5) underflow at Jordan Narrows. Specified-flux boundaries also are used to simulate discharge to wells, canals, and springs. A specified-flux boundary condition allows the flow rate across a given boundary to be specified as a function of location and time. Flow rates across these boundaries are specified in advance in flow-model simulations and, thus, are not affected by simulated events in the ground-water system. The location of specified-flux boundaries used to simulate recharge from consolidated rock and unconsumed irrigation water, and seepage from canals within the transport-model area, is shown in figures 7 and 8.

Head-dependent flux boundaries are used in the flow model to simulate (1) ground-water flow to and seepage from the Jordan River and the lower reaches of its principal tributaries, (2) inflow from consolidated rock at the northern end of the Oquirrh Mountains, (3) discharge from the shallow unconfined aquifer to drains, and (4) discharge by evapotranspiration. A head-dependent flux boundary allows the flow rate across the boundary surface to change in response to changes in water level in the aquifer adjacent to the boundary. Flow rate across the boundary is therefore a function of the water level in the adjacent aquifer and may vary during the simulation period. Only seepage at head-dependent flux boundaries representing the Jordan River and its tributaries is simulated within the transport-model area (fig. 8).

A no-flow boundary at the base of the flow model corresponds to the contact between consolidated rock of pre-Tertiary age and basin-fill material, or to a depth in the basin fill below which sediments were assumed not to contribute substantially to the basin-fill groundwater flow system. On the west and east sides of the flow model, no-flow boundaries correspond to the contact between the consolidated rock of the mountains and the basin fill. The northern border of the flow model approximates a flow line and is also treated as a no-flow boundary. The shore of Great Salt Lake in the northwestern part of the flow model is treated as a constant-head boundary representing the altitude of the lake surface.

The Salt Lake Valley flow model was calibrated to steady-state conditions represented in 1968 and to transient-state conditions during 1969-91 (Lambert, 1995a). Development of these simulations and the ability of the model to reproduce measured hydrologic conditions during these periods are described in detail by Lambert (1995a).

For the transport analysis, the ground-water flow model of Salt Lake Valley (Lambert, 1995a) was applied to simulate hydrologic conditions for the periods defined in the preliminary, calibration, and projection transport simulations. The flow-model simulations were derived from simulations developed previously during the calibration of the ground-water flow model by Lambert (1995a), and from projection simulations

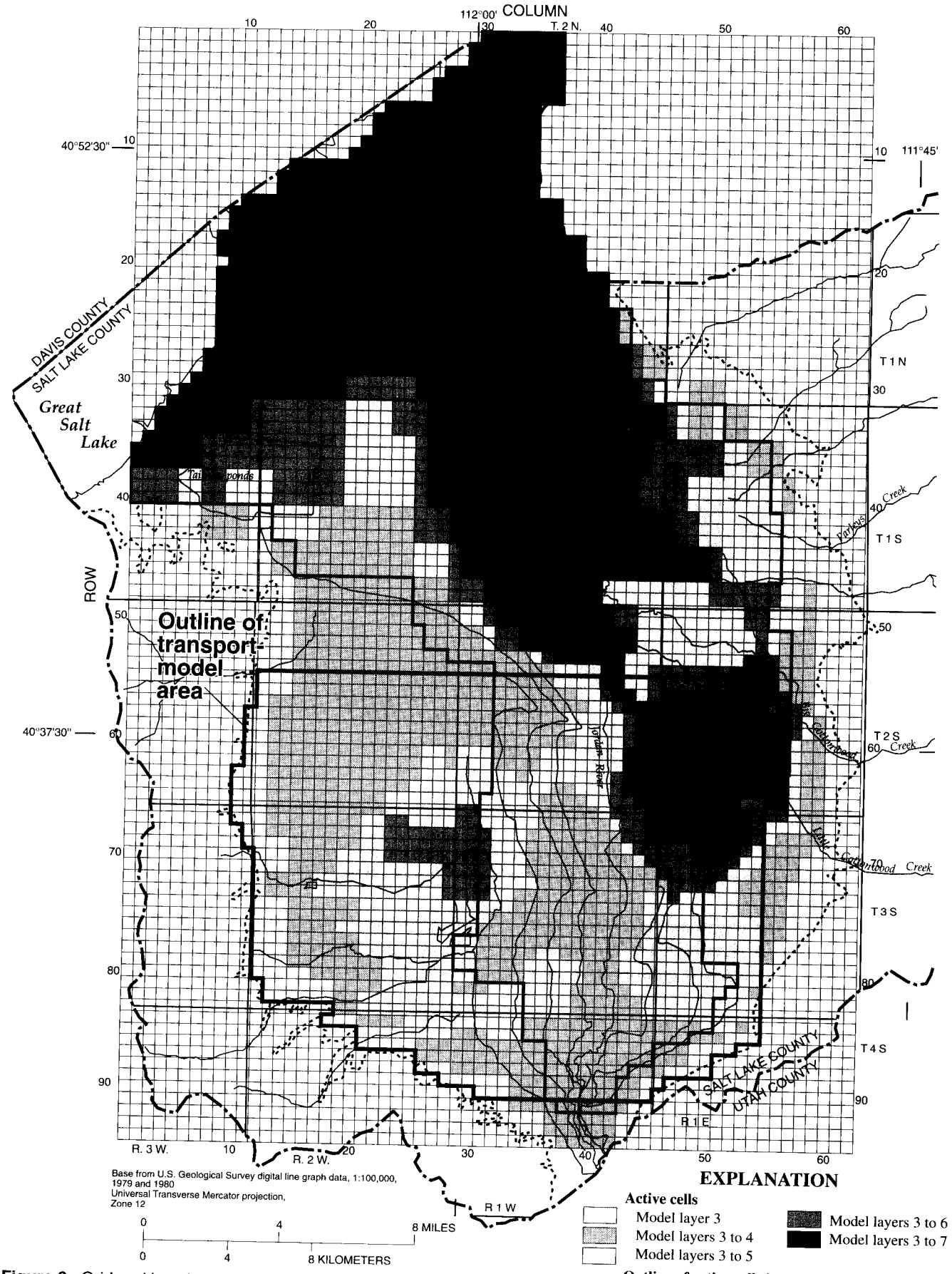

Figure 6. Grid and location of active cells in the ground-water flow model of Salt Lake Valley, Utah, and location of transport-model area.

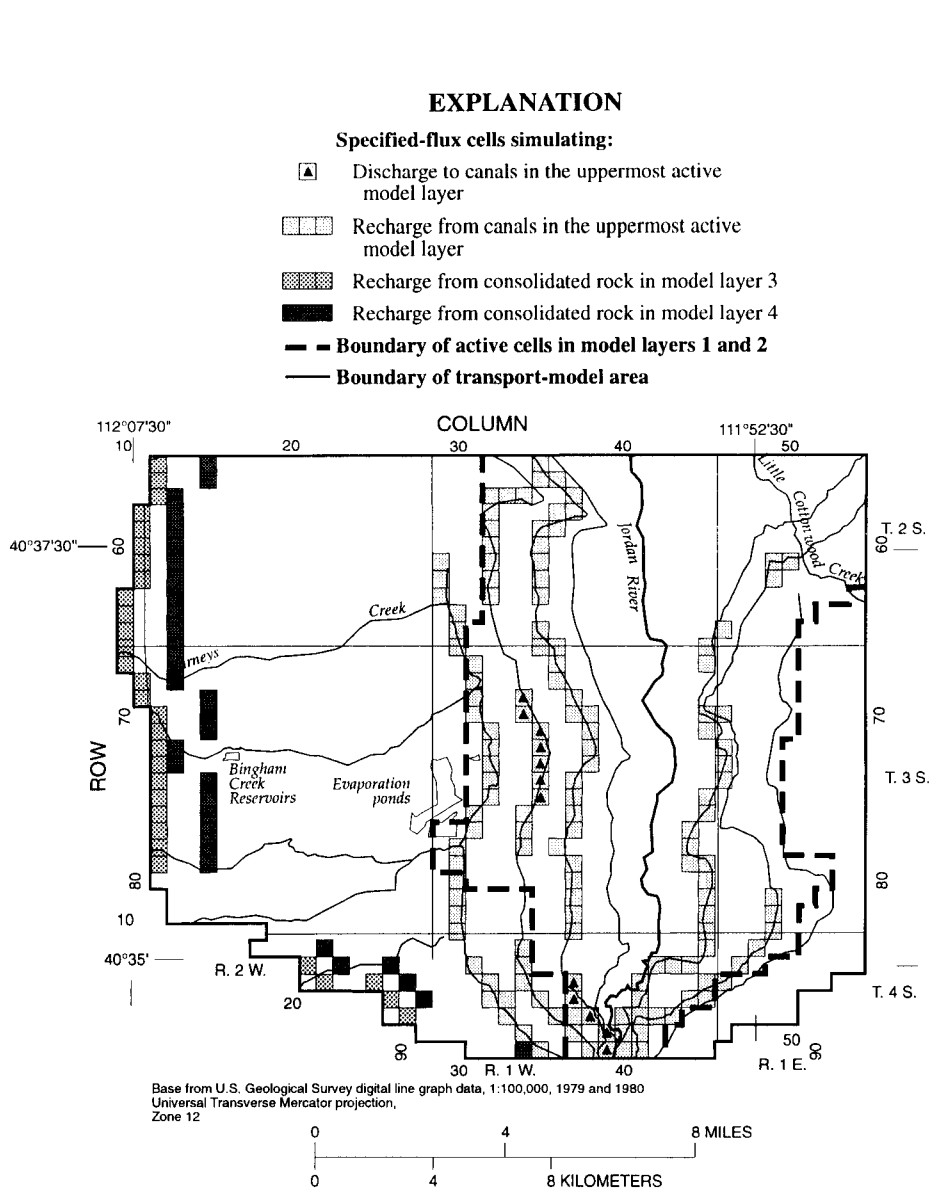

Figure 7. Location of specified-flux cells used to simulate recharge from consolidated rock and seepage at canals within the transport-model area in southwestern Salt Lake Valley, Utah.

used in a previous analysis of ground-water flow near selected wells in the valley (Lambert, 1995b). Output from these flow-model simulations defining hydraulic heads and intercell ground-water flow rates was incorporated in corresponding transport-model simulations.

Two steady-state flow-model simulations representing conditions prior to 1965 were developed to provide head and flow data to the preliminary transport simulation. The first flow-model simulation represented conditions prior to the construction and use of the South Jordan evaporation ponds in 1935. The second simulation represented conditions during 1935-64 and incorporated recharge from seepage from the evaporation ponds. The simulations were derived from the steady-state flow-model simulation by Lambert (1995a) representing flow conditions during 1968 that was used to calibrate the Salt Lake Valley flow model. The 1968 steady-state simulation by Lambert (1995a) was altered to more closely represent conditions during the simulation periods by eliminating simulated recharge from the Bingham Creek Reservoirs and discharge from large pumping wells in the area constructed after 1964. Seepage from the South Jordan evaporation ponds was incorporated in the flow-model simulation that represents conditions during 1935-64.

A flow-model simulation representing conditions during 1965-93 was developed to provide input to the calibration transport simulation. This simulation was derived from a previously developed transient-state simulation (Lambert, 1995a) representing flow condi-

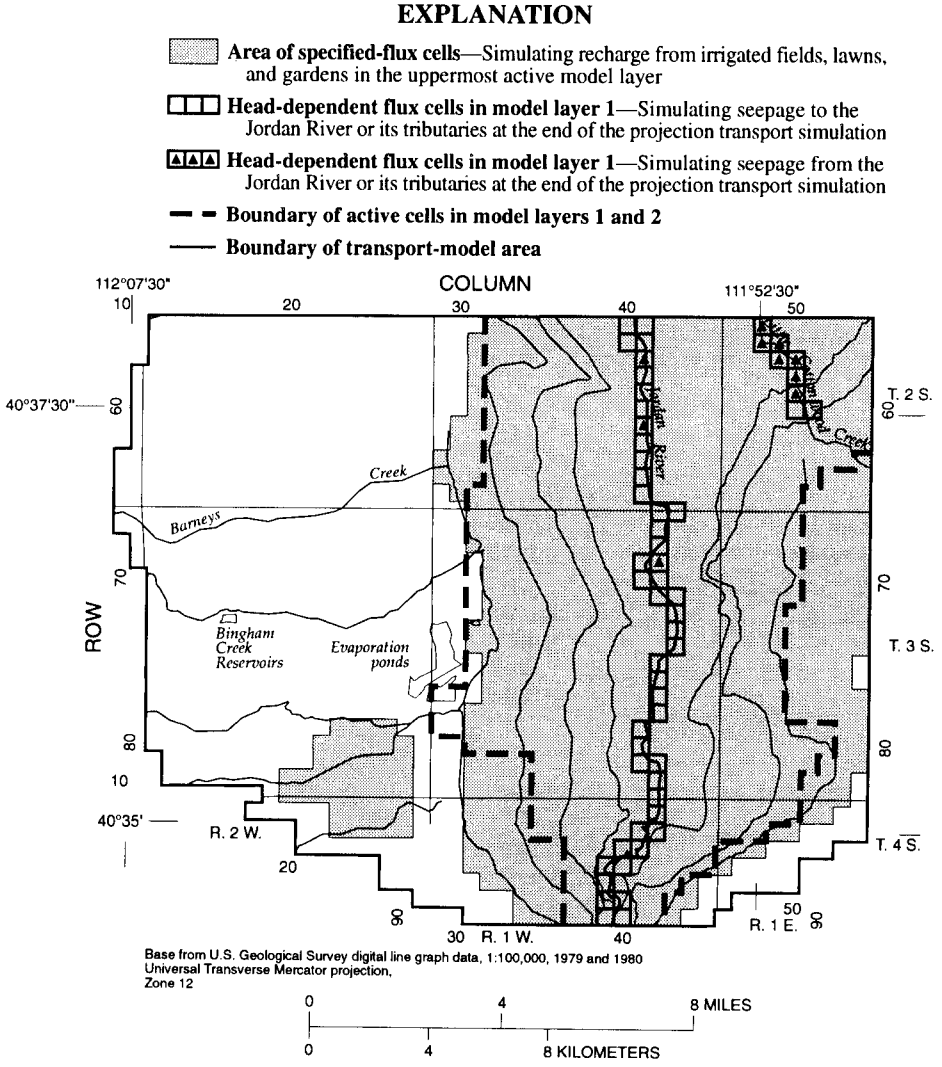

Figure 8. Location of specified-flux cells used to simulate recharge from irrigated fields and lawns and gardens, and head-dependent flux cells within the transport-model area in southwestern Salt Lake Valley, Utah.

tions during 1969-91 that was used to calibrate the Salt Lake Valley flow model. A single stress period was added to the beginning of the 1969-91 simulation by Lambert (1995a) to simulate conditions during 1965-68. Specified discharge and recharge rates incorporated in this stress period were the same as those used in the 1968 steady-state simulation by Lambert (1995a). Annual changes in ground-water withdrawals and storage in the valley during 1965-68 were relatively small (Waddell and others, 1987a) and the 1968 steady-state stress conditions defined by Lambert (1995a) were assumed to adequately represent conditions during 1965-68. A single stress period was added to the end of the 1969-91 simulation by Lambert (1995a) to simulate conditions during 1992-93. Average pumping from wells during 1992-93 was estimated on the basis of values reported by water users and was incorporated into the stress period. Recharge from seepage from irrigated fields, lawns, and gardens was set equal to the values used in the final stress period of the 1969-91 simulation by Lambert (1995a). All other specified discharge and recharge rates incorporated in the stress period were the same as those used in the 1968 steady-state simulation by Lambert (1995a). Rates of recharge and discharge either specified or computed at the end of the 1965-93 flow-model simulation are listed in table 1. Annual recharge rates for seepage from the Bingham Creek Reservoirs and the South Jordan evaporation ponds incorporated in the 1965-93 simulation are shown in figure 9.

A simulation representing projected conditions for 1994-2043 was developed to provide input to the projection transport simulation. This simulation was

Table 1. Ground-water budget for Salt Lake Valley flow-model simulations incorporated in the transport model of southwestern Salt Lake Valley, Utah

[Data in acre-feet per year]

Budget element	Specified or computed at the end of the calibration transport simulation (1965-93)	Specified or computed at the end of the projection transport simulation (1994-2043)
Recharge from		
Consolidated rock	135,000	136,000
Irrigated fields, lawns, and gardens	41,000	41,000
Precipitation	67,000	67,000
Canals	30,000	30,000
Streams and channel fill	16,000	16,000
Underflow at Jordan Narrows	2,500	2,500
Reservoirs and evaporation ponds	0	0
Jordan River and tributaries	2,100	11,000
Storage	20,000	4,700
Total (rounded)	314,000	308,000
Discharge to		
Jordan River and tributaries	111,000	71,000
Wells (total)	131,000	174,000
Public-supply	(78,000)	(120,000)
Stock and domestic	(30,000)	(30,000)
Industrial	(20,500)	(21,000)
Irrigation	(2,500)	(2,500)
Evapotranspiration	32,000	28,000
Springs	19,000	19,000
Drains	7,600	6,700
Great Salt Lake	1,300	1,200
Canals	9,200	9,200
Storage	2,300	0
Total (rounded)	313,000	309,000

based on a projection simulation developed previously by Lambert (1995b). The 1994-2043 simulation incorporated a projected increase in pumpage from public-supply wells of about 55 percent from 1993 to 2025 (table 1) as estimated by Utah State University College of Engineering; Utah Department of Natural Resources, Division of Water Resources; and Bureau of Reclamation (1993), in a water-demand/water-supply model for the valley and by the Salt Lake County Water Conservancy District (written commun., 1994). Discharge from public-supply wells was held constant in the simulation for 2026 to 2043. Projected withdrawals at a planned well to be used by Kennecott Utah Copper to test the feasibility of long-term extraction of contaminated ground water east of the Bingham Creek Reservoirs (David Cline, Kennecott Utah Copper, oral commun., 1995) also were incorporated in the simulation (table 2). Discharge from other wells used for industrial, stock, irrigation, and domestic purposes was assumed not to change during the simulation period. The location of major public-supply and industrial wells in the vicinity of sulfate plumes is shown in figure 3, and information for those wells is listed in table 2. Projected increases in pumpage at four public-supply wells in the southwestern part of the valley owned by West Jordan City (table 2 and fig. 3) were not incorpo-

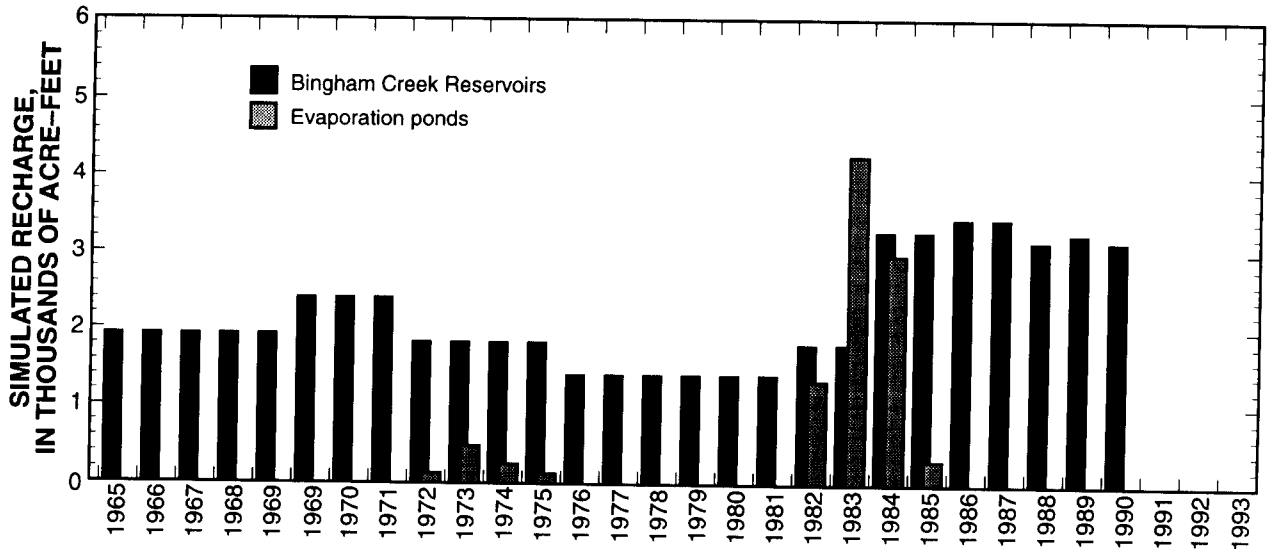

Figure 9. Simulated recharge at specified-flux cells that represent seepage from the Bingham Creek Reservoirs and from the South Jordan evaporation ponds in southwestern Salt Lake Valley, Utah, 1965-93.

rated in the projection flow-model simulation. Initial simulations incorporating projected increases in pumping for these wells during a previous study resulted in water-level declines near the wells considered unacceptable by State water managers (Lambert, 1995b, p. 10). Instead, 1992-93 average rates of withdrawal for these wells were incorporated in the projection simulation. Rates of recharge and discharge, either specified or computed, at the end of the projection flow-model simulation are listed in table 1.

Simulation of Sulfate Movement

The transport model incorporates user-specified values for transport-model parameters and output from flow-model simulations to simulate sulfate movement in southwestern Salt Lake Valley (fig. 1) as a result of advection, dispersion, and chemical reactions. In the following subsections, the approach to mathematically simulating solute transport in the transport-model area is presented.

Table 2. Selected public-supply and industrial wells simulated as withdrawing more than 400 acre-feet per year of ground water in projection transport simulations in southwestern Salt Lake Valley, Utah

Well designation	Name	Owner	Use	Discharge simulated during projection transport simulation, in acre-feet per year
1	Barney Creek	West Jordan City	Public supply	970
2	New Bingham	West Jordan City	Public supply	1,650
3	Well #6	West Jordan City	Public supply	1,990
4	Fire Station	West Jordan City	Public supply	1,960
5	Oakstreet	Midvale City	Public supply	160-540
6	Copperview	Sandy City	Public supply	770-790
7	Well #190	Kennecott Utah Copper	Industrial	2,100
8	Well #60	Kennecott Utah Copper	Industrial	1,800
9	Extraction well[1]	Kennecott Utah Copper	Industrial	400

[1] Planned extraction well to determine the feasibility of long-term extraction of contaminated ground water as part of a remediation study. Discharge from the well is simulated in the projection transport simulation beginning in 1996.

The MT3D Solute-transport Model—Governing Equations And Solution Techniques

The governing equation for solute transport, sometimes referred to as the transport equation or the advection-dispersion-reaction equation (Zheng, 1992, p. 1-1 and 2-1), describes the transport of miscible solute in ground water and can be written as

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x_i}\left(D_{i,j}\frac{\partial C}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(v_i C) + \frac{q_s}{\theta}C_S + \sum_{k=1}^{N} R_k \qquad (2a)$$

where

C is the concentration of the dissolved solute (M/L^3),

t is time (T),

x_i is the distance along the respective Cartesian coordinate axis (L),

$D_{i,j}$ is the dispersion coefficient (L^2/T),

v_i is the seepage or average linear velocity of ground water (L/T),

q_s is the volumetric flux of water per unit volume of aquifer representing sources (positive) and sinks (negative) (1/T),

C_s is the concentration of sources or sinks, (M/L^3),

θ is the porosity of the porous medium (dimensionless), and

$\sum_{k=1}^{N} R_k$ is the chemical reaction term and R_k is the rate of production of the solute in reaction k of N different reactions (M/L^3T).

The first term on the right side of equation 2a is the dispersion term, which allows for the representation of the spreading of solute over a greater region than would be predicted solely by advection. The second term on the right side of equation 2a is the advection term and describes the transport of a solute at the same velocity as that of the ground water. The third term on the right side of equation 2a is the source-sink term and represents dissolved solute mass entering or leaving the modeled ground-water system. The fourth term in equation 2a describes chemical reactions involving the solute. In the transport model used in this analysis, it is assumed that only equilibrium-controlled sorption reactions are involved in the chemical reactions affecting solute transport and that the chemical reaction term in equation 2a can be expressed as

$$\sum_{k=1}^{N} R_k = -\frac{\rho_b}{\theta}\frac{\partial \overline{C}}{\partial t} = -\frac{\rho_b}{\theta}\frac{\partial C}{\partial t}\frac{\partial \overline{C}}{\partial C} \qquad (2b)$$

where

ρ_b is the bulk density of the porous medium (M/L^3) and

$\overline{C}$ is the concentration of solid-phase contaminant sorbed on the porous medium (M/M).

By substituting equation 2b into 2a and rearranging, equation 2a can be written as

$$R\frac{\partial C}{\partial t} = \frac{\partial}{\partial xi}Di,j\frac{\partial C}{\partial xj} - \frac{\partial}{\partial xi}v_i C + \frac{q_s}{\theta}C \qquad (2c)$$

where R is the retardation factor and is defined as

$$R = 1 + \frac{\rho_b}{\theta}\frac{\partial \overline{C}}{\partial C} \qquad (2d)$$

The representation of chemical reactions in the transport model and the computation of the retardation factor (R) is discussed in more detail later in the report.

Equation 2c is the governing equation underlying the MT3D solute-transport model (Zheng, 1992, p. 2-2) used in this analysis. The transport model uses two approaches to obtain a solution to the equation. The transport model employs the forward tracking method of characteristics (MOC) to solve the advection term of the transport equation and then computes the additional change in concentration resulting from dispersion, mixing of fluid sources, and chemical reactions by solving an explicit finite-difference equation. This approach has been referred to as a mixed Eulerian-Lagrangian method (Neuman, 1984, and Zheng, 1992). The MOC technique solves the advection term of the transport equation by distributing a set of moving particles in the simulated flow field that can be tracked with reference to a stationary finite-difference grid. A concentration and a position in the grid are associated with each particle. Particles are tracked forward in proportion to the flow velocity using small time increments. At the end of each increment the concentration within a given cell resulting from advection is evaluated from the concentration of all the moving particles in the cell. The additional change in concentration resulting from dispersion, fluid sources and sinks, and chemical reactions is computed by solving an explicit finite-difference equation. The MT3D model includes two modified versions of MOC designed to reduce the total number of moving particles needed in a given simulation. These methods were not used, however, during this analysis.

The governing transport equation is a mathematical statement of mass conservation and requires the

change in mass stored in an elemental volume of the system to be equal to the net mass flux, plus the difference between the mass contributed by sources and withdrawn by sinks in that volume (S.S. Papadopulos and Associates, Inc., 1994, p. 1). The particle-based MOC technique implemented in the MT3D transport model (Zheng, 1992) however, does not require strict adherence to mass conservation to produce an accurate solution. Still, mass balance discrepancies computed by the model can serve as an indicator of the validity of the numerical solution. It is generally assumed that mass balance discrepancies that oscillate about zero, diminish through time, and do not consistently exceed 10 to15 percent indicate an accurate solution (S.S. Papadopulos and Associates, Inc., 1994, p. 5).

The MOC technique was used in a U.S. Geological Survey two-dimensional transport model (Konikow and Bredehoeft, 1978) and has been widely used in field studies (Zheng, 1992, p. 3-3). One of the benefits of using the MOC technique is that it generally does not introduce significant numerical dispersion during problem solution. Numerical dispersion is an artificial dispersion resulting from the numerical calculation process that is evident in many standard numerical procedures. For advection-dominated problems where large concentration gradients occur, many numerical approaches to solving the advection term of the transport equation, including the finite-difference method, are susceptible to excessive numerical dispersion and are limited by small grid spacing. The MOC approach utilized in MT3D is not limited by grid spacing in advection-dominated problems with sharp concentration fronts.

Zheng (1992 and 1993) described the extension of the MOC method to the simulation of solute transport in three dimensions in the MT3D model. Zheng (1992 and 1993) also evaluated the accuracy of the model by applying it to two- and three-dimensional, uniform-flow test problems and comparing the model results with analytical solutions to those problems. The test problems used in Zheng's evaluation included (1) two-dimensional transport of solute injected continuously from a point source (Zheng, 1992, p. 7-4 - 7-6), (2) two-dimensional transport of solute injected from a fully penetrating well into a radial flow field (Zheng, 1992, p. 7-8 - 7-10), and (3) three-dimensional solute transport from a continuous point source in a spherical flow field (Zheng, 1993, p. 462). Zheng (1992 and 1993) reported good agreement between model-computed and analytical solutions for all test problems.

The accuracy of the MT3D model also was evaluated during this study by comparing model results with analytical solutions to two test problems presented by Wexler (1992, p. 31 and 53): (1) two-dimensional solute transport in an aquifer of infinite areal extent with a continuous point source, and (2) three-dimensional solute transport in an aquifer of finite width and height with a solute source of finite width and height. For both test problems the evaluation of model-computed solutions indicated a good match between model-computed and analytically derived solute concentration.

The MT3D transport model requires input specified by the user and data computed in an existing ground-water flow model to solve the transport equation. The parameter requirements of the model and the techniques used in the model to represent the individual components of the transport equation are discussed in the following paragraphs.

Advection and Dispersion

The transport equation (eq 2c) is linked to the flow equation, solved by the ground-water flow model, through the advection term $(\frac{\partial}{\partial x_i}(v_i C))$ and the relation defined in equation 1 written here for multi-dimensional flow as

$$v_i = -\frac{K_{ii}}{\theta}\frac{\partial h}{\partial x_i} \qquad (3)$$

where

v_i is the average linear velocity in the direction of x_i (L/T), and

K_{ii} is the principal component of the hydraulic-conductivity tensor (L/T).

The hydraulic head (h) in equation 3 is obtained from the solution of the three-dimensional ground-water flow equation in the ground-water flow model. Porosity (θ) in the advection term must be specified for all active cells in the transport-model area.

The dispersion coefficient (D_{ij}) in the dispersion term (eq 2c) is also related to the velocity of ground-water flow as defined in the equation set below. This equation set defines the hydrodynamic dispersion tensor for a porous media (Bear, 1979, p. 234-235, and Zheng, 1992, p. 2-6).

$$D_{xx} = \alpha_L \frac{v_x^2}{|v|} + \alpha_{TH} \frac{v_y^2}{|v|} + \alpha_{TV} \frac{v_z^2}{|v|} + D' \quad (3a)$$

$$D_{yy} = \alpha_L \frac{v_y^2}{|v|} + \alpha_{TH} \frac{v_x^2}{|v|} + \alpha_{TV} \frac{v_z^2}{|v|} + D' \quad (3b)$$

$$D_{zz} = \alpha_L \frac{v_z^2}{|v|} + \alpha_{TV} \frac{v_x^2}{|v|} + \alpha_{TV} \frac{v_y^2}{|v|} + D' \quad (3c)$$

$$D_{xy} = D_{yx} = (\alpha_L - \alpha_{TH}) \frac{v_x v_y}{|v|} \quad (3d)$$

$$D_{xz} = D_{zx} = (\alpha_L - \alpha_{TV}) \frac{v_x v_z}{|v|} \quad (3e)$$

$$D_{yz} = D_{zy} = (\alpha_L - \alpha_{TV}) \frac{v_y v_z}{|v|} \quad (3f)$$

where

α_L is the longitudinal dispersivity (L),

α_{TH} is the horizontal transverse dispersivity (L),

α_{TV} is the vertical transverse dispersivity (L),

D' is the effective molecular diffusion coefficient (L^2/T),

v_x, v_y, v_z are components of the velocity vector along the x, y, and z axes (L/T), and

$|v| = \left(v_x^2 + v_y^2 + v_z^2 \right)^{1/2}$ is the magnitude of the velocity vector (L/T).

The longitudinal (α_L) and transverse (α_{TH}, α_{TV}) dispersivities are specified by the user and the dispersion coefficients are computed by the transport model.

Chemical Reactions

Chemical reactions occurring within the sulfate plume east of the Bingham Creek Reservoirs discussed in the "History and pattern of contamination" section are complex and not fully understood. It was not within the scope of this analysis and the capabilities of the transport model to explicitly incorporate these reactions in model simulations. It was assumed, however, that the observed retarding effect of chemical reactions on sulfate movement in the low-pH zone of the plume could be represented in the transport model using a simplified approach.

The method used to simulate the retardation of sulfate movement in the plume east of the Bingham Creek Reservoirs included simulating a distribution and transfer of sulfate between the dissolved phase (dissolved sulfate in ground water) and the solid phase (sulfate in the aquifer material) by using an equilibrium-controlled sorption reaction. Although this simplified approach is not explicitly representing precipitation, which is the principal mechanism of sulfate attenuation in the plume, the approach does allow for the simulation of three principal aspects of the conceptual geochemical model of the plume: (1) the removal of dissolved sulfate from ground water in a localized area of the plume, (2) the transfer of sulfate from the dissolved phase to the solid phase, and (3) the availability of solid-phase sulfate for resolutioning.

The sorption and desorption of sulfate is simulated in the transport model as a functional relation between the dissolved- and solid-phase sulfate concentrations. This relation is defined by an isotherm known as the Freundlich sorption isotherm (fig. 10) that can define both a linear or a nonlinear relation between the dissolved- and solid-phase sulfate and can be expressed as

$$\overline{C} = K_f C^a \quad (4)$$

where

K_f is the Freundlich isotherm constant $(L^3/M)^a$ and

a is the Freundlich isotherm exponent (dimensionless).

The Freundlich isotherm constant (K_f) and exponent (a) are specified in the transport model. The transport model uses the isotherm to simulate a distribution of sulfate between the dissolved and solid phase and to calculate the retardation factor (R) in the transport equation (eq 2c) as follows (Zheng, 1992):

$$R = 1 + \frac{\rho_b}{\theta} \frac{\partial \overline{C}}{\partial C} = 1 + \frac{\rho_b}{\theta} a K_f C^{a-1} \quad (5)$$

Boundary Conditions and Sources and Sinks

The transport model requires that specific types of mathematical boundaries be assigned in the model to simulate sulfate flux at model surfaces or at internal sources and sinks. Two types of boundary conditions were used in this analysis: (1) a specified-concentration boundary, and (2) a specified-solute-flux boundary.

A specified-concentration boundary was defined at vertical columns of model cells at the border of the transport-model area (fig. 11). At this boundary the concentration of sulfate in ground water was specified

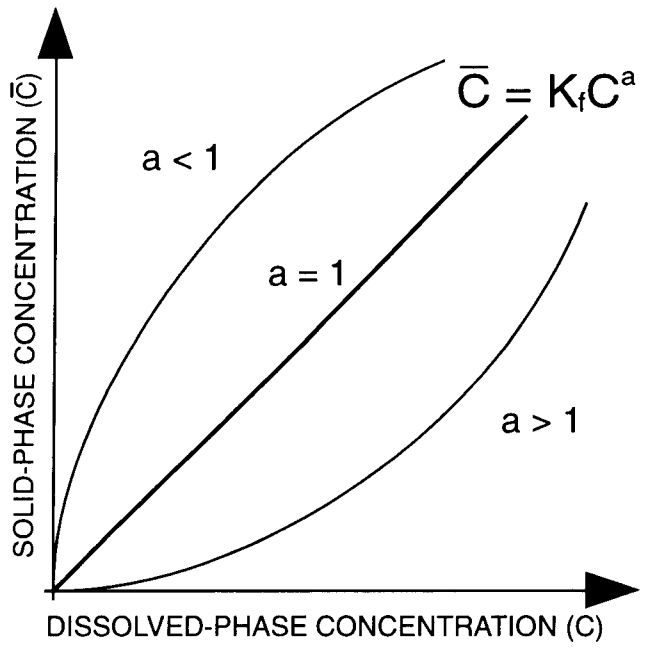

Figure 10. Example of Freundlich sorption isotherm.

and remained unchanged throughout the simulations. The boundary can act as a source providing sulfate mass to the modeled system, or as a sink taking sulfate mass out of the modeled system.

Sulfate flux across all other model boundaries and at internal sources and sinks was specified in transport-model simulations by assigning sulfate concentrations to recharge and discharge simulated in the transport-model area. These fluxes are represented by the source/sink term in the transport equation (eq 2c).

ESTIMATION OF TRANSPORT-MODEL PARAMETERS AND AVAILABLE DATA

Data from previous work in Salt Lake Valley were used to evaluate transport-model parameters. These parameters included effective porosity of the aquifer (θ), longitudinal (α_L) and transverse (α_{TH}, α_{TV}) dispersivities, the Freundlich isotherm constant (K_f) and exponent (a), and dissolved-sulfate concentration at model boundaries and internal sources and sinks (C_S). Estimates of the sulfate concentration of simulated recharge to and discharge from the transport-model area were made independent of the model and were not adjusted during calibration. All other transport-model parameters were considered to be calibration variables that could be adjusted within prescribed ranges during model calibration. In the following subsections, data used to determine sulfate concentration of simulated recharge and discharge are discussed. Ranges of probable values for calibration variables are defined where possible, and preliminary estimates of those parameters used in the transport model are presented. Final estimates of transport-model parameters resulting from model calibration are presented later in the "Model calibration and sensitivity" section of the report.

Porosity

Effective porosity, as required in the transport model, refers to the percentage of interconnected pore space available for fluid movement (Lohman, 1979, p. 10). Estimates of effective porosity are not incorporated in the Salt Lake Valley flow model (Lambert, 1995a) linked to the transport model and were estimated during this analysis on the basis of literature values of total porosity for unconsolidated sediments compiled by Freethey and others (1994). Although total porosity, which does not require pore connection, can be more than one order of magnitude larger than effective porosity in consolidated and fractured rock (Domenico and Schwartz, 1990, p. 25 and table 21), the difference is typically much less in unconsolidated sediments. It was assumed that an estimated range of total porosity, termed porosity for the purposes of this discussion, could be used as a probable range for effective porosity for transport-model development and calibration.

On the basis of the compiled literature values, Freethey and others (1994, p. 12 and 15, and fig. 7) estimated that porosity of the sediments in the principal aquifer in southeastern Salt Lake Valley may range from 15 to 35 percent. Literature values included in the compilation by Freethey and others (1994, table 2) indicate that typical porosity values for fine-grained sediments such as the mixtures of silt and clay that compose the shallow confining layer and the shallow unconfined aquifer range from 30 to 60 percent (Davis, 1969, and Bedinger and others, 1986). The scope of this investigation did not include the determination of spatial variation in sediment porosity throughout the basin-fill ground-water system. Rather, the ranges for these porosity values were assumed to be representative of conditions in southwestern Salt Lake Valley and were used to assign initial porosity values to model layers representing the principal aquifer, and the finer-grained sediments of the shallow confining layer and the shallow unconfined aquifer. A porosity of 30 percent was

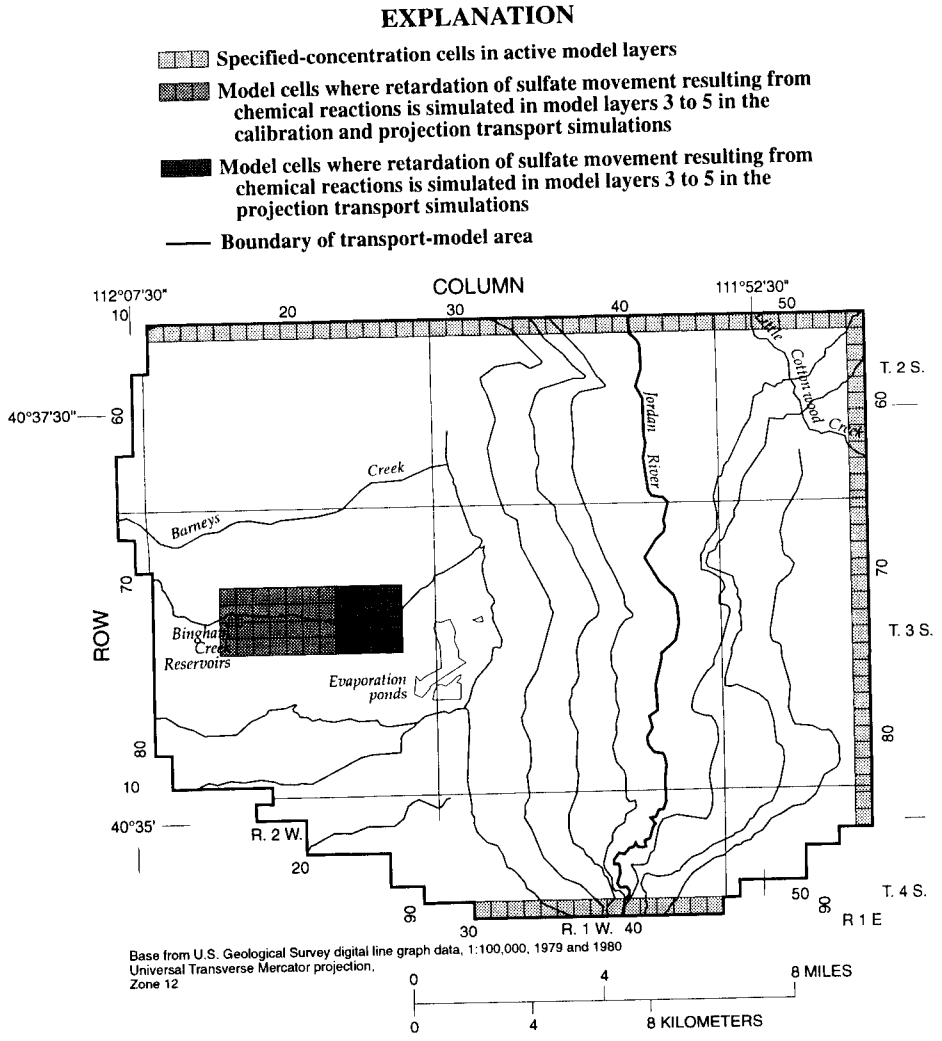

Figure 11. Location of specified-concentration cells and model cells where retardation of sulfate movement resulting from chemical reactions is simulated in the transport model of southwestern Salt Lake Valley, Utah.

used for model layers 3 to 7, representing the principal aquifer. A porosity of 40 percent was assumed for model layers 1 and 2, representing the shallow unconfined aquifer and the shallow confining layer, respectively. During calibration, it was assumed that porosity values could be adjusted within the ranges defined above.

Dispersivity

The transport model computes the dispersion coefficient ($D_{i,j}$) in the dispersion term of the transport equation (eq 2c) as a function of specified longitudinal (α_L) and transverse (α_{TH}, α_{TV}) dispersivities. Dispersivities have units of length and are characteristic properties of the aquifer material that quantify, in theory, mechanical dispersion in the material (Domenico and Schwartz, 1990, p. 372). A range of values of α_L, α_{TH}, and α_{TV} for use during model calibration was defined on the basis of literature values compiled by Gelhar and others (1992) and on estimates of dispersivities resulting from previous studies in the southwestern part of the valley (Holdsworth, 1985, and Dames and Moore, 1989).

A complicating factor in estimating dispersivity in a field problem is the indication from available data that dispersivity is scale dependent. Gelhar and others (1992, p. 1971) reported that a review of field observations of dispersivity indicated a clear trend of systematic increase of longitudinal dispersivity with scale. Gelhar and others (1992, p. 1971) reported that the pattern of dependency is less clear when only field data classified as being highly reliable is considered but that

the "high-quality" data still indicate that some overall trend in increasing dispersivity with scale is plausible.

In an effort to obtain a reasonable range of dispersivity values to be used during model calibration, only estimates of dispersivity reported for field problems of similar scale to that of the discretized transport-model area were considered. Estimates of longitudinal dispersivity compiled by Gelhar and others (1992, table 1) from field studies at scales of 500 to 2,000 ft (model cells in the transport model are 1,848 ft on a side) range from about 15 to about 600 ft. Gelhar and others (1992, p. 1972) reported that their analysis of the available data provides evidence favoring the use of dispersivity values in the lower half of a range at any given scale. Estimates of longitudinal dispersivity from previous modeling analyses using a similar scale of discretization to that of the transport model documented here ranged from 30 to 300 ft (Holdsworth, 1985, table 2, and Dames and Moore, 1989, table 2). On the basis of these reported values, a probable range of 15 to 300 ft for longitudinal dispersivity was defined for use during model calibration. Longitudinal dispersivity was initially specified to be 50 ft throughout the transport-model area.

Horizontal-transverse (α_{TH}) and vertical-transverse (α_{TV}) dispersivities are incorporated in the transport model by specifying the ratios α_{TH}/α_L and α_{TV}/α_L. Estimated values of α_{TH}/α_L compiled by Gelhar and others (1992, fig. 6) for appropriate scale problems range from 0.05 to about 0.33. Estimated values for α_{TV}/α_L ranged from 0.001 to 0.1 (Gelhar and others, 1992, fig. 6). Initially, α_{TH}/α_L and α_{TV}/α_L were specified in the transport model to be 0.1 and 0.01, respectively. During model calibration, it was assumed that α_{TH}/α_L and α_{TV}/α_L could be adjusted within the ranges defined above.

Sulfate Concentration at Model Boundaries and Sources and Sinks

The transport model requires data defining the sulfate concentration of ground water entering or leaving the system at model boundaries and internal sources or sinks. Constant-concentration cells at the north, east, and south borders of the transport model (fig. 11) were assigned a sulfate concentration of 100 mg/*l*. This concentration was assumed to represent an average background sulfate concentration for the area (Dames and Moore, 1989, table 3, and Shepherd Miller, Inc., 1995, table 3). Recharge from the infiltration of precipitation, which is simulated throughout the model area, was also assigned a sulfate concentration of 100 mg/*l*. The sulfate concentration of seepage from precipitation was not determined from field measurements but was selected to simulate recharge with background sulfate concentration.

Sulfate concentrations specified for recharge simulated in the transport model as infiltration of unconsumed irrigation water, seepage from canals, inflow from consolidated rock at the margin of the valley, seepage from the Jordan River and its principal tributaries, and seepage from the Bingham Creek Reservoirs and the South Jordan evaporation ponds were estimated on the basis of values reported in previous studies and are summarized in table 3. Irrigated fields generally receive irrigation water from canals that divert water from the Jordan River near Jordan Narrows (fig. 1). The average sulfate concentration of water in the Jordan River at Jordan Narrows has been reported to be about 350 mg/*l* (Hely and others, 1971, table 13). Thus, simulated recharge from seepage of unconsumed irrigation water (fig. 8) was assigned a sulfate concentration of 350 mg/*l*. Recharge from seepage from major canals carrying water through the transport-model area, except the Provo Reservoir Canal, also was simulated as having a sulfate concentration of 350 mg/*l*. The Provo Reservoir Canal has transported water to southwestern Salt Lake Valley from the Provo River from the early 1900s to 1992. The average sulfate concentration of water delivered to the Provo Reservoir Canal has been reported to be about 50 mg/*l* (Hely and others, 1971, p. 74). Since 1992, the Provo Reservoir Canal has diverted irrigation water mainly from the Jordan River near Jordan Narrows. The sulfate concentration of simulated recharge from seepage from the Provo Reservoir canal was specified in the transport model to be 50 mg/*l* prior to 1992 and 350 mg/*l* after 1992.

Inflow from consolidated rock is simulated in the transport-model area along the west margin of the valley (fig. 7). A sulfate concentration of 350 mg/*l* was assigned to this inflow on the basis of estimates from a previous modeling analysis in the area (Dames and Moore, 1988, table 3). The incorporated value falls within the range of background values for ground water in volcanic bedrock in southwestern Salt Lake Valley, recently reported by Kennecott Utah Copper (Shepherd Miller, Inc., 1995, table 3).

Seepage from the Jordan River and the lower reaches of its principal tributaries is computed in flow-

Table 3. Specified sulfate concentrations of simulated recharge incorporated in the transport model of southwestern Salt Lake Valley, Utah

[Data in milligrams per liter]

Recharge element	Specified sulfate concentrations in transport simulations		
	Preliminary simulation (1935-64)	Calibration simulation (1965-93)	Projection simulation (1994-2043)
Consolidated rock	350	350	350
Irrigated fields, lawns, and gardens	350	350	350
Precipitation	100	100	100
Canals[1]	350	350	350
South Jordan evaporation ponds[2]	3,500	3,500	0
Bingham Creek Reservoirs[3]	0	40,000	0
Cemetery Pond[4]	0	350	0
Jordan River	0	0	410-430
Little Cottonwood Creek	0	0	150

[1]Simulated recharge as seepage from the Provo Reservoir Canal was assigned a sulfate concentration of 50 milligrams per liter in the preliminary transport simulation and in the calibration transport simulation during 1965-91.

[2]Recharge as seepage from the South Jordan evaporation ponds was simulated in the calibration transport simulation during 1965, 1972-75, and 1982-86.

[3]Recharge as seepage from the Bingham Creek Reservoirs was simulated in the calibration transport simulation during 1965-90.

[4]Recharge as seepage from the Cemetery Pond was simulated in the calibration transport simulation during 1984-90.

model simulations and incorporated in the transport model in areas where the simulated water-table altitude is below the model-computed water level in river cells. This condition does not occur within the transport-model area in the preliminary and calibration transport simulations used in this analysis. Seepage from the Jordan River and the lower reaches of its tributaries, however, is simulated within the transport-model area in the projection transport simulation (fig. 8). The specified sulfate concentration of simulated seepage from the Jordan River was defined on the basis of reported concentrations of Jordan River water from Jordan Narrows north to 2100 South Street (fig. 1) (Hely and others, 1971, table 13) and ranged from 410 mg/*l* to 430 mg/*l*. Water in Little Cottonwood Creek has been reported to have a sulfate concentration of about 150 mg/*l*, and the sulfate concentration of simulated seepage from the creek (fig. 8) was specified in the transport model at that value.

Sulfate concentrations of seepage from the Bingham Creek Reservoirs, the Cemetery Pond, and the South Jordan evaporation ponds were specified on the basis of estimates discussed in the "History and pattern of contamination" section of this report and are summarized in table 3. As noted in that section, sulfate concentration of water diverted to the South Jordan evaporation ponds during 1935-65 was estimated to be as high as 40,000 mg/*l*. Sulfate concentrations measured in ground water directly downgradient of the evaporation ponds, however, have generally not exceeded 2,000 mg/*l*. Previous field investigations and modeling efforts in this area indicate that sulfate concentration of water seeping from the ponds may have been reduced by the precipitation of gypsum in sediments in the evaporation ponds or directly below the ponds (Dames and Moore, 1989, and Adrian Brown Consultants, oral commun., 1994). It was assumed in transport-model simulations, therefore, that the sulfate concentration of pond seepage entering the groundwater system was limited to 3,500 mg/*l*.

Sorption Isotherm Coefficients and Retardation

Retardation of sulfate movement resulting from chemical reactions was simulated in the calibration and projection transport simulations in the low-pH zone of the sulfate plume east of the Bingham Creek Reservoirs (fig. 11). Values for the Freundlich isotherm constant (K_f) and exponent (a) defining the relation between dis-

solved-phase and solid-phase sulfate were specified at model cells in that area.

A range of values for the Freundlich isotherm constant (K_f) to be used during model calibration was estimated from solid-phase chemistry data collected inside and outside the of low-pH zone of the sulfate plume east of the Bingham Creek Reservoirs. Adrian Brown Consultants, Inc. and Adrian Smith Consulting, Inc. (1990, table 5.6) reported a typical solid-phase sulfate concentration inside the low-pH zone of the plume to be 1,145 mg/kg and a typical concentration outside of the zone to be 100 mg/kg or less. Dissolved-sulfate concentration may exceed 30,000 mg/*l* in the low-pH zone and is typically less than 3,500 mg/*l* outside of that area. Assuming an initial value for the Freundlich isotherm exponent (a) equal to 2, equation 4 was rearranged and solved for K_f as follows:

$$K_f = \frac{\overline{C}}{C^2}$$

By using the values for solid-phase sulfate concentration ($\overline{C}$) and dissolved-phase concentration (C) reported above, an order-of-magnitude range of K_f values of 1×10^{-12} l^2/mg^2 to 1×10^{-11} l^2/mg^2 was estimated for use during model calibration. Initially, K_f was set equal to 5×10^{-12} l^2/mg^2.

By defining the Freundlich isotherm exponent (a) to be a number greater than 1, the distribution of sulfate between the dissolved phase and the solid phase is represented as a nonlinear relation (fig. 10). Setting a equal to 2 initially in the calibration simulation resulted in very little retardation of sulfate movement at specified cells (fig. 11) when computed dissolved-sulfate concentrations at these cells were less than 3,500 mg/*l*. This representation is consistent with the conceptual geochemical model of the plume east of the Bingham Creek Reservoirs, which assumes that at concentrations less than 3,500 mg/*l*, sulfate movement is not affected by chemical reactions.

An estimate of bulk density (ρ_b) also is required by the transport model to compute the retardation factor (R) (eq 5). Bulk density (ρ_b) was estimated on the basis of its relation to the particle density of the aquifer (ρ_p) defined in the following equation (Freeze and Cherry, 1979, p. 337):

$$\rho_b = \rho_p (1 - \theta) \qquad (6)$$

A bulk density (ρ_b) value of 1.8 g/cm^3 was estimated assuming that particle density (ρ_p) was equal to 2.65 g/cm^3 (Freeze and Cherry, 1979, p. 337) and porosity (θ) was equal to 0.3. The estimate used for ρ_p was not based on measurements made in the study area, but rather is a typical value reported in the literature for mineral soils. The value used for porosity also is a general estimate selected from the estimated range of porosity for the principal aquifer reported in a previous section of 0.15 to 0.35. Thus, the calculated value for ρ_b was assumed to be a rough estimate for use in all transport simulations incorporating retardation and was not adjusted during the analysis.

Initial Conditions

Dissolved-sulfate concentrations defining initial conditions for each transport-model simulation are required at all active cells in the transport model. An initial background sulfate concentration of 100 mg/*l* was used throughout the model area in the preliminary transport simulation. Results of the preliminary simulation, which incorporated sulfate influx from natural sources, irrigation water, and seepage from the South Jordan evaporation ponds during 1935-64, were used as initial conditions for the calibration transport simulation.

Model-computed sulfate concentrations were not used to define initial conditions at all model cells for the projection transport simulation from 1994 to 2043. The calibration transport simulation was developed for model calibration purposes and was not assumed to produce accurate estimates of sulfate concentration throughout the model area. Initial sulfate concentrations representing the distribution of dissolved sulfate at the end of 1993 were estimated, where possible, on the basis of field measurements made that year (fig. 3). Where sulfate concentration could not be defined from measured values, concentrations computed at the end of the calibration transport simulation were used.

MODEL CALIBRATION AND SENSITIVITY

The transport model was applied during model calibration to compute changes in sulfate concentration in ground water in southwestern Salt Lake Valley during 1965-93. Hydraulic heads and fluxes computed in the ground-water flow model of Salt Lake Valley for the calibration period were incorporated in the simulation.

Representation of contaminant sources during calibration was limited to seepage from the Bingham Creek Reservoirs and the South Jordan evaporation ponds. Generally, estimates of seepage from these

sources (fig. 9) were based on theoretical calculations and their accuracy is unknown. Also, because of chemical reactions that may have occurred in sediments directly beneath the evaporation ponds, estimates of the sulfate concentration of seepage from the ponds that actually recharged the ground-water system are uncertain. Because of these limitations, calibration of the transport model focused mainly on sulfate movement in the principal area of contamination downgradient of the Bingham Creek Reservoirs and attempted only to match the general pattern of sulfate movement in that area. Although less emphasis was placed on sulfate movement east of the evaporation ponds during calibration, simulated patterns of movement in the vicinity of the ponds were compared periodically to observed patterns.

During calibration, transport-model parameters defined as calibration variables were adjusted within probable ranges to improve the match between model-computed and observed sulfate concentration in the principal aquifer for 1977, 1984, and 1993. Where possible, computed sulfate-concentration patterns in individual model layers were compared to observed patterns in zones of the aquifer represented by those model layers (figs. 12-14). Zones of observed sulfate concentration in 1993 shown in figure 13a were defined on the basis of measured sulfate concentrations at observation wells and on information from sulfate concentration maps for previous years (Kennecott Utah Copper, written commun., 1994).

Comparisons between computed data at the end of calibration and actual data indicate that the transport model reproduces the general dimension of the sulfate plume east of the Bingham Creek Reservoirs. The model also simulates, in general, the observed pattern of distribution of dissolved sulfate within the plume, which is believed to be a result, in part, of the effects of water/rock chemical reactions. A discrepancy, however, is indicated at each observation period between the model-computed and the observed orientation of the axis of the plume (figs. 12-14). Results of the simulation indicate sulfate movement in an easterly direction from the reservoirs, and the observed distributions of dissolved sulfate in 1977, 1984, and 1993 indicate sulfate movement in a southeasterly direction immediately downgradient from the reservoirs and movement in an easterly direction as the plume migrates toward the evaporation ponds. Efforts to more accurately simulate the observed orientation of the plume by adjusting transport-model parameters were unsuccessful. Possible causes for the consistent discrepancy between the simulated and observed orientation of the plume include (1) inaccuracies in simulated recharge west of the reservoirs and (2) inaccuracies in the representation of aquifer properties in the vicinity of the reservoirs. It is also possible that density variations between highly contaminated ground water and surrounding fresher water not simulated in the transport model has affected sulfate movement east of the reservoirs. These limitations are discussed in more detail later in the report.

Although the simulated orientation of the axis of the sulfate plume east of the Bingham Creek Reservoirs was not sensitive to changes in transport-model parameters, the simulated dimensions of the plume and the pattern of distribution of dissolved sulfate within the plume were substantially affected by adjusting values for the Freundlich isotherm constant (K_f) and exponent (a), and porosity (θ). Model-computed sulfate concentration was affected to a lesser degree by changes in longitudinal and transverse dispersivities (α_L, α_{TH}, and α_{TV}).

Simulated retardation of sulfate movement in the low-pH zone of the sulfate plume east of the reservoirs was varied during calibration by adjusting the Freundlich isotherm constant (K_f) and exponent (a). The final value for K_f and a resulting from calibration was 4×10^{-12} l^2/mg^2 and 2, respectively. Retardation factors (R) computed at cells shown in figure 11 ranged from 1 to 3.6 during the final calibration simulation.

Estimated porosity (θ) of the principal aquifer was reduced from 0.3 to 0.2 during calibration to improve the match between the simulated and observed length of the sulfate plume east of the reservoirs. The final estimate for longitudinal dispersivity (α_L) was 40 ft. Ratios defining transverse dispersivities (α_{TH} and α_{TV}) were not changed from initial estimates. Final values for all other calibration variables did not change from initial values.

The simulated extent of the sulfate plume east of the South Jordan evaporation ponds as defined by ground water with sulfate concentration greater than 500 mg/l (figs. 13 and 14) is substantially smaller than the observed extent of that plume. Possible causes for the discrepancies between model-computed and observed sulfate concentration in this area include (1) inaccuracies in estimated rates and sulfate concentrations of seepage from the evaporation ponds, (2) seepage of high-sulfate water from sources not represented in the transport model, and (3) errors in the model-computed flow field east of the ponds.

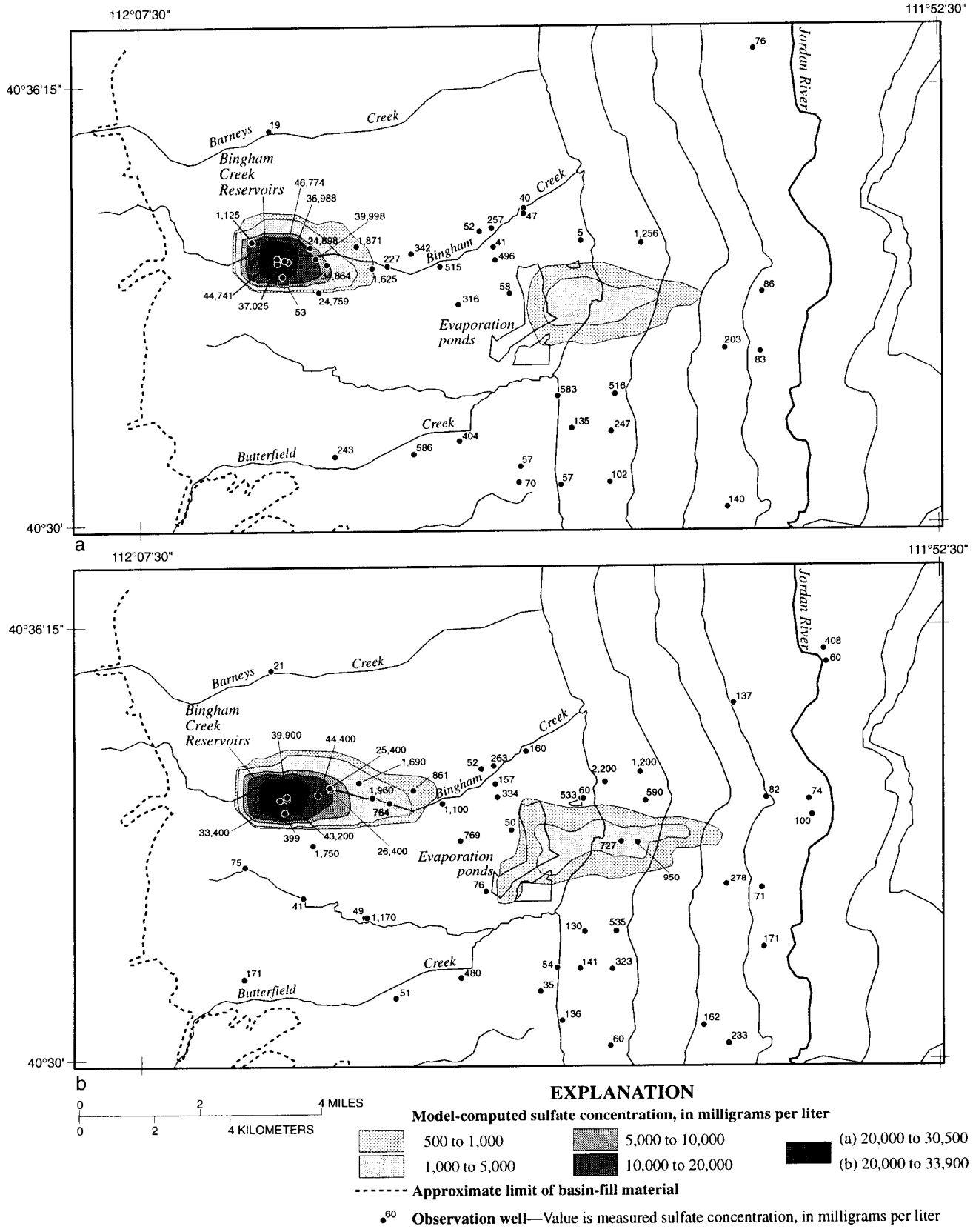

Figure 12. Model-computed and measured sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, represented in model layer 3 (a) for 1977 and (b) for 1984.

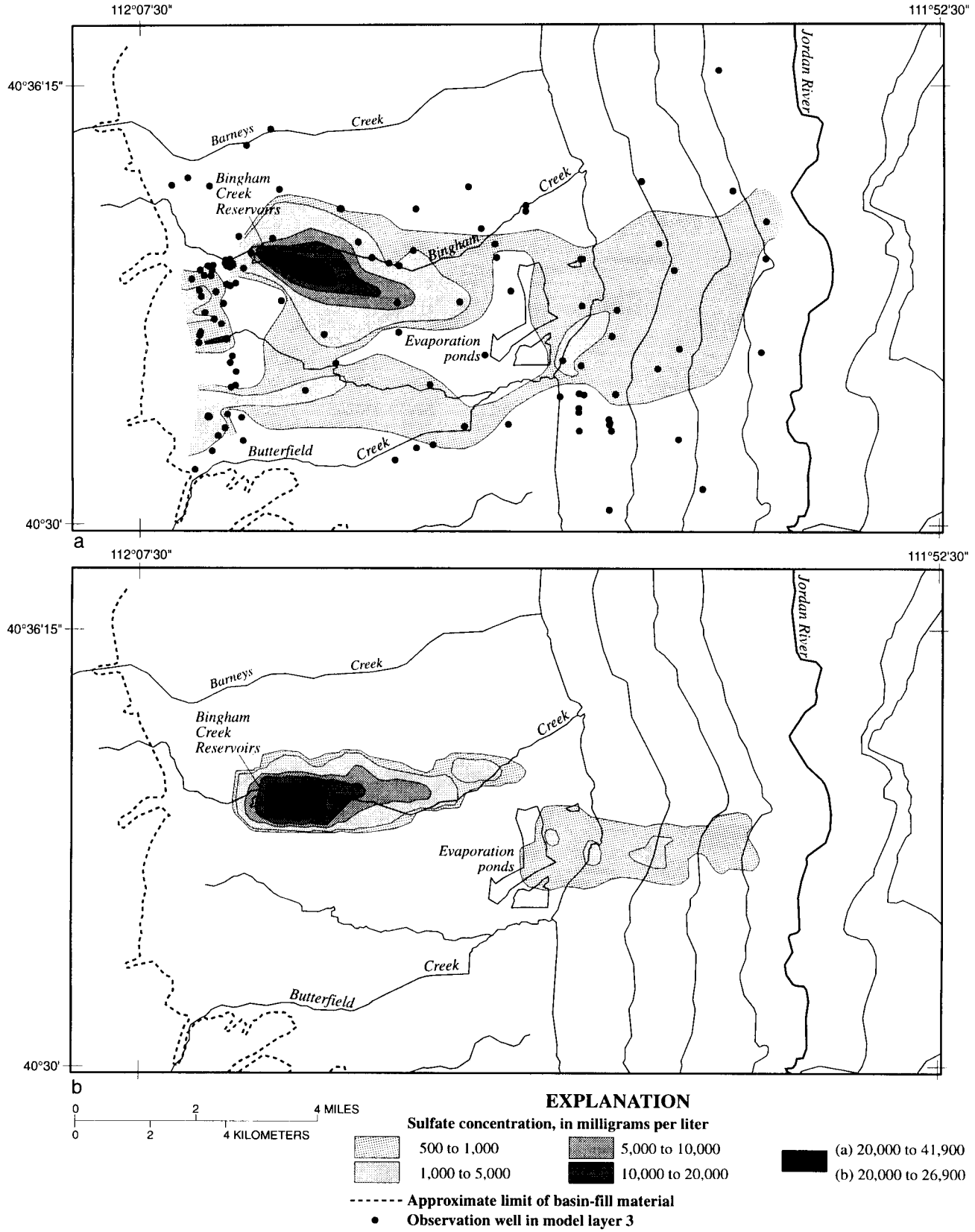

Figure 13. (a) Observed and (b) model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, represented in model layer 3 for 1993.

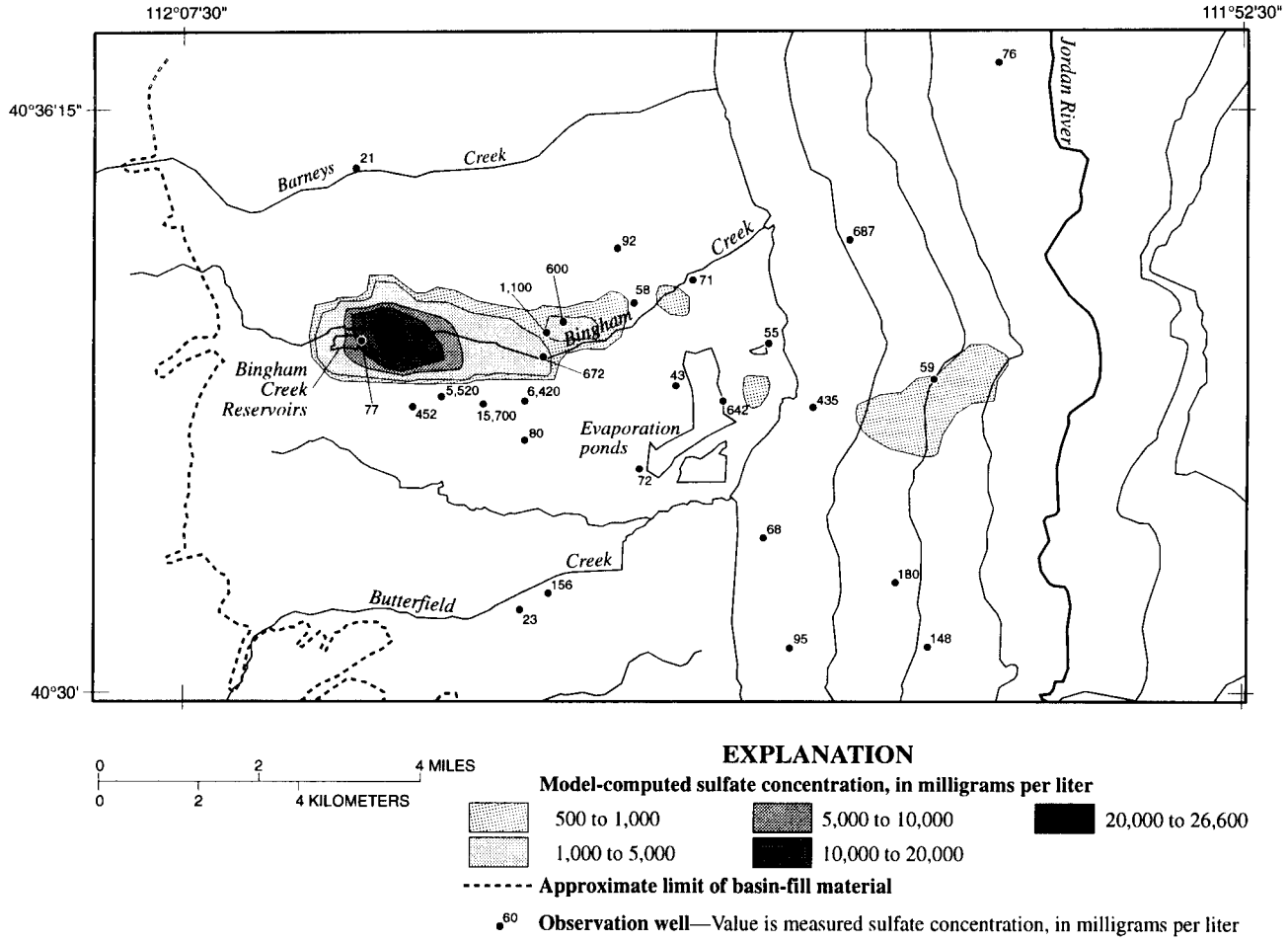

Figure 14. Model-computed and measured sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, represented in model layer 4 for 1993.

Mass-balance calculations were evaluated during the calibration transport simulation to check the numerical accuracy of the model simulations. Errors in mass balance averaged about 8 percent for the simulation and did not exceed 11.5 percent during problem solution.

MODEL LIMITATIONS AND PROJECTION CAPABILITIES

The hydrogeologic system in southwestern Salt Lake Valley is complex and cannot be defined completely with available data. The solute-transport model of southwestern Salt Lake Valley is based on a mathematical representation of ground-water flow and solute transport and on a simplified set of assumptions about the system. As a result, the model has limitations that must be considered when evaluating simulation results and the usefulness of the model as a tool to estimate future sulfate movement.

The principal limitations of the transport model that affect the accuracy of simulated sulfate movement include (1) uncertainties in model parameters and boundary conditions, (2) the simplified representation of the effects of chemical reactions between low-pH, high-sulfate ground water and aquifer material, and (3) the inability of the model to simulate the effects of density-driven flow on ground-water and solute movement. Because the transport model is coupled with a regional ground-water flow model of Salt Lake Valley (Lambert, 1995a), transport-model results are affected by uncertainty in both flow-model and transport-model parameters. Uncertainty in flow-model parameters and boundary conditions was theoretically reduced by calibration of the flow model to observed historical hydrologic conditions (Lambert, 1995a). The resulting set of calibrated parameters incorporated in the flow model, however, does not represent a unique solution. Different combinations of data entered into the flow model

might yield similar results. Also, available data defining hydrologic conditions used during calibration of the flow model were sparse in some areas of southwestern Salt Lake Valley and the accuracy of model results in those areas is poorly defined.

During calibration of the transport model, a consistent discrepancy between the simulated and observed orientation of the sulfate plume east of the Bingham Creek Reservoirs was observed. The discrepancy appears to be most substantial in the area from the reservoirs to about 2 to 3 mi to the east where the observed movement of the plume is to the southeast. The general direction of observed sulfate movement beyond that point appears to be eastward (fig. 13a), the flow direction simulated during model calibration. The discrepancy may be the result of limitations of the flow model's representation of recharge and aquifer properties in the vicinity of the reservoirs rather than errors in transport-model parameters.

During calibration of the ground-water flow model of Salt Lake Valley (Lambert, 1995a), it was noted that small adjustments in simulated recharge west of the reservoirs, representing inflow from consolidated rock, had a substantial effect on computed water levels near the margin of the valley and in the immediate vicinity of the reservoirs. The magnitude of the effects of changes in specified recharge at the mountain front on water levels decreased farther to the east. The discrepancy between model-computed and observed sulfate movement near the reservoirs may indicate that the distribution of specified recharge along the mountain front is inaccurate, resulting in a model-computed flow field that does not accurately represent actual conditions.

Errors in flow-model parameters defining aquifer properties in the vicinity of the Bingham Creek Reservoirs also might contribute to the discrepancy between model-computed and observed sulfate movement east of the reservoirs. Local heterogeneities in the aquifer near the reservoirs may cause solute to move along preferred pathways of sediment with relatively high hydraulic conductivity such as stream-channel deposits or other sedimentary features not represented in the model.

Simulation of future sulfate movement using the existing regional ground-water flow model of Salt Lake Valley (Lambert, 1995a) will not incorporate a southeasterly trend in ground-water flow or sulfate movement near the Bingham Creek Reservoirs. If the discrepancy between model-computed and observed sulfate movement in this area is, in fact, the result of errors in the model-computed flow field, projection simulations using the transport model may result in simulated migration of the sulfate plume that is too far to the north.

Another limitation of the ground-water flow model of Salt Lake Valley (Lambert, 1995a) that may affect the accuracy of projection transport-model simulations is the simplified representation of recharge from consolidated rock along the mountain front. Recharge from consolidated rock along the southwestern margin of the valley is specified in the ground-water flow model (Lambert, 1995a) and is not affected by simulated events in the ground-water flow system. In the physical system, however, flow from the consolidated-rock aquifer to the basin-fill aquifer is head dependent, controlled by the difference in water level between the two aquifers and the hydrologic properties existing at the contact between the two aquifers. Large declines in water level in the basin-fill aquifer locally near the margins of the valley may increase inflow from consolidated rock. Such effects on flow resulting from drawdowns near the margins of the valley are not simulated by the transport model. Temporal variation in recharge from consolidated rock in this area in response to declining water levels may affect hydraulic gradient, and thus the velocity of ground-water flow and sulfate movement near the margin of the valley.

Estimates of transport-model parameters selected from ranges of probable values were tested during model calibration. Although the final estimates were reasonable and within the defined ranges, variations in these estimates could produce a similar match between computed and observed patterns of sulfate movement. Considering the limited scope of transport-model calibration and the uncertainty in final parameter estimates, projection simulations using the transport model should be developed and analyzed incorporating a range of transport parameter values.

The transport model simulates the effects of complex chemical reactions between low-pH, high-sulfate ground water and the aquifer material on the movement of dissolved sulfate by using a nonlinear sorption isotherm to distribute sulfate between the dissolved and solid phase. Parameters defining the sorption isotherm were defined on the basis of sparse field data and on calibration of the transport model to the observed distribution of dissolved sulfate within the sulfate plume east of the Bingham Creek Reservoirs. Although the use of a sorption isotherm produced a reasonable match

between the computed and observed distributions of dissolved sulfate within the plume, the simplified representation of the effects of chemical reactions occurring in the aquifer will affect the accuracy of projected sulfate movement in projection simulations. It is not clear whether future simulations using the same approach and the same set of isotherm parameters will produce an accurate representation of future sulfate movement. Future attenuation of sulfate within the plume as it moves downgradient will depend, in part, on the availability of calcite in the aquifer and the ion-exchange capacity of the aquifer material. These properties of the aquifer downgradient of the sulfate plumes are not well defined, and the magnitude of the effects of chemical reactions on future attenuation of dissolved sulfate within the plume is unknown.

Ground-water and solute movement resulting from variation in fluid density is not simulated in the transport model. Variations in ground-water density do occur, however, immediately east of the Bingham Creek Reservoirs and may affect sulfate movement in that area.

Flow in a porous medium where ground-water density varies spatially may be driven by either differences in fluid pressures (differences in hydraulic head) or by unstable variations in fluid density, or both. Pressure-driven flows are directed from regions of high pressure (high hydraulic head) toward regions of lower pressure (low hydraulic head). Density-driven flows occur when gravitational forces act on denser regions of fluid and cause them to flow downward relative to fluids that are less dense. Evaluation of density-related effects on ground-water flows by Davies (1987) indicates that it is the relative magnitude of density-driven versus pressure-driven flow components that determines whether density effects will be significant in a given situation. Thus, even small variations in density in ground water can substantially affect ground-water flow if hydraulic gradients are small. Conversely, where hydraulic gradients are large, density-driven flow may not be significant even though density differences in ground water are large.

Ground-water density may vary substantially east of the Bingham Creek Reservoirs from zones of highly contaminated ground water to surrounding fresher water. Horizontal hydraulic gradients in the same area are relatively large, however, and it is possible that although density variations in ground water occur, pressure-driven flow may still dominate the flow system. Density effects on flow patterns in the area could include the movement of the zone of relatively-dense, low-pH, high-sulfate ground water in a more downward direction than would be defined by hydraulic gradients. It is also possible that high-density ground water could move in a different direction horizontally than surrounding ground water. This could occur if high-density water were moving down a sloping surface such as the contact between the principal aquifer and an underlying confining material. If the base of the aquifer were dipping in a different direction than ground-water flow as defined by hydraulic gradients, density-driven flow in the down-dip direction could influence the direction of movement of dense, high-sulfate ground water.

The capabilities and limitations of the transport model defined during model development and calibration must be considered when evaluating the utility of the model for projection of future sulfate movement. The strengths of the modeling approach include the ability of the model to incorporate, in three dimensions, regional ground-water flow conditions defined by hydrologic stresses occurring throughout the valley and not just within the transport-model area. Although the accuracy of model results is affected by the simplified representation of the actual system and by uncertainties in model parameters, analyses using the model can provide an important tool for the preliminary evaluation of sulfate movement based on regional changes in ground-water use and can provide valuable insight for future modeling work. Work necessary to reduce model limitations and improve the accuracy of future transport analyses using this or other computer models is discussed later in the report.

PROJECTED SULFATE MOVEMENT, 1994-2043

The transport model of southwestern Salt Lake Valley was used to estimate future sulfate movement on the basis of projected increased pumpage in the valley. Output from the flow-model simulation representing projected conditions from 1994-2043 was incorporated in the transport model for this purpose.

The projection transport simulation incorporated projected increases in pumping from public-supply wells and assumed no increases in pumping from wells for other purposes. The simulated potentiometric surface of the principal aquifer in the transport-model area computed at the end of the projection transport simulation is shown in figure 15. Ground water generally can be assumed to move perpendicular to potentiometric

contours. The simulated effects of withdrawals by public-supply and industrial wells east of the Bingham Creek Reservoirs on water-level altitudes and flow patterns are indicated in the model-computed potentiometric contours. Also, ground water is simulated as moving toward discharging wells in the south-central part of the valley from the west under the Jordan River, a pattern of flow that is not indicated in current water-level data.

Initial conditions that define starting sulfate-concentration values in model layers 3 and 4 used in the projection simulations were derived mainly on the basis of measured sulfate concentrations in 1993 (figs. 13 and 14). Data defining 1993 sulfate concentration in the aquifer system represented by model layers 1, 2, and 5-7 were sparse and starting concentrations in those model layers were based mainly on the final concentrations computed in the calibration transport simulation.

For the projection transport simulation it was assumed that mining-related sources of dissolved sulfate were eliminated through source-control actions established by Kennecott Utah Copper (1993, p. 3-2 to 3-4). Dissolved sulfate was simulated as entering the system only by means of natural recharge and from seepage from canals and irrigated fields (table 3).

Projection simulations were made using three sets of transport-parameter values defined from probable ranges of values described earlier in the report and from the results of model calibration (table 4). Different parameter-value sets were incorporated to simulate a probable range of model-computed transport rates and distributions of dissolved sulfate during the simulation period. Parameter set 1 contains parameter values determined during model calibration. In parameter set 2, effective porosity was decreased to 0.15 and the Freundlich isotherm constant (K_f) was decreased to 1×10^{-12} l^2/mg^2, thus decreasing simulated retardation of sulfate movement east of the Bingham Creek Reservoirs. Parameter set 2 was assumed to produce fast computed transport rates relative to calibrated parameter values. In parameter set 3, assumed to produce slow rates of sulfate movement relative to calibrated parameter values, porosity was increased to 0.35, and K_f was doubled to increase the simulated retarding effects of chemical reactions on sulfate movement immediately east of the reservoirs.

Distributions of dissolved sulfate in the model area were evaluated at 10, 25, and 50 years from the beginning of the projection simulation period (1994). Model-computed sulfate changes at cells containing selected public-supply wells in the area (fig. 15), including four wells northeast of the Bingham Creek Reservoirs operated by West Jordan City and two wells in the south-central part of the valley operated by Sandy City and Midvale City, were evaluated.

The model-computed distribution of dissolved sulfate in the principal aquifer for selected times and model layers based on calibrated parameter values in set 1 (table 4) is shown in figures 16 to 18. The simulated pattern of sulfate movement east of the Bingham Creek Reservoirs indicates the effects of ground-water withdrawals near the plume. The simulation projects movement of the plume during the 50-year period east toward the Kennecott production wells, northeast toward public-supply wells, and then east toward the Jordan River (figs. 16-18). The part of the plume characterized by sulfate concentration exceeding 5,000 mg/l is generally limited by dilution and simulated withdrawal from Kennecott production wells to the area west of the production wells. Simulation results also indicate the continued dilution of the sulfate plume east of the South Jordan evaporation ponds due to mixing and dispersion. The area of ground water east of the ponds with sulfate concentration greater than 500 mg/l in model layer 4 increases after the first 10 years in the simulation as simulated water levels decline and ground water moves from model layer 3 to model layer 4 (figs. 16-18). Ground water with sulfate concentration greater than 500 mg/l is simulated as moving west to east under the Jordan River toward public-supply wells during the final 25 years of the simulation period (2018 to 2043) in model layers 3 and 4 (figs. 17 and 18).

The simulated distributions of dissolved sulfate in southwestern Salt Lake Valley for 2018 based on the incorporation of parameter-value sets 2 and 3 (table 4) are shown in figures 19 and 20, respectively. The projection transport simulation incorporating parameter set 2 was assumed to produce the fastest transport rates of the three projection simulations. As indicated in figure 19, decreasing porosity and the Freundlich isotherm constant (K_f) in set 2 increased the rate of movement of the sulfate plume east of the Bingham Creek Reservoirs relative to computed results using the calibrated parameter values (fig. 17). In the projection simulation using parameter set 2 (table 4), withdrawals from the Kennecott production wells and the effects of mixing and dispersion decrease model-computed concentrations to less than 5,000 mg/l east of the reservoirs by 2018 (fig. 19).

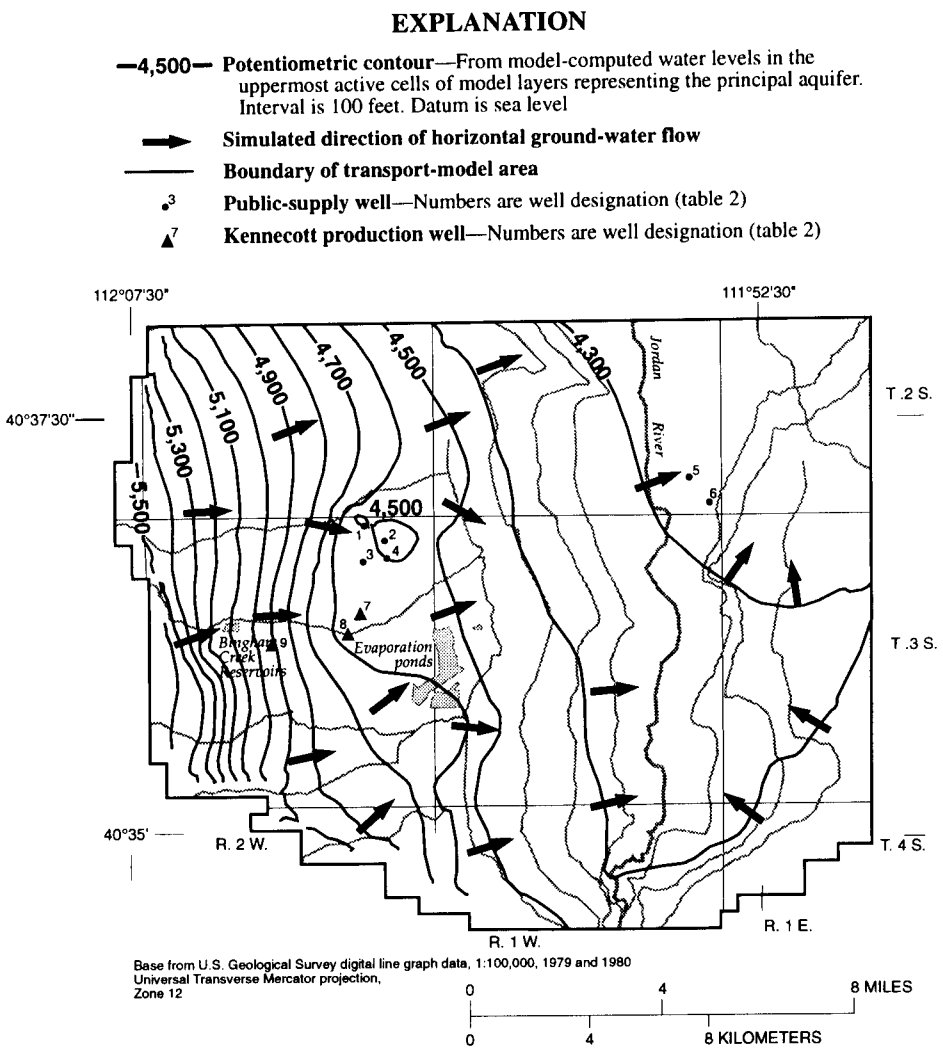

Figure 15. Model-computed potentiometric contours of the principal aquifer in southwestern Salt Lake Valley, Utah, for 2043, based on the simulation of projected increased pumpage.

Results of the projection transport simulation incorporating parameter-value set 3 in table 4 indicates slower movement of sulfate within the plume east of the Bingham Creek Reservoirs (fig. 20) relative to the results of the simulation using calibrated parameter values (fig. 17). Increasing porosity and the Freundlich isotherm constant (K_f) in set 3 effectively decreases the rate of movement of the zone of the plume characterized by sulfate concentration greater than 5,000 mg/*l*. In this simulation, the zone of the plume with sulfate concentration greater than 5,000 mg/*l* is present in 2018 and has not passed through the Kennecott industrial production wells (fig. 20). The results of the simulation also indicate that the plume, as defined by ground water with sulfate concentration greater that 500 mg/*l*, does not reach the public-supply wells northeast of the reservoirs by 2018.

Model-computed changes in sulfate concentration at cells containing public-supply wells in the transport-model area were recorded during projection transport simulations (fig. 21). Computed concentrations at these wells indicate a possible range of effects on sulfate concentration at these wells resulting from future migration of sulfate plumes.

Results of all three projection simulations indicate that the largest increase in computed sulfate concentration at public-supply wells northeast of the Bingham Creek Reservoirs occurred at West Jordan City Fire Station well, where computed concentrations increased from about 200 mg/*l* in 2006 to about 4,100 mg/*l* by 2022 in the projection simulation using parameter-value set 2. Model-computed sulfate concentration at the cell containing the West Jordan City Fire Station well did not increase substantially prior to 2010 regard-

Table 4. Sets of transport-parameter values incorporated in projection transport simulations in southwestern Salt Lake Valley, Utah

Transport parameter	Parameter values		
	Set 1	Set 2	Set 3
Porosity (dimensionless)	0.2	0.15	0.35
Longitudinal dispersivity (α_L) (in feet)	40	40	40
α_L / α_{TH} [1] (dimensionless)	.1	.1	.1
α_L / α_{TV} [2] (dimensionless)	.01	.01	.01
Freundlich isotherm constant (in liters squared per milligram squared)	4×10^{-12}	1×10^{-12}	8×10^{-12}
Freundlich isotherm exponent (dimensionless)	2	2	2

1 α_{TH} is horizontal-transverse dispersivity.
2 α_{TV} is vertical-transverse dispersivity.

less of the parameter-value set used. Sulfate concentrations at model cells containing three other West Jordan City wells northeast of the reservoirs generally did not exceed 400 mg/*l* during the projection simulations.

Model-computed sulfate concentrations increased at model cells containing the Midvale City Oakstreet and Sandy City Copperview public-supply wells east of the Jordan River (fig. 21). The largest increase at the cells containing the Midvale Oakstreet well, from 130 mg/*l* in 2016 to 240 mg/*l* in 2040, was simulated using parameter set 2 in table 4. Model-computed sulfate concentration at the model cell containing the Sandy City Copperview well increased in the simulation using parameter set 2 from about 150 mg/*l* in 2024 to about 340 mg/*l* in 2043. Model-computed sulfate concentration in that cell did not increase in the simulation using parameter set 3 (table 4).

NEED FOR FUTURE WORK

The hydrologic system of southwestern Salt Lake Valley is complex and not completely represented in current conceptual and computer models of the area. Although a substantial amount of data has been collected in the area during previous studies, more information is needed to improve the understanding of ground-water flow and solute movement in the area. Specific elements of the hydrologic system in southwestern Salt Lake Valley that are poorly defined and require additional investigation are discussed in the following paragraphs.

Estimates of the rate and distribution of recharge occurring along the mountain front in southwestern Salt Lake Valley incorporated in the transport model are based mainly on the results of calibration of the Salt Lake Valley flow model (Lambert, 1995a). Discrepancies observed during calibration of the transport model of southwestern Salt Lake Valley between model-computed and observed sulfate concentration east of the Bingham Creek Reservoirs may be the result of errors in the model-computed flow field near the mountain front. Field measurements of water levels in the bedrock aquifer and the adjacent basin-fill aquifer, and accurate estimates of the physical characteristics of the aquifer near the valley margin, are needed to better define the actual distribution of recharge along the mountain front. Field data defining water levels north and northwest of the reservoirs also are needed to determine the direction of ground-water flow near the reservoirs.

Results of the simulation analysis indicate that more data defining the physical properties of the basin-fill aquifer in the vicinity of the reservoirs may be

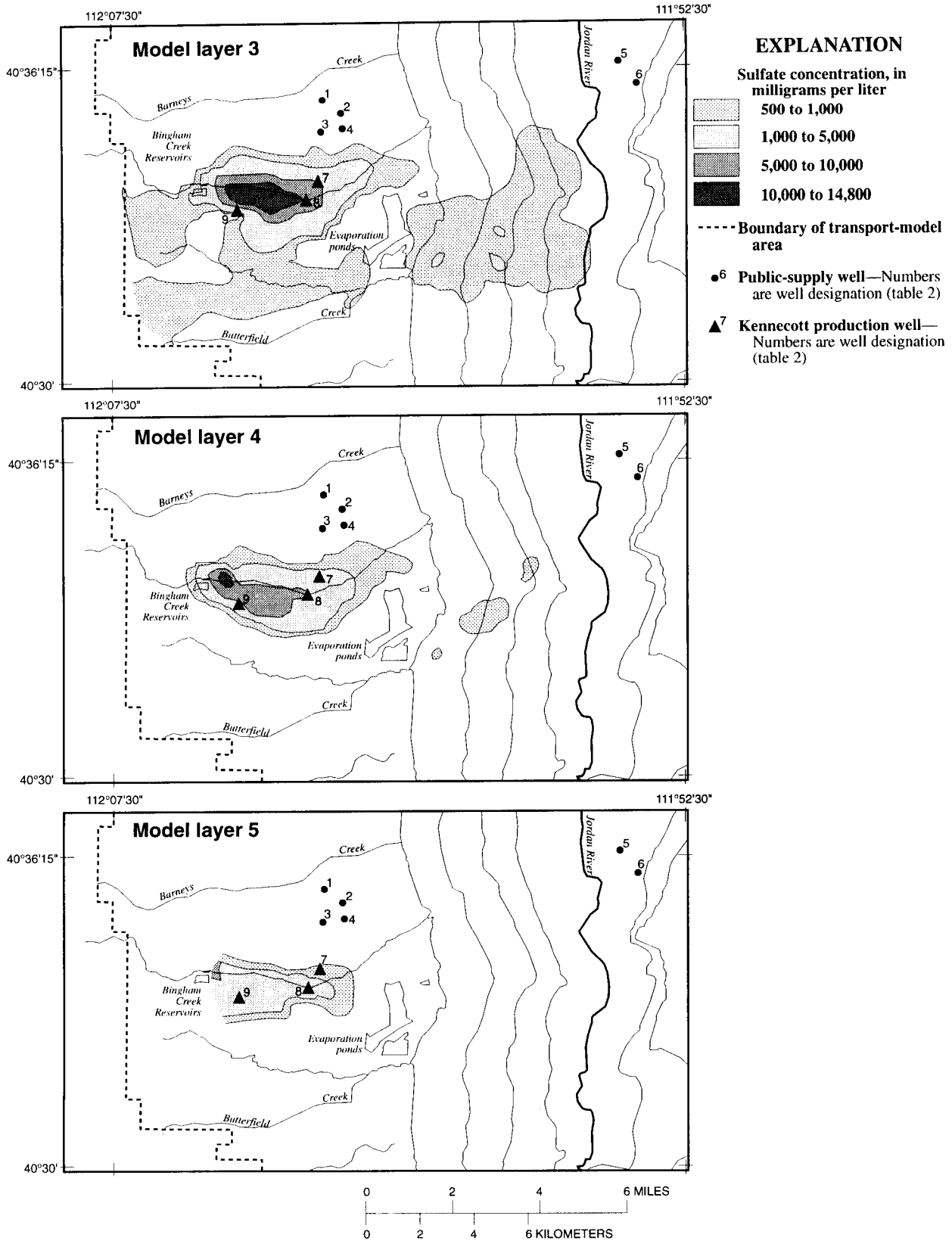

Figure 16. Model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, for 2003, based on calibrated transport-parameter values in set 1 in table 4.

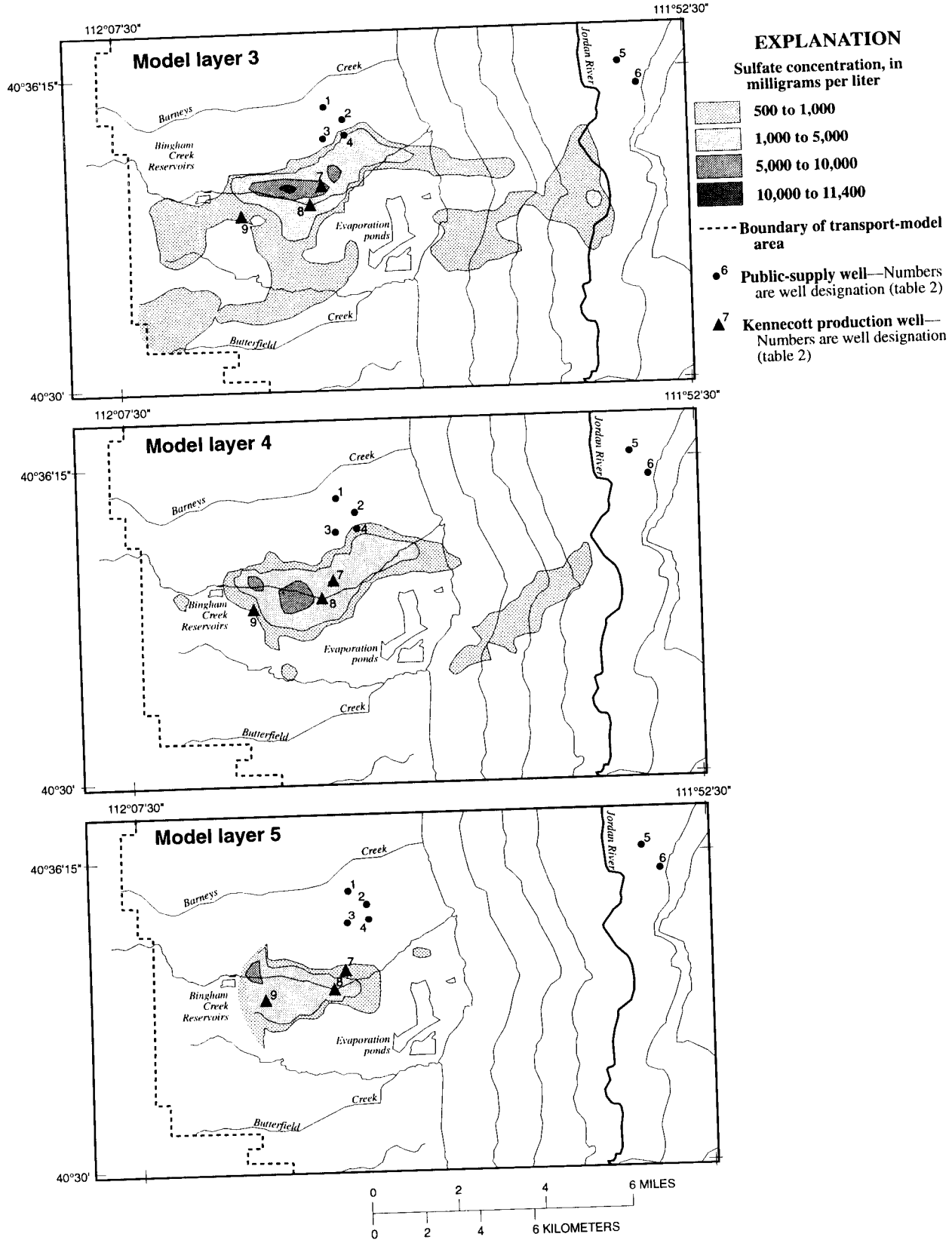

Figure 17. Model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, for 2018, based on calibrated transport-parameter values in set 1 in table 4.

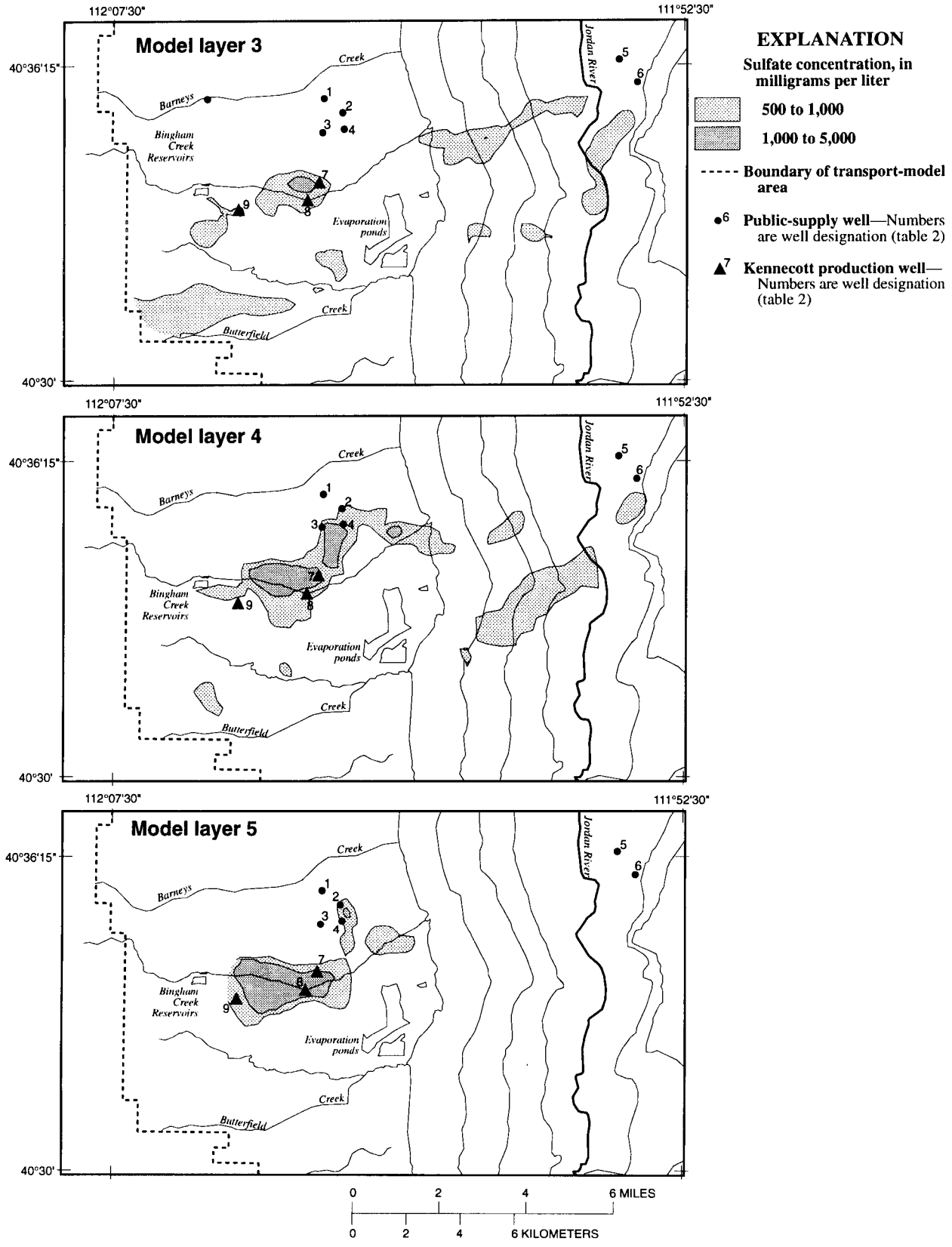

Figure 18. Model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, for 2043, based on calibrated transport-parameter values in set 1 in table 4.

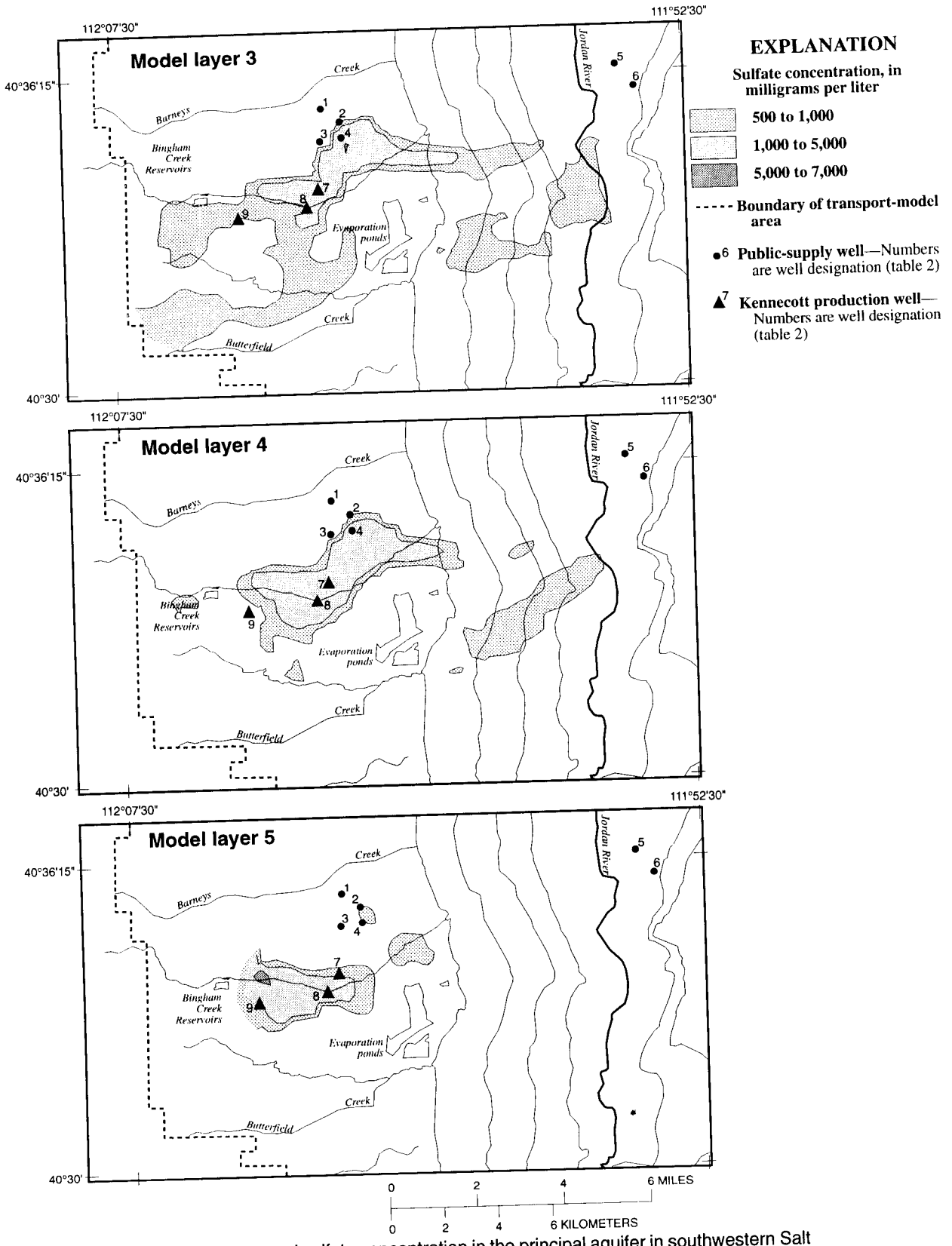

Figure 19. Model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, for 2018, based on transport-parameter values in set 2 in table 4.

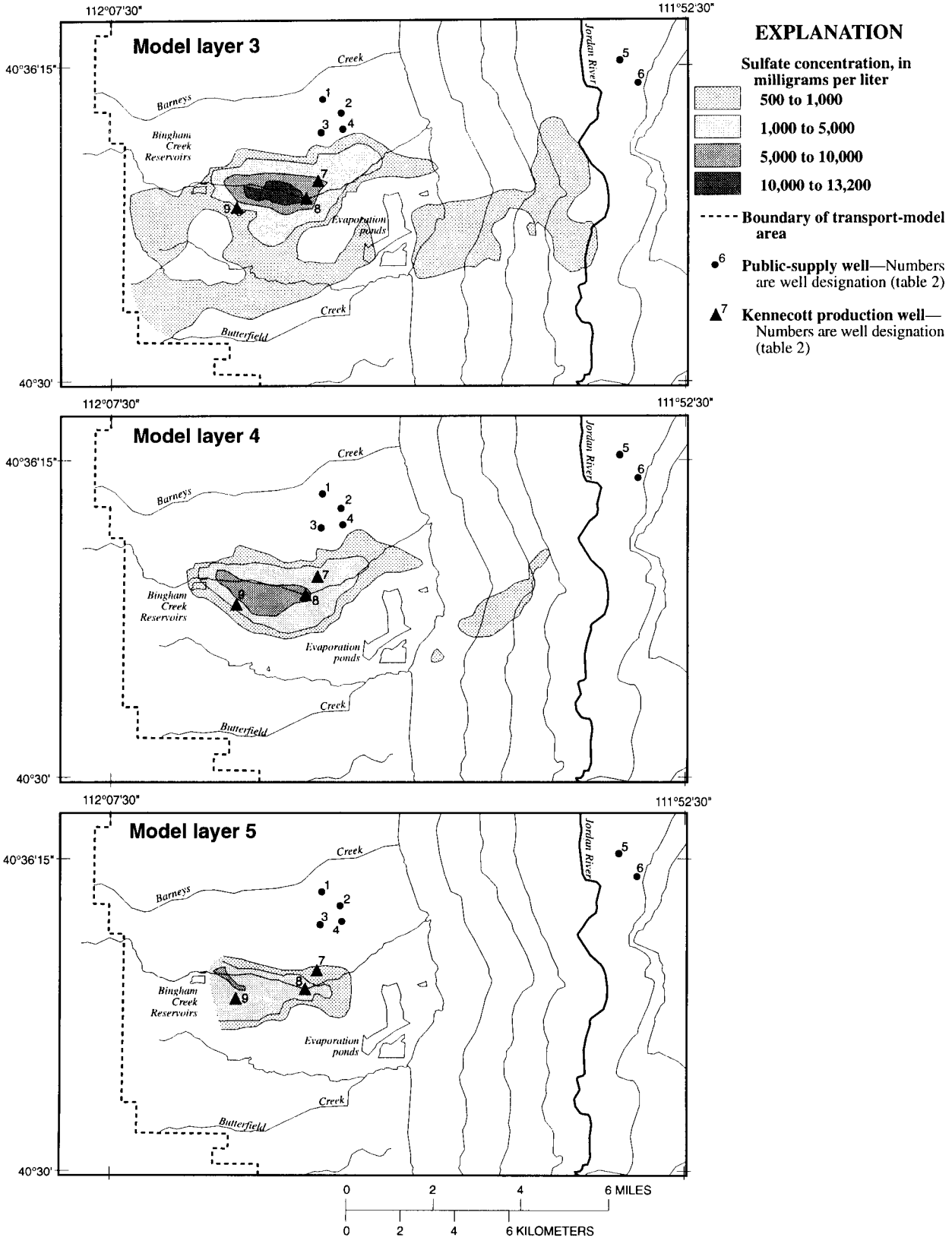

Figure 20. Model-computed sulfate concentration in the principal aquifer in southwestern Salt Lake Valley, Utah, for 2018, based on transport-parameter values in set 3 in table 4.

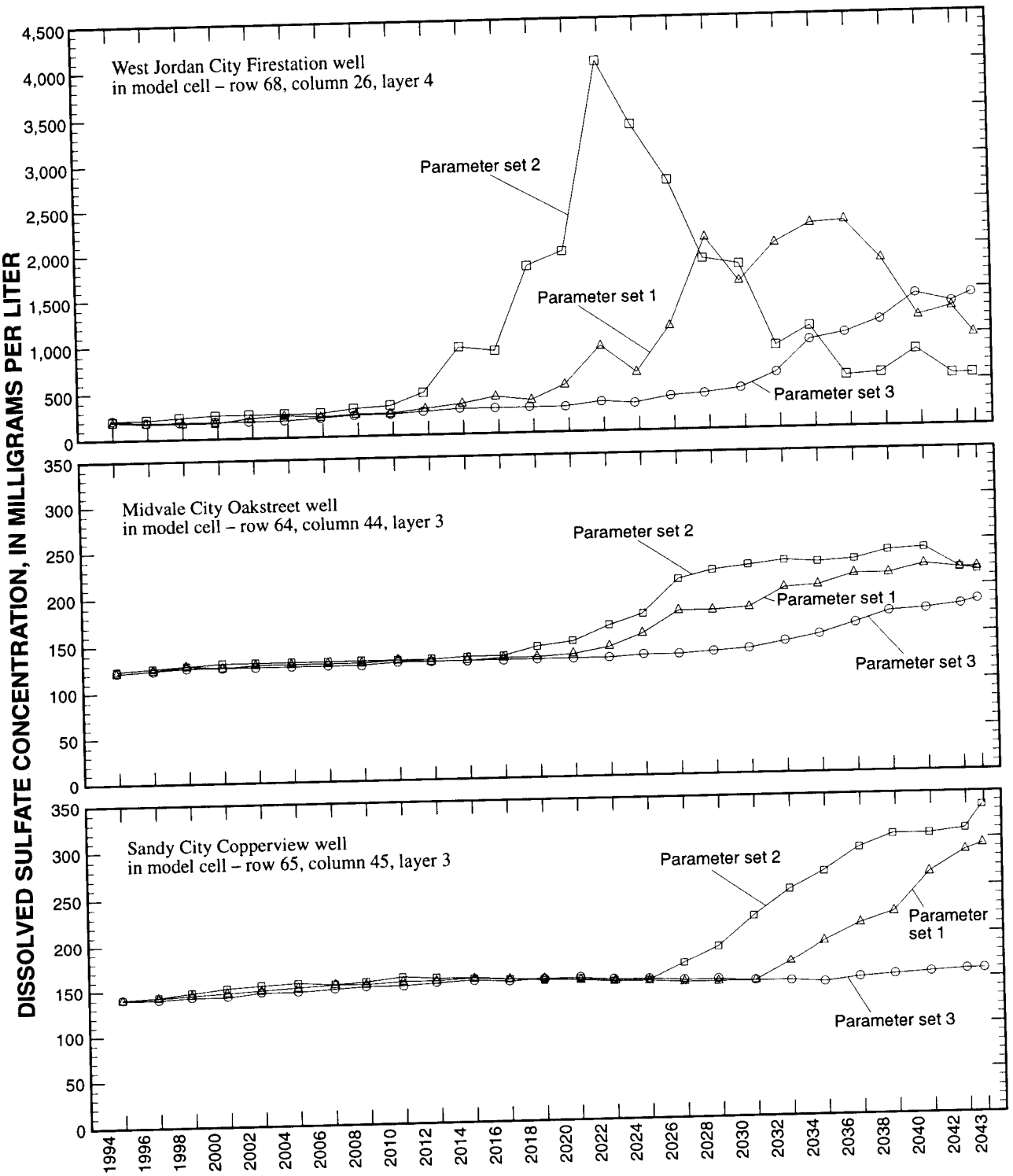

Figure 21. Simulated dissolved-sulfate concentration during 1993-2043 at selected model cells containing public-supply wells in southwestern and south-central Salt Lake Valley, Utah, based on three sets of transport-parameter values.

required to improve future transport simulations. Information describing the subsurface lithology of the aquifer east of the mouth of Bingham Canyon and measurements of the hydrologic properties of the aquifer can help determine if preferred pathways for ground-water and solute movement exist and allow for the incorporation of these features in future simulation analyses.

The current conceptual model of geochemical processes occurring in the low-pH part of the sulfate plume east of the Bingham Creek Reservoirs (Kennecott Utah Copper, 1993, p. 1-24–1-29 and A2-4–A2-8) is based on sparse field data. More site-specific data defining the solid-phase chemistry of aquifer material inside and outside of the plume are needed to better define these reactions and their effect on sulfate movement.

The representation of chemical reactions occurring in the low-pH zone of the sulfate plume east of the Bingham Creek Reservoirs is very limited. Future simulations should consider either adding more sophisticated chemical reactions to the MT3D transport model (Zheng, 1993) or using a separate geochemical model to simulate the reactions and coupling geochemical-model results to three-dimensional simulations of flow and transport.

As noted in the "Model limitations and projection capabilities" section of this report, the density of ground water probably varies in the vicinity of the Bingham Creek Reservoirs where highly contaminated ground water is surrounded by fresher water. Presently, it is not known whether variable-density ground water in this area has significantly affected ground-water flow patterns. A detailed analysis of hydraulic gradients and ground-water density would need to be done to determine if the movement of the plume east of the reservoirs will be affected by density-driven flow in the future or whether flow patterns will be defined mainly by hydraulic gradients.

Results of the analysis are limited by the representation of the system in the transport model based on available data. Presently, work is being done by Kennecott Utah Copper (1995) in southwestern Salt Lake Valley to improve the assessment of hydrologic conditions in the area and to evaluate possible alternatives to remediate the part of the ground-water system affected by seepage of leachate and mine-related wastewater. Analysis of these data could yield important insight into the effects of recent source-control measures and ongoing withdrawals from wells on ground-water and solute movement. Incorporation of new information, particularly new data defining the components of the hydrologic system discussed in this section into future simulation analyses of ground-water flow or solute movement, can reduce simulation limitations and improve the accuracy of simulation results.

SUMMARY

Southwestern Salt Lake Valley is the site of the Bingham Canyon Mine, one of the world's largest open-pit copper mines. Mine waste has been artificially leached in the area since the early 1930s. As a result, in part, of seepage from wastewater and leachate-collection systems along the foot of the Oquirrh Mountains and containment reservoirs and evaporation ponds east of the mouth of Bingham Canyon, ground water in parts of the principal aquifer between the mouth of the canyon and the Jordan River about 10 mi to the east has been contaminated. Ground water downgradient from the mouth of the canyon that has been affected by the seepage of leachate and wastewater is characterized by elevated concentrations of dissolved solids including sulfate, and in areas, low pH and elevated metal concentrations. Previous ground-water studies have indicated that the movement of contaminated ground water east of the canyon may affect the quality of water withdrawn in the future from nearby public-supply wells. To provide State officials and water users with information concerning the potential movement of contaminated ground water to points of withdrawal in the area, an analysis of contaminant transport was done using a solute-transport model of southwestern Salt Lake Valley coupled with a regional ground-water flow model of Salt Lake Valley.

The modular three-dimensional solute-transport model MT3D was used in conjunction with a previously developed ground-water flow model of Salt Lake Valley to simulate sulfate movement in ground water in southwestern Salt Lake Valley resulting from advection, dispersion, and chemical reactions. Development and calibration of the transport model focused mainly on the simulated movement of sulfate plumes east of the Bingham Creek Reservoirs and the South Jordan evaporation ponds, two of the principal contaminant sources in the area. Three transport-model simulations were made: (1) a preliminary transport simulation representing conditions prior to 1965, (2) a calibration transport simulation representing conditions during 1965-93, and (3) a projection transport simulation representing projected conditions for 1994-2043. The pre-

liminary transport simulation was made to estimate the distribution of sulfate in ground water in 1965 for use as starting conditions for the calibration simulation. Specified transport-model parameters were adjusted during the calibration simulation to match model-computed distributions of dissolved sulfate with the distributions defined from field measurements at selected times during the calibration period. After model calibration was completed, the transport model was used to simulate future sulfate movement for 1994-2043 on the basis of projected increases in municipal pumpage in Salt Lake Valley.

Available data were evaluated to define ranges of probable values for transport-model parameters incorporated in the model. Transport parameters adjusted during model calibration included porosity of the aquifer, longitudinal and transverse dispersivities, and distribution coefficients incorporated in the model to simulate the effects of chemical reactions on sulfate movement.

Comparisons between model-computed and observed patterns of sulfate distribution during calibration indicate that the transport model reproduced the general dimension of the sulfate plume east of the Bingham Creek Reservoirs and the general pattern of distribution of dissolved sulfate within the plume. A consistent discrepancy was observed during calibration, however, between the model-computed and the observed orientation of the axis of the plume. Results of the calibration simulation indicate sulfate movement in an easterly direction from the reservoirs, and the observed distributions of dissolved sulfate indicate movement of the plume in a southeasterly direction immediately downgradient from the reservoirs and movement in an easterly direction as the plume migrates toward the evaporation ponds. Efforts to more accurately simulate the observed orientation of the plume by adjusting transport-model parameters were unsuccessful. The discrepancy between the simulated and observed orientation of the plume may be the result of errors in the model-computed flow field caused by (1) errors in the simulated distribution of recharge west of the reservoirs or (2) errors in estimates of the properties of the aquifer in the vicinity of the reservoirs. If the discrepancy is, in fact, a result of the errors in the computed flow field, projection simulations using the transport model may result in simulated migration of the sulfate plume east of the reservoirs that is too far to the north.

The projection transport simulation incorporated projected increases in pumping from public-supply wells and assumed that mining-related sources of dissolved sulfate were eliminated as a result of source-control actions established by Kennecott Utah Copper. Initial conditions defining starting sulfate-concentration values for the projection simulation were based, where possible, on measured values. Considering the limitations of the transport model, which included uncertainty in transport parameter values, projection simulations were made using a range of parameter values. Three sets of transport-parameter values were used in the simulations including (1) estimated values resulting from model calibration, (2) values assumed to produce a fast rate of transport relative to calibrated values, and (3) values assumed to produce a slow rate of transport relative to calibrated values. The parameter values used in the three projection simulations were selected from ranges of probable values defined from available data and were used to provide a probable range of transport rates based on reasonable parameter estimates. Distributions of dissolved sulfate in the model area were evaluated at 10, 25, and 50 years from the beginning of the simulation period (1994). Model-computed sulfate changes at cells containing selected public-supply wells in the area, including four wells northeast of the Bingham Creek Reservoirs operated by West Jordan City, and two wells in the south-central part of the valley operated by Sandy City and Midvale City, were evaluated.

In general, results of the projection simulations indicate movement of the plume east of the Bingham Creek Reservoirs to the east toward Kennecott production wells, northeast toward public-supply wells, and then east toward the Jordan River. Ground water east of the South Jordan evaporation ponds with sulfate concentration greater than 500 mg/*l* is simulated as moving west to east under the Jordan River toward public-supply wells during the final 25 years of the simulation period in model layers 3 and 4. Rates of sulfate movement and magnitudes of sulfate-concentration fluctuations varied among the three projection simulations. Sulfate concentration computed in the simulation producing relatively fast transport rates increased at one West Jordan City well northeast of the reservoirs from 200 mg/*l* in 2006 to about 4,100 mg/*l* in 2022. That increase was the most substantial of the four West Jordan City wells. Results of the simulations also indicate that sulfate concentration in model cells containing public-supply wells in south-central Salt Lake Valley just east of the Jordan River increased as a result of

migration of the sulfate plume east of the South Jordan evaporation ponds. Model-computed sulfate concentration at a cell containing the Midvale City Oakstreet well increased from about 130 mg/*l* in 2016 to about 240 mg/*l* in2040 in the simulation using calibrated parameter values. Model-computed sulfate concentration at the model cell containing the Sandy City Copperview well incrxeased in the simulation producing relatively fast transport rates from about 150 mg/*l* in 2024 to about 340 mg/*l* in 2042. Model-computed sulfate concentration in that cell did not increase in the simulation producing relatively slow transport rates.

The hydrologic system in southwestern Salt Lake Valley is complex and not completely represented by the transport model that is documented in this report. Limitations of the model, including inaccuracies in model-computed flow fields and transport-model parameters, affect the accuracy of the simulation of future sulfate movement using the model. More information is needed to improve the conceptual model and future computer models of the system; particularly data defining the following system components: (1) the distribution of recharge from consolidated rock at the margin of the valley, (2) the hydrologic properties of the basin-fill aquifer in the vicinity of the Bingham Creek Reservoirs, (3) the geochemical processes taking place within the sulfate plume east of the Bingham Creek Reservoirs, and (4) the effects of density-driven flow on flow patterns in the vicinity of the reservoir.

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