science for a changing world

National Water-Quality Assessment Program

Conceptual Understanding and Groundwater Quality of Selected Basin-Fill Aquifers in the Southwestern United States

Professional Paper 1781

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

Conceptual Understanding and Groundwater Quality of Selected Basin-Fill Aquifers in the Southwestern United States

Edited by Susan A. Thiros, Laura M. Bexfield, David W. Anning, and Jena M. Huntington

National Water-Quality Assessment Program

Professional Paper 1781

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

U.S. Department of the Interior
KEN SALAZAR, Secretary

U.S. Geological Survey
Marcia K. McNutt, Director

U.S. Geological Survey, Reston, Virginia: 2010

For more information on the USGS—the Federal source for science about the Earth, its natural and living resources, natural hazards, and the environment, visit http://www.usgs.gov or call 1-888-ASK-USGS

For an overview of USGS information products, including maps, imagery, and publications, visit http://www.usgs.gov/pubprod

To order this and other USGS information products, visit http://store.usgs.gov

Suggested citation:

Entire publication
Thiros, S.A., Bexfield, L.M., Anning, D.W., and Huntington, J.M., eds., 2010, Conceptual understanding and groundwater quality of selected basin-fill aquifers in the Southwestern United States: U.S. Geological Professional Paper 1781, 288 p.

Example for section within the publication
Thiros, S.A., 2010, Section 2. Conceptual understanding and groundwater quality of the basin-fill aquifer in Salt Lake Valley, Utah, *in* Thiros, S.A., Bexfield, L.M., Anning, D.W., and Huntington, J.M., eds., 2010, Conceptual understanding and groundwater quality of selected basin-fill aquifers in the Southwestern United States: U.S. Geological Survey Professional Paper 1781, p. 13–24.

Foreword

The U.S. Geological Survey (USGS) is committed to providing the Nation with reliable scientific information that helps to enhance and protect the overall quality of life and that facilitates effective management of water, biological, energy, and mineral resources (http://www.usgs.gov/). Information on the Nation's water resources is critical to ensuring long-term availability of water that is safe for drinking and recreation and is suitable for industry, irrigation, and fish and wildlife. Population growth and increasing demands for water make the availability of that water, measured in terms of quantity and quality, even more essential to the long-term sustainability of our communities and ecosystems.

The USGS implemented the National Water-Quality Assessment (NAWQA) Program in 1991 to support national, regional, State, and local information needs and decisions related to water-quality management and policy (http://water.usgs.gov/nawqa). The NAWQA Program is designed to answer: What is the quality of our Nation's streams and groundwater? How are conditions changing over time? How do natural features and human activities affect the quality of streams and groundwater, and where are those effects most pronounced? By combining information on water chemistry, physical characteristics, stream habitat, and aquatic life, the NAWQA Program aims to provide science-based insights for current and emerging water issues and priorities. From 1991 to 2001, the NAWQA Program completed interdisciplinary assessments and established a baseline understanding of water-quality conditions in 51 of the Nation's river basins and aquifers, referred to as Study Units (http://water.usgs.gov/nawqa/studies/study_units.html).

In the second decade of the Program (2001–2012), a major focus is on regional assessments of water-quality conditions and trends. These regional assessments are based on major river basins and principal aquifers, which encompass larger regions of the country than the Study Units. Regional assessments extend the findings in the Study Units by filling critical gaps in characterizing the quality of surface water and groundwater, and by determining water-quality status and trends at sites that have been consistently monitored for more than a decade. In addition, the regional assessments continue to build an understanding of how natural features and human activities affect water quality. Many of the regional assessments employ modeling and other scientific tools, developed on the basis of data collected at individual sites, to help extend knowledge of water quality to unmonitored, yet comparable areas within the regions. The models thereby enhance the value of our existing data and our understanding of the hydrologic system. In addition, the models are useful in evaluating various resource-management scenarios and in predicting how our actions, such as reducing or managing nonpoint and point sources of contamination, land conversion, and altering flow and (or) pumping regimes, are likely to affect water conditions within a region.

Other activities planned during the second decade include continuing national syntheses of information on pesticides, volatile organic compounds (VOCs), nutrients, trace elements, and aquatic ecology; and continuing national topical studies on the fate of agricultural chemicals, effects of urbanization on stream ecosystems, bioaccumulation of mercury in stream ecosystems, effects of nutrient enrichment on stream ecosystems, and transport of contaminants to public-supply wells.

The USGS aims to disseminate credible, timely, and relevant science information to address practical and effective water-resource management and strategies that protect and restore water quality. We hope this NAWQA publication will provide you with insights and information to meet your needs, and will foster increased citizen awareness and involvement in the protection and restoration of our Nation's waters.

The USGS recognizes that a national assessment by a single program cannot address all water-resource issues of interest. External coordination at all levels is critical for cost-effective management, regulation, and conservation of our Nation's water resources. The NAWQA Program, therefore, depends on advice and information from other agencies—Federal, State, regional, interstate, Tribal, and local—as well as nongovernmental organizations, industry, academia, and other stakeholder groups. Your assistance and suggestions are greatly appreciated.

William H. Werkheiser
USGS Associate Director for Water

This page intentionally left blank.

Contents

Conversion Factors, Datum, and Abbreviations and Acronyms

Conversion Factors

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
acre	0.004047	square kilometer (km^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m^3)
million gallons (Mgal)	3,785	cubic meter (m^3)
acre-foot (acre-ft)	1,233	cubic meter (m^3)
	Flow rate	
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year (m^3/yr)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
foot per year (ft/yr)	0.3048	meter per year (m/yr)
inch per year (in./yr)	25.4	millimeter per year (mm/yr)
	Mass	
pound, avoirdupois (lb)	0.4536	kilogram (kg)
ton per year (ton/yr)	0.9072	metric ton per year
	Hydraulic conductivity	
foot per day (ft/d)	0.3048	meter per day (m/d)
	Hydraulic gradient	
foot per mile (ft/mi)	0.1894	meter per kilometer (m/km)
	Radioactivity	
picocurie per liter (pCi/L)	0.037	becquerel per liter (Bq/L)
	Transmissivity*	
foot squared per day (ft^2/d)	0.09290	meter squared per day (m^2/d)

*Transmissivity: The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft^3/d)/ft^2]ft. In this report, the mathematically reduced form, foot squared per day (ft^2/d), is used for convenience.

Temperature in degrees Celsius (°C) and degrees Fahrenheit (°F) may be converted by using the following equations:

$$°F=(1.8\times°C)+32$$

$$°C=(°F-32)/1.8$$

Conversion Factors and Datum—Continued

Datum

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88), except as otherwise noted on some figures where the National Geodetic Vertical Datum of 1929 (NGVD 29) was used or a vertical datum was not specified. Altitude, as used in this report, refers to distance above the vertical datum. Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

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

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

Milligrams per liter is equivalent to parts per million (ppm) and micrograms per liter is equivalent to parts per billion (ppb).

Tritium content in water is reported as tritium units or picocuries per liter. The ratio of 1 atom of tritium to 10^{18} atoms of hydrogen is equal to 1 tritium unit or 3.2 picocuries per liter.

Abbreviations and Acronyms

CAP	Central Arizona Project
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFC	chlorofluorocarbon
CUP	Central Utah Project
ET	evapotranspiration
GAMA	Groundwater Ambient Monitoring and Assessment (California program)
InSAR	Synthetic aperture radar interferometry
LRL	laboratory reporting level
MCL	Maximum Contaminant Level
MRL	minimum reporting level
NAWQA	National Water-Quality Assessment (USGS program)
NLCD	National Land Cover Database (USGS)
NWIS	National Water Information System (USGS)
RASA	Regional Aquifer-System Analysis (USGS program)
SWPA	Southwest Principal Aquifers (NAWQA)
TANC	Transport of Anthropogenic and Natural Contaminants to Public-Supply Wells (NAWQA topical study)
VOC	volatile organic compound
WWTP	wastewater treatment plant

Conversion Factors and Datum—Continued

Organizations

ADEQ	Arizona Department of Environmental Quality
CDWR	Colorado Division of Water Resources
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

Selected Chemical Names

CFC-11	Trichlorofluoromethane
CFC-12	Dichlorodifluoromethane
CFC-113	Trichlorotrifluoroethane
BTEX	Benzene, Toluene, Ethylbenzene, Xylene
DBCP	1,2-Dibromo-3-chloropropane
DDE	1,1-Dichloro-2,2-bis(chlorophenyl)ethylene (degradation product of DDT)
DDT	1,1,1-Trichloro-2,2-bis(chlorophenyl)ethane
^{3}H-^{3}He	Tritium-helium-3
MTBE	Methyl *tert*-butyl ether
PCE	Perchloroethene, tetrachloroethylene, tetrachloroethene
TCA	1,1,1-Trichloroethane
TCE	Trichloroethene

Conceptual Understanding and Groundwater Quality of Selected Basin-Fill Aquifers in the Southwestern United States

Edited by Susan A. Thiros, Laura M. Bexfield, David W. Anning, and Jena M. Huntington

Executive Summary

The National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey (USGS) has been conducting a regional analysis of water quality in the principal aquifer systems in the southwestern United States (hereinafter, "Southwest") since 2005. Part of the NAWQA Program, the objective of the Southwest Principal Aquifers (SWPA) study is to develop a better understanding of water quality in basin-fill aquifers in the region by synthesizing information from case studies of 15 basins into a common set of important natural and human-related factors found to affect groundwater quality.

The synthesis consists of three major components:

1. Summary of current knowledge about the groundwater systems, and the status of, changes in, and influential factors affecting quality of groundwater in basin-fill aquifers in 15 basins previously studied by NAWQA (this report).
2. Development of a conceptual model of the primary natural and human-related factors commonly affecting groundwater quality, thereby building a regional understanding of the susceptibility and vulnerability of basin-fill aquifers to contaminants.
3. Development of statistical models that relate the concentration or occurrence of specific chemical constituents in groundwater to natural and human-related factors linked to the susceptibility and vulnerability of basin-fill aquifers to contamination.

As illustrated by the sections in this report describing the groundwater and water-quality characteristics of the 15 case-study basins, similarities in the hydrogeology, land- and water-use practices, and water-quality issues for the SWPA study area enable a regional analysis of those characteristics. Regional analysis begins by determining the primary factors that affect water quality—and the associated susceptibility and vulnerability of basin-fill aquifers to contamination—on the basis of data and information from a subset of information-rich, basin-fill aquifers in the study area. Conceptual and mathematical models formed for these basins can then be used to provide insight on areas that are hydrologically similar, but that are lacking groundwater-quality data and interpretive studies, or on areas where water development has not progressed as far as in the modeled basins. Regional-scale models and other decision-support tools that integrate aquifer characteristics, land use, and water-quality monitoring data will help water managers to evaluate water-quality conditions in unmonitored areas, to broadly assess the sustainability of water resources for future supply, and to help develop cost-effective groundwater monitoring programs.

Basin-fill aquifers occur in about 200,000 mi^2 of the 410,000 mi^2 SWPA study area and are the primary source of groundwater supply for cities and agricultural communities. Four of the principal aquifers or aquifer systems of the United States are included in the basin-fill aquifers of the study area: (1) the Basin and Range basin-fill aquifers in California, Nevada, Utah, and Arizona; (2) the Rio Grande aquifer system in New Mexico and Colorado; (3) the California Coastal Basin aquifers; and (4) the Central Valley aquifer system in California. Because of the generally limited availability of surface-water supplies in the arid to semiarid climate, cultural and economic activities in the Southwest are particularly dependent on supplies of good-quality groundwater. Irrigation and public-supply withdrawals from basin-fill aquifers in the study area account for about one quarter of the total withdrawals from all aquifers in the United States.

Basin-fill aquifers in the Southwest consist primarily of sand and gravel deposits that partly fill structurally formed depressions and are bounded by mountains. In some areas, fine-grained deposits of silt and clay are interbedded with the more permeable sand and gravel deposits, forming confining units that impede the movement of groundwater. The primary source of natural recharge to the deep parts of most basin-fill aquifers is precipitation on the surrounding mountains.

Mountain runoff seeps into the coarse-grained stream-channel and alluvial-fan deposits near the basin margins or enters the basin as subsurface inflow from consolidated rock. Low precipitation rates combined with high evaporation rates in the Southwest result in a relatively small contribution of groundwater recharge from precipitation that falls on the basin floor (generally less than 5 percent of annual precipitation). Before human development of water resources began in the alluvial basins, discharge from the groundwater systems typically resulted from evapotranspiration from the lowest parts of the basins and along stream channels, from springs, and as seepage to streams flowing through the basin. Artesian conditions exist in the groundwater discharge areas of several basins where the upward flow of water is impeded by low-permeability layers of clay, creating large vertical hydraulic gradients. Constrictions in the surrounding consolidated rock and faulting restrict groundwater flow out of many of the basins.

Although there are many similarities between the SWPA case-study basins and their aquifers, there are also major differences. For example, basin areas range from about 23 mi^2 for Eagle Valley in Nevada, to about 20,000 mi^2 for the Central Valley in California. Population densities in 2005 ranged from about 15 people/mi^2 in the San Luis Valley in Colorado and New Mexico to about 7,000 people/mi^2 in the Santa Ana Coastal Basin in California. The area of irrigated agriculture in the case-study basins in 2001 ranged from less than 1 percent in Las Vegas Valley in Nevada and in the Upper Santa Cruz Basin in Arizona to about 60 percent in the Central Valley.

Water development has caused considerable change in some basin groundwater systems in the Southwest. Imported surface water and the redistribution of water from within the basin to areas that previously did not receive recharge have resulted in increased flow velocities, greater saturated thicknesses, and changes in flow directions for some basins. Recharge from excess irrigation water and discharge by pumping groundwater for irrigation and public supply are much greater than natural sources of recharge and discharge in some basins. For example, groundwater recharge under modern conditions is about seven times that of predevelopment conditions in the West Salt River Valley (Phoenix area) in Arizona and about six times that of predevelopment conditions in the Central Valley. The infiltration of pumped groundwater and surface water applied for irrigation has resulted in recharge water that has been exposed to agricultural chemicals and natural salts concentrated by evapotranspiration. Infiltration of this water changes the chemistry of groundwater in the shallow part of the aquifer system. Other artificial or human-related sources of recharge to Southwest basins include seepage of water applied to lawns; seepage from canals, leaky distribution and sewer pipes, and septic systems; infiltration at retention basins, recharge basins, and dry wells used to receive storm runoff; and seepage of treated wastewater through irrigated fields and through streambeds as a means of disposal or artificial recharge.

Withdrawal from wells has become the primary source of groundwater discharge from many of the basins at the expense of discharge to streams and evapotranspiration. Water-level declines and changes in flow directions and magnitudes occur where groundwater withdrawals are large. Water levels in the west-central part of West Salt River Valley have declined between 300 and 400 ft since the early 1900s. Recharge and discharge associated with water development have resulted in an increase in the flow of water through parts of many basin-fill groundwater systems, especially flow from the land surface to shallow and unconfined parts of the aquifer. Water development, therefore, typically results in aquifers being more susceptible to water-quality degradation by human activities at the land surface and more vulnerable to contamination where contaminant sources are present.

Many factors influence the quality of groundwater in the 15 case-study basins, but some common factors emerge from the basin summaries presented in this report. These factors include the chemical composition of the recharge water, consolidated rock geology and composition of aquifer materials derived from consolidated rock, and land and water use. Groundwater is generally oxic (oxygen-rich) in the coarser grained alluvial-fan deposits and is usually anoxic (oxygen-poor) in the finer grained deposits that are predominant near the centers of the basins. Geochemically reduced conditions commonly occur in discharge zones where long flow paths terminate and residence time and organic matter content increase.

The amount of coarse-grained sediments near the land surface can be a major factor in the susceptibility of groundwater to nitrate contamination. Sediment texture influences rates of infiltration and groundwater flow, which in turn control how rapidly water at the land surface (which may have elevated concentrations of nitrate as a result of human activities) can infiltrate the soil and move downward into the aquifer. Elevated concentrations of nitrate have been measured in shallow groundwater in many of the case-study basins. Probable sources of nitrate in the groundwater include leaching of applied nitrate fertilizers, flushing of natural vadoze-zone deposits, irrigation using treated sewage effluent, leaking sewer pipes, infiltration of water contaminated by animal waste, and septic-system effluent.

The effects of human activities on groundwater quality are most commonly observed in shallow parts of the basin-fill aquifers. Where the vertical hydraulic gradient is downward and where confining layers are discontinuous, the potential exists for contaminants from the land surface to be transported through shallow saturated sediment to deeper parts of the aquifer. Pumping and resulting alterations of hydraulic gradients can cause changes in groundwater quality by enhancing the downward movement of shallow groundwater and the vertical or lateral movement of water from adjacent bedrock to parts of the basin-fill aquifer used for water supply.

Chloroform, a byproduct of the chlorination of water for drinking, was the most frequently detected volatile organic compound (VOC) in groundwater sampled from urban areas of the case-study basins. Possible sources of chloroform in shallow groundwater include leaky water distribution lines and sewer pipes and the use of disinfected public-supply water to irrigate lawns and gardens. The pesticide atrazine and its degradation product deethylatrazine were among the most frequently detected pesticides in groundwater samples collected from the case-study basins. Although the concentrations of these compounds are typically very small and not a health concern, their presence in the aquifer indicates the potential for their movement from the land surface and the possibility that higher concentrations may occur in the future.

The major water-quality issues in many of the developed case-study basins are increased concentrations of dissolved solids, nitrate, and VOCs in groundwater as a result of human activities. For instance, most of the recharge to the three Santa Ana groundwater basins in southern California occurs artificially at facilities that receive local streamflow, treated municipal wastewater, or imported surface water, all of which have influenced groundwater quality. The addition of water to the basin-fill deposits in the Coastal Basin of the Santa Ana Basin by artificial recharge and the removal of water by pumping have increased the lateral rate of groundwater flow through the system, resulting in a widespread distribution of chemicals in the recharge areas. Although the Coastal Basin is a highly urbanized area, wells downgradient from the recharge areas are screened in confined aquifers that are generally insulated from the effects of overlying land uses. The confining layers impede the vertical movement of water from the land surface and make this part of the aquifer less vulnerable to contaminant sources in the immediate area.

Water imported from Lake Mead has enabled population growth in Las Vegas Valley. Recharge to the shallow groundwater system, mostly from excess landscape irrigation water (known as secondary recharge), is increasing with the expansion of urban areas in the valley, especially onto the areas underlain by coarse-grained sediments near the mountain fronts. This recharge water has to move through natural barriers of fine-grained sediment and caliche to recharge the deeper groundwater system. The mixing of secondary recharge water and artificially recharged, imported surface water with native groundwater could potentially result in an increase in concentrations of dissolved solids in parts of the basin-fill aquifer.

In the Salt Lake Valley in Utah, seepage of excess water from irrigated crops and urban turf areas, and from leaking canals, water distribution pipes, sewer lines, storm drains, and retention basins are now sources of recharge to the basin-fill aquifer. This valley recharge is more susceptible to transporting man-made chemicals than is runoff from the mountains (mountain-front recharge) and subsurface inflow from the adjacent mountains (mountain-block recharge). Dissolved-solids concentrations have increased more than 20 percent in some areas near the Jordan River and on the east side of the valley over approximately a 10-year period. Groundwater pumping has caused the vertical and lateral groundwater-flow gradients to change, which could allow shallow groundwater or water from other parts of the deeper aquifer with higher concentrations of dissolved solids to reach the wells in these areas.

Changes in urban water-supply strategies through time to ensure efficient use of limited regional water sources can introduce new potential effects on groundwater quality. For example, because of limited groundwater availability, a water-supply strategy was recently (during 2008) implemented in the Middle Rio Grande Basin of New Mexico to replace most groundwater pumping for public supply to Albuquerque residents with direct use of surface water. Additional strategies being implemented or planned to reduce groundwater withdrawals include the use of treated municipal wastewater, recycled industrial wastewater, and nonpotable surface water to irrigate urban turf areas. These water sources have the potential to impact groundwater quality in new ways if an unconsumed (excess) component recharges the basin-fill aquifer.

The information presented and the citations listed in this report serve as a resource for those interested in the groundwater-flow systems in the NAWQA case-study basins. The summaries of water-development history, hydrogeology, conceptual understanding of the groundwater system under both predevelopment and modern conditions, and effects of natural and human-related factors on groundwater quality presented in the sections on each basin also serve as a foundation for the synthesis and modeling phases of the SWPA regional study.

This page intentionally left blank.

Section 1.—Conceptual Understanding and Groundwater Quality of Selected Basin-Fill Aquifers in the Southwestern United States—Background and Study Approach

By Susan A. Thiros, Laura M. Bexfield, David W. Anning, Jena M. Huntington, and Tim S. McKinney

Introduction

The National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey (USGS) has been conducting a regional analysis of water quality in the principal aquifer systems in the southwestern United States (hereinafter, "Southwest") since 2005. The Southwest Principal Aquifers (SWPA) study within the NAWQA Program is building a better understanding of the factors that affect water quality in basin-fill aquifers in the region by synthesizing the baseline knowledge of groundwater-quality conditions in basin-fill aquifers previously studied by the Program. Resulting improvements in the understanding of the sources, movement, and fate of contaminants are assisting in the development of tools that water managers can use to help assess aquifer susceptibility and vulnerability to contamination. Regional assessments are being done across the country that focus on water-quality issues of concern at the principal-aquifer scale (Lapham and others, 2005).

The ease with which water enters and moves through an aquifer is described as its intrinsic susceptibility (Focazio and others, 2002). Aquifer susceptibility is dependent on the aquifer properties and other characteristics such as recharge rate, the presence or absence of an overlying confining unit, vertical hydraulic gradient, groundwater travel time, thickness and characteristics of the unsaturated zone, and pumping. The vulnerability of groundwater to contamination is the probability for contaminants to reach a specified part of an aquifer after being introduced, usually at the land surface. Vulnerability to contamination is dependent on the properties of the groundwater system (susceptibility), the existence of contaminant sources, and the contaminant's chemical characteristics.

Purpose and Scope

This report documents and provides a review of the conceptual models and water-quality conditions for basin-fill aquifers in 15 case-study basins in the SWPA study area. Specifically, each basin summary describes the following:

1. A conceptual model of the groundwater-flow system in the basin, how it has been modified by development, and groundwater quality conditions that are based on published reports of NAWQA studies and other investigations.

2. Effects of components of the groundwater-flow system and other natural and human-related factors on groundwater quality in the basin-fill aquifers, with a focus on factors that contribute to the susceptibility of the aquifer and the vulnerability of the groundwater to contamination.

The information presented and citations listed in this report serve as a resource for those interested in the groundwater-flow systems in the NAWQA case-study basins. The basin summaries also serve as a foundation for subsequent development of regional-scale conceptual models and statistical models of the primary factors affecting water quality of basin-fill aquifers in the Southwest.

Background on the Southwest Principal Aquifers Study

Basin-fill aquifers occur in about 200,000 mi^2 of the 410,000 mi^2 SWPA study area and are the primary source of groundwater supply for cities and agricultural communities. In several areas, these aquifers provide baseflow to streams that support important aquatic and riparian habitats. When aggregated across the study area, the basin-fill aquifers comprise four of the principal aquifers or aquifer systems of the United States: (1) the Basin and Range basin-fill aquifers in California, Nevada, Utah, and Arizona; (2) the Rio Grande aquifer system in New Mexico and Colorado; (3) the California Coastal Basin aquifers; and (4) the Central Valley aquifer system in California (fig. 1; U.S. Geological Survey, 2003a).

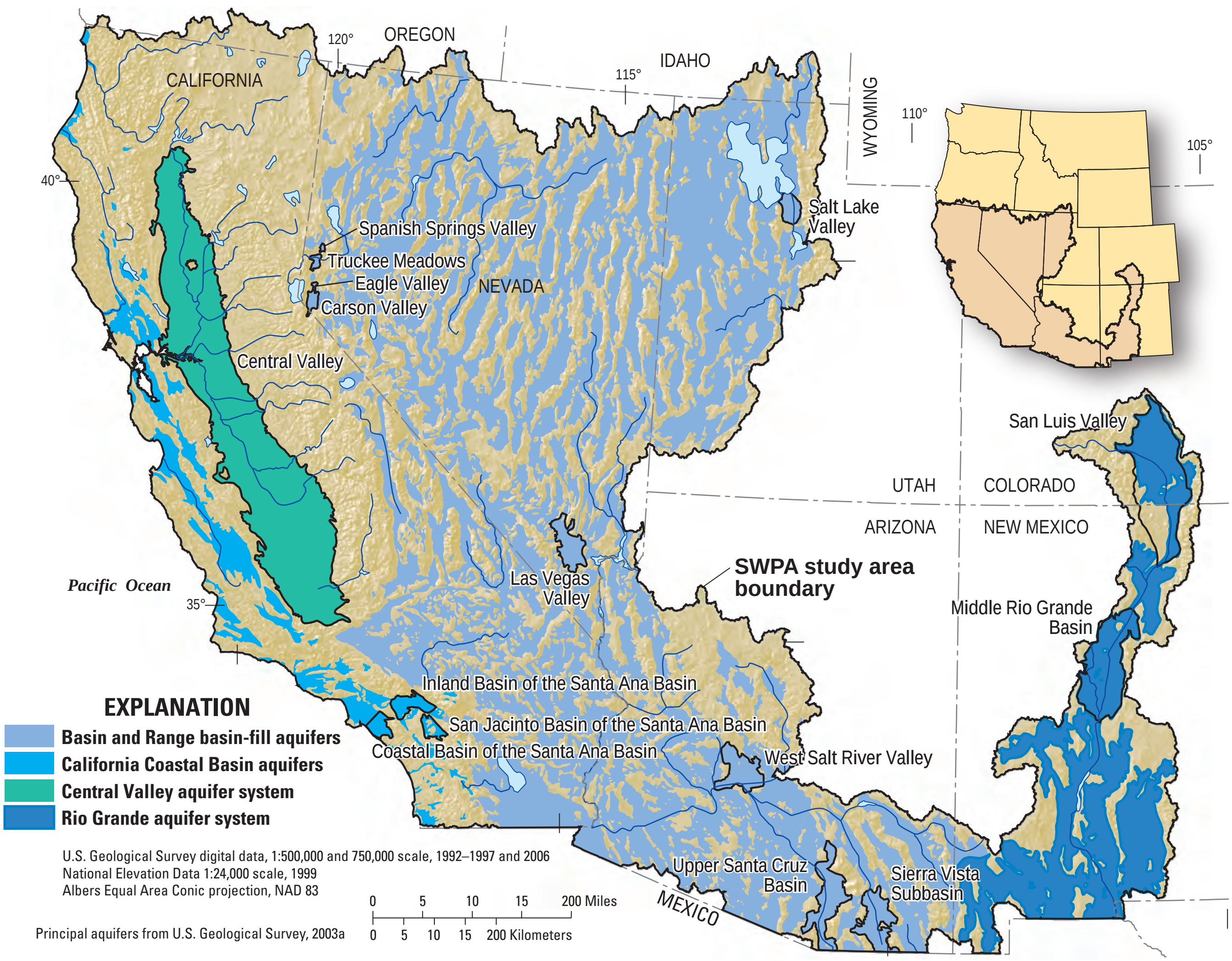


Figure 1. Principal aquifers and locations of 15 basins previously studied by the National Water-Quality Assessment Program in the Southwest Principal Aquifers (SWPA) study area.

About 46.6 million people live in the SWPA study area (Oak Ridge National Laboratory, 2005), mostly in urban metropolitan areas; a smaller percentage live in rural agricultural communities that tend about 14.4 million acres of cropland (U.S. Geological Survey, 2003b). Other rural areas have small communities with mining, retirement, and(or) tourism- and recreational-based economies. Because of the generally limited availability of surface-water supplies in parts of the Southwest, cultural and economic activities in the region are particularly dependent on good-quality groundwater supplies. In the year 2000, about 33.7 million acre-ft of surface water was diverted from streams and about 23.0 million acre-ft of groundwater was withdrawn from aquifers in the SWPA study area, mostly for agricultural uses (U.S. Geological Survey, 2004). Withdrawals from basin-fill aquifers in the study area for irrigation and public supply in the year 2000 were about 18.0 million acre-ft and 4.1 million acre-ft, respectively, and together account for about one quarter of the total withdrawals from all aquifers in the United States (Maupin and Barber, 2005, table 1). Although irrigation and public supply are the primary uses of groundwater in the study area, water use varies locally by basin, and withdrawals for industrial, mining, and electric power generation are also significant in some areas.

Basin-Fill Aquifers

Basin-fill aquifers in the Southwest consist primarily of sand and gravel deposits that partly fill structurally formed depressions that are commonly bounded by mountains (fig. 2). In some areas, silt and clay layers interbedded with the more-permeable sand and gravel deposits form confining units that impede the vertical movement of groundwater. Most basins contain thousands of feet of deposits, and the sediments become more compacted and less permeable with depth and in the topographically lower parts of basins. Many basins are drained by a stream that flows through a gap in the surrounding consolidated rock or they coalesce with a topographically lower basin, although some are closed basins from which groundwater and surface water are removed naturally only by evapotranspiration.

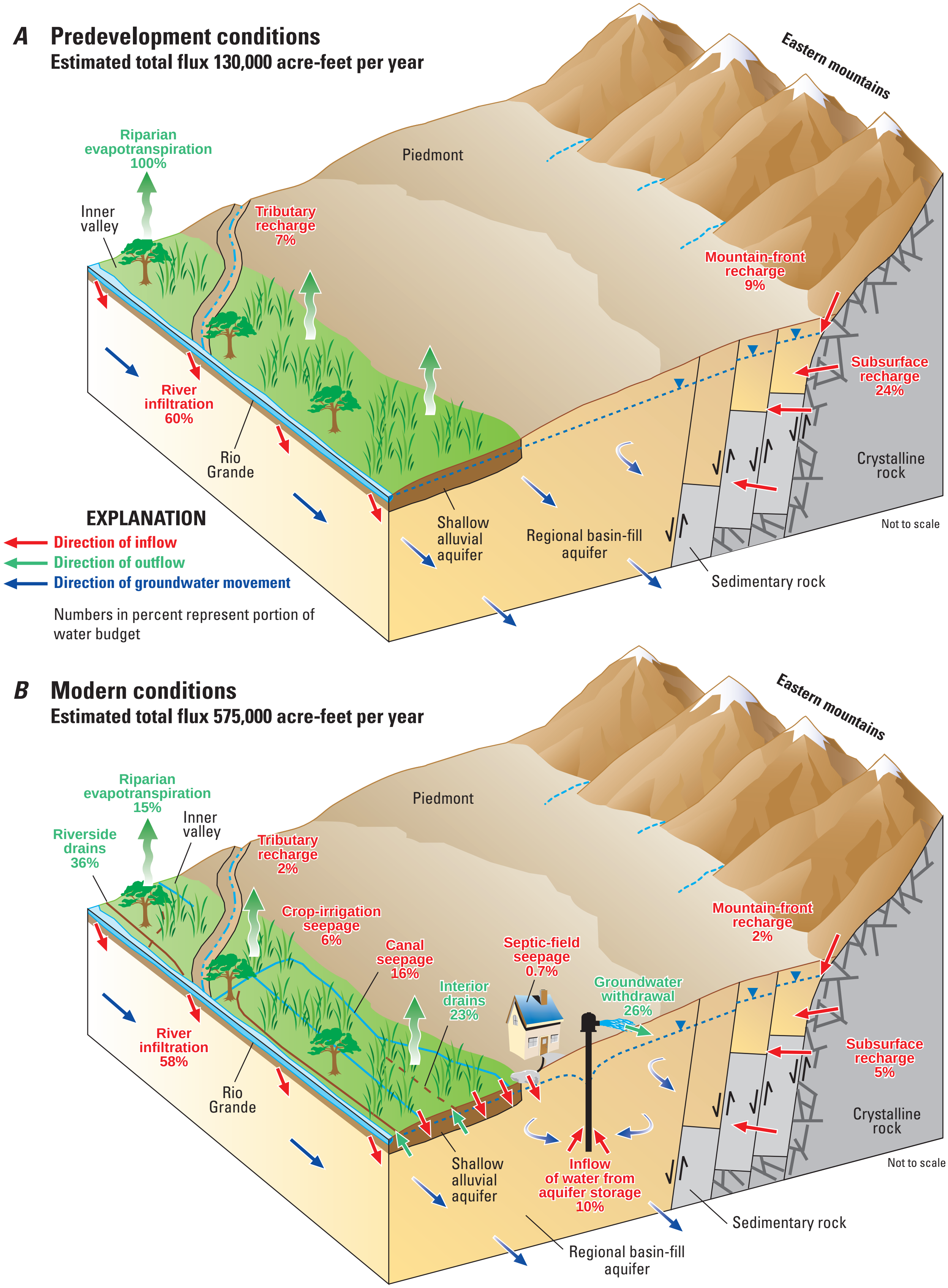


Figure 2. Generalized diagram for the Middle Rio Grande Basin, New Mexico, showing components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Generally, high-energy streams form alluvial fans of coarse-grained deposits along the mountain fronts, where a thick unsaturated zone is underlain by an unconfined aquifer. Steep alluvial fans transition to a relatively flat valley floor, where lacustrine and fluvial depositional environments commonly have created layers of fine-grained sediment interbedded with more permeable layers of sand and gravel. In groundwater discharge areas in the topographically lowest parts of some basins, this depositional sequence results in confined and artesian conditions as the upward flow of water is impeded by fine-grained layers of sediment. Somewhat continuous clay layers typically occur within about 100 ft of the land surface in many basins, forming the base of a shallow aquifer system that can be perched or that can contribute to or receive water from the underlying confined aquifer.

The primary source of natural recharge to the deeper parts of most Southwest basin-fill aquifers is precipitation on the surrounding mountains. Mountain runoff seeps into the coarse-grained stream-channel and alluvial-fan deposits near the basin margins. Precipitation also can infiltrate the consolidated rock of the mountains where the rock is fractured or weathered and move into the basin-fill deposits as subsurface inflow. Low precipitation rates combined with high evaporation rates in the Southwest result in a relatively small contribution of groundwater recharge from precipitation that falls on the basin floor (generally less than 5 percent of annual precipitation). Mountain-front recharge to the basin-fill aquifer includes both the runoff and subsurface inflow components.

Before human development of water resources began in the alluvial basins, sources of discharge from the groundwater systems typically were evapotranspiration, springs, and seepage to streams flowing through the basin. Constrictions in the surrounding bedrock and faulting restrict groundwater flow out of many of the basins. Playa lakes or wet playas were present in the topographically low areas of basins with no through-flowing drainage. Artesian areas existed in several basins where groundwater flowed to the land surface through layers of less permeable material. The cities of Las Vegas (Nevada), Tucson (Arizona), and San Bernardino (California) owe their locations to the availability of groundwater that historically discharged to streams or springs throughout the year.

Changes to the Basin-Fill Aquifers

Some basin groundwater systems in the Southwest have changed considerably with water development. Imported surface water and the redistribution of water from various sources to areas that previously did not receive recharge have resulted in increased flow velocities, greater saturated thicknesses, and changes in flow directions for some basins. New sources of recharge include seepage of excess irrigation water applied to crops and lawns; seepage from canals, leaking water-distribution and sewer pipes, and septic systems; infiltration of stormwater runoff from retention basins, recharge basins and wells used to receive runoff (dry wells); and seepage of treated wastewater through streambeds or irrigated fields as a means of disposal, artificial recharge, or as excess irrigation water. As an example of the effects of water development on an aquifer, the change in groundwater recharge and discharge in the Middle Rio Grande Basin from predevelopment to modern conditions is shown in figure 2.

Withdrawal from wells has become the primary path of groundwater discharge from many of the basins at the expense of discharge to streams and evapotranspiration. Water-level declines and changes in flow directions and magnitudes occur where groundwater withdrawals are large. The recharge and discharge quantities associated with water development have resulted in the acceleration of flow through parts of many basin-fill groundwater systems, especially from the land surface to the shallower parts of the aquifer. Groundwater withdrawals from relatively deep wells that are typically used for public supply also have resulted in enhanced movement of groundwater from shallower to deeper parts of basin-fill aquifers. Water development and urbanization, therefore, typically result in aquifers that are more susceptible to water-quality degradation by human activities occurring at the land surface and more vulnerable to contamination where contaminant sources are present. Changes in flow directions, geochemical conditions, or vertical mixing in a groundwater system that has small rates of flow, long residence times, and slow rates of contaminant degradation can make treatment of contaminated groundwater difficult. Contamination can affect whether the groundwater resource can feasibly be used as a drinking-water supply for many years. Therefore, it is imperative to understand the natural and human-related factors associated with the susceptibility and vulnerability of these aquifers to contamination, allowing water managers to plan for their optimal protection and utilization.

Regional Analysis

Similarities in the hydrogeology, land- and water-use practices, and water-quality issues within the SWPA study area allow for regional analysis. Regional analysis begins by determining the primary factors that affect water quality—and the associated susceptibility and vulnerability of basin-fill aquifers to contamination—on the basis of data and information from a subset of information-rich basin-fill aquifers in the study area. Conceptual and mathematical models formed for these basins can then be used to provide insight on areas that are hydrologically similar, but that are lacking groundwater-quality data and interpretive studies, or on areas where water development has not progressed as far as in the modeled basins. Regional analysis, therefore, is a cost-effective means of providing water managers of multiple basins with information that could be used to determine the likely level of susceptibility and vulnerability of their aquifers to contamination.

During its first data-collection and analysis phase from 1991 to 2001, NAWQA Program scientists sampled wells and established baseline water-quality conditions for basin-fill aquifers in 15 basins across the study area (fig. 1 and table 1). Groundwater quality also was evaluated for its relation to natural and human-related factors on the basis of a wide suite of constituents, including major ions, nutrients, trace elements, pesticides, and volatile organic compounds (VOCs). These studies resulted in the identification and detailed understanding of local conditions and factors affecting groundwater quality in each basin. The SWPA study described here develops a regional understanding by synthesizing information from the 15 case-study basins into a common set of important natural and human-related factors found to affect water quality in basin-fill aquifers across the Southwest. The synthesis consists of the following major components:

1. Summary of current knowledge about the groundwater systems, and the status of, changes in, and influential factors affecting groundwater quality of basin-fill aquifers in the 15 basins previously studied by NAWQA (this report).

2. Development of a conceptual model of the primary natural and human factors commonly affecting groundwater quality, thereby building a regional understanding of the susceptibility and vulnerability of basin-fill aquifers to contaminants.

3. Development of statistical models that relate the concentration or occurrence of specific chemical constituents in groundwater to natural and human-related factors linked to the susceptibility and vulnerability of basin-fill aquifers to contamination.

Regional-scale models and other decision-support tools that integrate aquifer characteristics, land use, and water-quality monitoring data will help water managers to evaluate water-quality conditions in unmonitored areas, to broadly assess the sustainability of water resources for future supply, and to help develop cost-effective groundwater monitoring programs.

Table 1. Alluvial basins in the southwestern United States described in this report.

Section	Case-study alluvial basin	Principal aquifer system
2	Salt Lake Valley, Utah	Basin and Range basin-fill aquifers
3	Truckee Meadows, Nevada	Basin and Range basin-fill aquifers
4	Eagle Valley, Nevada	Basin and Range basin-fill aquifers
4	Carson Valley, Nevada	Basin and Range basin-fill aquifers
5	Spanish Springs Valley, Nevada	Basin and Range basin-fill aquifers
6	Las Vegas Valley, Nevada	Basin and Range basin-fill aquifers
7	West Salt River Valley, Arizona	Basin and Range basin-fill aquifers
8	Upper Santa Cruz Basin, Arizona	Basin and Range basin-fill aquifers
9	Sierra Vista Subbasin of the Upper San Pedro Basin, Arizona	Basin and Range basin-fill aquifers
10	San Luis Valley, Colorado and New Mexico	Rio Grande aquifer aystem
11	Middle Rio Grande Basin, New Mexico	Rio Grande aquifer aystem
12	San Jacinto Basin of the Santa Ana Basin, California	California Coastal Basin aquifers
12	Inland Basin of the Santa Ana Basin, California	California Coastal Basin aquifers
12	Coastal Basin of the Santa Ana Basin, California	California Coastal Basin aquifers
13	Central Valley, California	Central Valley aquifer system

Previous Regional Investigations of Basin-Fill Aquifers in the Southwest

Previous NAWQA SWPA reports have described groundwater quality in the Southwest from a regional perspective. Anning and others (2007) studied the spatial distribution of dissolved solids in basin-fill aquifers and streams in the Southwest, along with sources of dissolved solids and the factors that affect observed concentrations. The effects of agricultural and urban land use on the quality of shallow groundwater was evaluated by Paul and others (2007) using data collected by the NAWQA Program for the SWPA study area from 1993 to 2004. Other USGS studies of large areas in the Southwest include those of the Regional Aquifer-System Analysis (RASA) and the Regional Groundwater Availability programs. In many of these studies, computer models were used to develop estimates of water availability at the time of the study and into the future. The National Ground Water Atlas, which was compiled using RASA findings (Miller, 1999), includes maps and information on the principal aquifer systems described in this report.

Publications that describe components of the groundwater budgets for several basin-fill aquifers in the Southwest include those by Hogan and others (2004), Anning and Konieczki (2005), Paschke (2007), Stonestrom and others (2007), and Reilly and others (2008). Hogan and others (2004) and Stonestrom and others (2007) focus particularly on arid and semiarid recharge mechanisms and quantities. Anning and Konieczki (2005) focus on classification of basins based on hydrogeologic characteristics, whereas Reilly and others (2008) focus on groundwater availability. Paschke (2007) includes discussion of regional groundwater budgets, general groundwater quality characteristics, and areas that groundwater-flow models simulated as contributing recharge to public-supply wells in four basins within the SWPA study area.

Study Approach

For each of the NAWQA case-study basins, information needed to understand the basin's groundwater system and its water-quality characteristics was compiled and presented in an individual section on the basin in this report (table 1). A spatial dataset of natural and human-related factors that may affect groundwater quality in the basin-fill aquifers of the Southwest was developed for the SWPA study (McKinney and Anning, 2009) and was used as the basis for describing the case-study basins. This dataset includes physical characteristics of the region such as geology, elevation, and precipitation, as well as human-related factors, such as population, land use, and water use.

Each section contains a basin overview and a description of the water-development history, hydrogeology, conceptual understanding of the groundwater system under both predevelopment and modern conditions, and the effects of natural and human-related factors on groundwater quality in the basin. The information was gathered from existing publications and summarized to provide a complete conceptual model for use in the next phase of the SWPA study, which is to synthesize the compiled information for the individual basins to provide a regional perspective on how water quality in Southwest basin-fill aquifers is affected by various natural and human-related factors. Some of the basins have more information available on the groundwater system and water quality than others, resulting in longer and more detailed sections.

The conceptual models presented in this report are formed from the results of previous studies, some of which included the construction of a numerical groundwater-flow model. Recharge to and discharge from the case-study basin-fill aquifers were separated into budget components that were generally consistent across the basins, such as recharge from precipitation on the basin and along the mountain front; subsurface inflow from bedrock and other basins; seepage from excess applied irrigation, canals, and artificial recharge facilities; and discharge from evapotranspiration, springs, wells, seepage to streams, and subsurface outflow from the basin. Estimates for groundwater recharge and discharge components under predevelopment and modern conditions are based, whenever possible, on flow-model simulations that utilize some measured data, such as water levels and engineered recharge amounts, and a calibration process to determine unmeasured components, such as subsurface inflow and outflow. For basins without available flow models, groundwater budgets have been compiled from information gleaned from other reports or were estimated for this study. The estimated budgets do not represent a rigorous analysis of individual budget components, and some estimates may be less certain than others. The groundwater budgets presented in this report are intended only to provide a basis for comparing the overall magnitude of recharge and discharge between predevelopment and modern conditions in a basin and to allow for comparisons across the case-study basins.

Concentrations of selected constituents and compounds in groundwater from the case-study basins were compared with drinking-water standards established by the U.S. Environmental Protection Agency (2009). Primary drinking-water standards limit the concentration levels of specific contaminants that can adversely affect public health. Examples of primary drinking-water standards are 10 milligrams per liter (mg/L) for nitrate (measured as nitrogen), 0.010 mg/L for arsenic, and 0.003 mg/L for the pesticide atrazine. These standards are the maximum contaminant levels (MCL) that are legally allowed in public water systems to protect drinking-water quality. Secondary drinking-water standards are non-enforceable guidelines for contaminants that may cause changes in cosmetic or aesthetic effects such as taste, odor, or color. Examples of secondary drinking-water standards are 500 mg/L for total dissolved solids and 250 mg/L for sulfate.

A variety of environmental tracers were used in many of the case-study basins to help determine the susceptibility of groundwater to the effects of human activities at the land surface. Most commonly, these tracers were used to estimate groundwater "age," which is defined as the time since the water being sampled reached the water table. The presence of at least a fraction of groundwater less than about 50 years old typically indicates parts of an aquifer that are susceptible to water-quality effects from human activities at the land surface. The quality of older groundwater that does not contain a discernable fraction of water that recharged within the past 50 years typically is considered not to have been affected by human activities, but rather by natural factors. One environmental tracer, tritium, which occurs naturally in precipitation, is a radioactive isotope of hydrogen with a half life of 12.4 years. Large amounts of tritium were introduced into the atmosphere by nuclear testing beginning in the early 1950s (atmospheric testing was banned in 1963). The presence of tritium in groundwater above a threshold concentration is used as an indicator that at least a component of the water was recharged since the early 1950s, and therefore, is "young."

The presence of chlorofluorocarbons (CFCs) in groundwater also is used as an indicator of young water and as a tool for estimating specific groundwater ages. CFCs are man-made organic compounds that are used in industrial processes and in the home. After their introduction in the 1930s, atmospheric concentrations increased nearly exponentially until the 1990s (Plummer and Busenberg, 2000). Three specific CFCs—CFC-11, CFC-12, and CFC-113—have long residence times and uniform concentrations in the atmosphere, making them valuable groundwater tracers once incorporated into the hydrologic cycle (Solomon and others, 1998; Cook and Herczeg, 2000). In populous areas, CFC contamination from leaking sewage systems and other sources besides the atmosphere is a good indicator of aquifer susceptibility to human activities, even though a specific age cannot be estimated for groundwater that has been contaminated with urban sources of CFCs.

Carbon-14 is a naturally occurring radioactive isotope of carbon that can be useful to estimate the age of "old" groundwater, or water that recharged an aquifer between about 1,000 and 40,000 years ago (Coplen, 1993). Most carbon-14 present in water that recharges an aquifer results from contact with carbon dioxide in the soil zone and(or) atmosphere. Knowledge of groundwater flow paths and the geochemical processes likely to affect carbon-14 along flow paths is necessary to properly adjust the carbon-14 measured in a groundwater sample prior to estimating an age through half-life calculations. Other factors, such as the addition of carbon-14 to the atmosphere through thermonuclear testing, also must be taken into account to arrive at an appropriate age estimate. Detailed discussions of the use of carbon-14 in estimating groundwater age can be found in Kalin (2000) and Kazemi and others (2006).

References Cited

Anning, D.W., Bauch, N.J., Gerner, S.J., Flynn, M.E., Hamlin, S.N., Moore, S.J., Schaefer, D.H., Anderholm, S.K., and Spangler, L.E., 2007, Dissolved solids in basin-fill aquifers and streams in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2006–5315, 168 p. Available at http://pubs.usgs.gov/sir/2006/5315/.

Anning, D.W., and Konieczki, A.D., 2005, Classification of hydrogeologic areas and hydrogeologic flow systems in the Basin and Range Physiographic Province, southwestern United States: U.S. Geological Survey Professional Paper 1702, 37 p. Available at http://pubs.usgs.gov/pp/2005/pp1702/.

Cook, P.G., and Herczeg, A.L., eds., 2000, Environmental tracers in subsurface hydrology: Boston, Kluwer Academic Publishers, 529 p.

Coplen, T.B., 1993, Uses of environmental isotopes, *in* Alley, W.M., ed., Regional ground-water quality: New York, Van Nostrand Reinhold, p. 227–254.

Focazio, M.J., Reilly, T.E., Rupert, M.G., and Helsel, D.R., 2002, Assessing ground-water vulnerability to contamination—Providing scientifically defensible information for decision makers: U.S. Geological Survey Circular 1224, 33 p. Available at http://pubs.usgs.gov/circ/2002/circ1224/.

Hogan, J.F., Phillips, F.M., and Scanlon, B.R., eds., 2004, Groundwater recharge in a desert environment—The southwestern United States: American Geophysical Union, Water Science and Applications Series, v. 9, 294 p.

Kalin, R.M., 2000, Radiocarbon dating of groundwater systems, *in* Cook, P.G., and Herczeg, A.L., eds., Environmental tracers in subsurface hydrology: Boston, Kluwer Academic Publishers, p. 111–144.

Kazemi, G.A., Lehr, J.H., and Perrochet, Pierre, 2006, Groundwater age: Hoboken, N.J., John Wiley and Sons, 325 p.

Lapham, W.W., Hamilton, P.A., and Myers, D.N., 2005, National Water-Quality Assessment Program—Cycle II regional assessments of aquifers: U.S. Geological Survey Fact Sheet 2005–3013, 4 p. Available at http://pubs.usgs.gov/fs/2005/3013/pdf/PASforWeb.pdf.

Maupin, M.A., and Barber, N.L., 2005, Estimated withdrawals from principal aquifers in the United States, 2000: U.S. Geological Survey Circular 1279, 46 p. Available at http://pubs.usgs.gov/circ/2005/1279/.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

Miller, J.A., 1999, Ground water atlas of the United States—Introduction and national summary: U.S. Geological Survey Hydrologic Atlas 730, 15 p. Available at http://pubs.usgs.gov/ha/ha730/ch_a/index.html.

Oak Ridge National Laboratory, 2005, LandScan global population database, accessed May 17, 2010 at http://www.ornl.gov/landscan/.

Paschke, S.S., ed., 2007, Hydrogeologic settings and ground-water flow simulations for regional studies of the transport of anthropogenic and natural contaminants to public-supply wells-studies begun in 2001: U.S. Geological Survey Professional Paper 1737–A, variously paged. Available at http://pubs.usgs.gov/pp/2007/1737a/.

Paul, A.P., Seiler, R.L., Rowe, T.G., and Rosen, M.R., 2007, Effects of agriculture and urbanization on quality of shallow ground water in the arid to semiarid western United States, 1993–2004: U.S. Geological Survey Scientific Investigations Report 2007–5179, 56 p. Available at http://pubs.usgs.gov/sir/2007/5179/.

Plummer, L.N., and Busenberg, E., 2000, Chlorofluorocarbons *in* Cook, P.G., and Herczeg, A.L., eds., Environmental tracers in subsurface hydrology: Boston, Kluwer Academic Publishers, p. 441–478.

Reilly, T.E., Dennehy, K.F., Alley, W.M., and Cunningham, W.L., 2008, Ground-water availability in the United States: U.S. Geological Survey Circular 1323, 70 p. Available at http://pubs.usgs.gov/circ/1323/.

Solomon, D.K., Cook, P.G., and Sanford, W.E., 1998, Dissolved gases in subsurface hydrology, *in* Kendall, C., and McDonnell, J.J, eds., Isotope tracers in catchment hydrology: Amsterdam, Elsevier Science, p. 291–318.

Stonestrom, D.A., Constantz, Jim, Ferré, T.P.A., and Leake, S.A., eds., 2007, Ground-water recharge in the arid and semiarid southwestern United States: U.S. Geological Survey Professional Paper 1703, 414 p. Available at http://pubs.usgs.gov/pp/pp1703/.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed June 12, 2009, at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003a, Principal aquifers of the 48 conterminous United States, Hawaii, Puerto Rico, and the U.S. Virgin Islands. Available at http://www.nationalatlas.gov/mld/aquifrp.html.

U.S. Geological Survey, 2003b, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010, at http://www.mrlc.gov/.

U.S. Geological Survey, 2004, Estimated use of water in the United States, county-level data for 2000, accessed October 7, 2008, at http://water.usgs.gov/watuse/data/2000/.

Section 2.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in Salt Lake Valley, Utah

By Susan A. Thiros

Basin Overview

Salt Lake Valley is an alluvial basin bounded by the Wasatch Range, the Oquirrh and Traverse Mountains, and Great Salt Lake in the northern part of Utah (fig. 1). The basin is about 28 mi long and 18 mi wide (about 417 mi^2) and is open at its northern end, where both surface and ground water drain to Great Salt Lake. Altitudes range from about 4,200 ft at Great Salt Lake to about 5,200 ft at the basin-fill deposit/mountain boundary. The hydrogeologic basin that surrounds the valley extends to the crests of the surrounding mountains and covers about 740 mi^2. Salt Lake Valley is within the Basin and Range Physiographic Province of Fenneman (1931) and is characterized by generally parallel, north- to northeast-trending mountain ranges separated by broad alluvial basins that are a result of regional extension. The normal faulting and subsequent mountain uplift and deposition of basin fill began in Miocene time and is ongoing (Mabey, 1992, p. C6). Topographic relief between the Wasatch Range and Salt Lake Valley along the Wasatch Fault is as much as 7,000 ft.

The climate in Salt Lake Valley is semiarid. Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an estimated average annual precipitation of about 17 in. over the alluvial basin as a whole (McKinney and Anning, 2009). Precipitation in the mountains can exceed 50 in/yr, falling mostly as snow in the winter. Recharge to the groundwater system is dependent primarily on the spring snowmelt runoff from the mountains. Water in the major mountain-front streams is diverted for municipal and agricultural use under current conditions. Lawns and gardens in the valley require irrigation to supplement precipitation during the growing season. The demand for water peaks during July through August, when lawns and gardens require more irrigation because of the summer heat. Public water systems that use surface-water sources also use groundwater during the summer to meet the increased demand. Water systems without surface-water sources rely on water from wells throughout the year.

Salt Lake Valley generally coincides with the populated part of Salt Lake County, which contains the Salt Lake City metropolitan area. The population in Salt Lake County in 2000 was about 898,000 (U.S. Census Bureau, 2002), and is growing rapidly. The population almost doubled between 1963 and 1994, corresponding to a large increase in land developed for residential and commercial use. Population in Salt Lake County is projected to be about 1,884,000 in 2050 (Utah Governor’s Office of Planning and Budget, 2008). Analysis of LandScan population data for 2005 (LandScan Global Population Database, 2005) indicated a population of 944,000 for the alluvial basin as a whole (McKinney and Anning, 2009), equating to a population density in the valley of about 2,260 people per mi^2. Because the natural boundaries of the valley restrict much expansion of residential areas, population growth will occur mainly through increased population density and will include urbanization of the remaining agricultural and rangeland areas.

The area of agricultural land in Salt Lake Valley decreased from 145 mi^2 in 1960 to 44 mi^2 in 2002, while the area of urban land increased from 89 to 270 mi^2 during the same period (Utah Department of Natural Resources, Division of Water Resources, 1999, 2007). Many of the developed residential/commercial areas are along the mountain front bounding the east side of the valley (fig. 2) and more recent development is also replacing agricultural areas on the west side of the valley. The main crop types mapped in 2002 were grains, pasture, and alfalfa. Historically, much of the industrial land use in Salt Lake Valley was near the Jordan River, with the urban area centered in the northeastern part of the valley in Salt Lake City and agricultural land primarily near the mountain-front streams or downgradient from irrigation canals. A major industry in the valley was processing ore mined from the Wasatch Range and Oquirrh Mountains beginning in about 1870 (Calkins and others, 1943, p. 73).

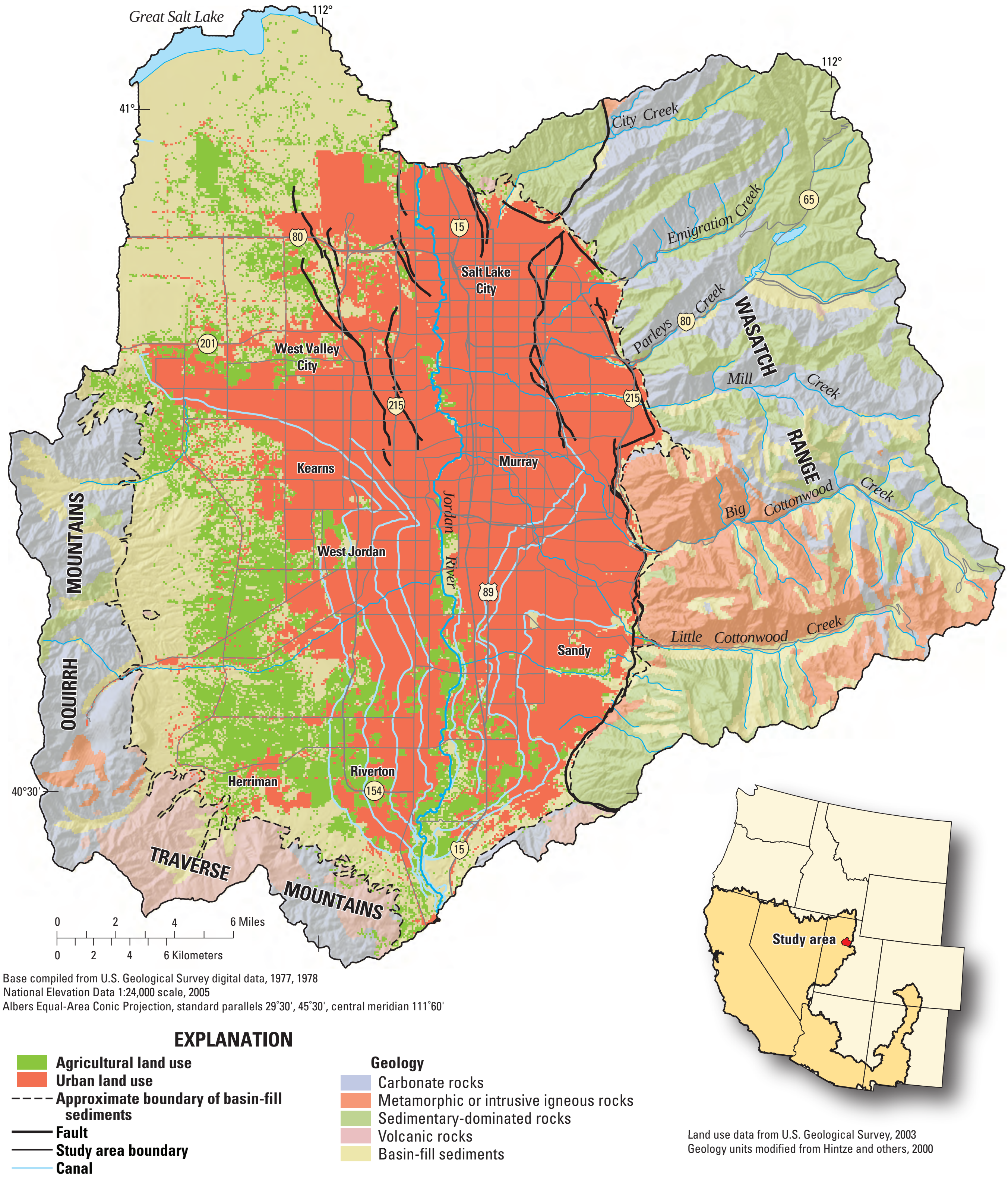


Figure 1. Physiography, land use, and generalized geology of Salt Lake Valley, Utah.

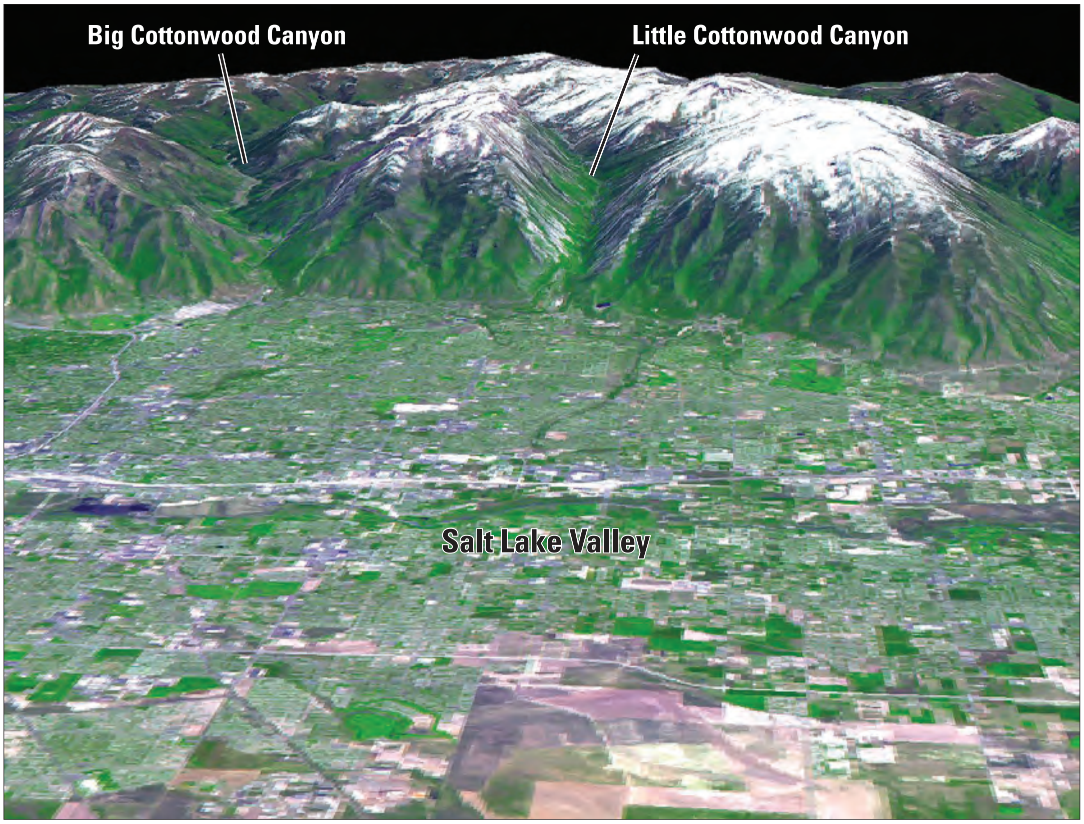


Figure 2. View of Salt Lake Valley, Utah, with Big and Little Cottonwood Canyons in the Wasatch Range in the background. Image acquired on May 28, 2000 with credit to the NASA/GSFC/METI/ERSDAC/JAROS and U.S./Japan ASTER Science Team (http://asterweb.jpl.nasa.gov/gallery-detail.asp?name=SaltLakeCity)

Changes in land use and water use in Salt Lake Valley have affected groundwater quality through changes in the sources, amount, and quality of water that recharges the basin-fill aquifer system. Human-related compounds such as volatile organic compounds (VOCs) and pesticides, and elevated concentrations of nitrate have been frequently detected in shallow ground water and to a lesser degree in the deeper basin-fill aquifer in areas of residential land use. Water that enters the aquifer in the valley (basin or valley recharge) is more susceptible to transporting man-made chemicals than is both surface flow and subsurface inflow from the adjacent mountains (mountain-front and mountain-block recharge). Seepage of excess water from irrigated crops, lawns, gardens, parks, and golf courses; and from leaking canals, water distribution pipes, sewer lines, storm drains, and retention basins are now sources of recharge to the basin-fill aquifer.

Water Development History

Salt Lake Valley was settled by Mormon pioneers beginning in July 1847 when they arrived in the valley and started building an irrigation system to distribute water from the mountain-front streams to croplands. City Creek, in what became downtown Salt Lake City, was the first stream to be diverted. By 1860, many farming communities had been established near the perennial Wasatch Range streams and the Jordan River. The 44-mi long Jordan River passes through the center of Salt Lake Valley, connecting two remnants of prehistoric Lake Bonneville: Utah Lake in Utah Valley to the south and Great Salt Lake to the north. Streamflow in the Jordan River averaged about 295,000 acre-ft/yr from 1914–1990 at the Jordan Narrows, just downstream from where the river enters the valley (Utah State Water Plan Coordinating Committee, 1997, p. 5-9).

As the population in Salt Lake City grew, a larger supply of mountain stream water was required to be transferred from agricultural use to municipal use. The farmers, however, needed a more consistent source of irrigation water through the summer months, when flow from the mountain streams diminished. Agreements were made to exchange water rights between Salt Lake City and area farmers that resulted in the diversion of Jordan River/Utah Lake water to the east side of the valley beginning in 1882. Water from the Jordan River is acceptable for irrigation, but not for potable uses because of turbidity and mineral content. Water in the Jordan River at the Jordan Narrows has higher concentrations of dissolved solids higher (1964–68 discharge-weighted average of 1,120 mg/L) than water that enters the valley from the major streams draining the Wasatch Range (1964–68 discharge-weighted averages range from 120 to 464 mg/L) due primarily to evaporation from Utah Lake. The effect of the water-rights exchanges was to spread water with higher concentrations of dissolved solids over a large part of the east side of Salt Lake Valley for irrigation and to distribute water with lower concentrations of dissolved solids from the mountain streams to residential areas along the east side of the valley rather than just along the natural stream channels.

Historically, water has been diverted from the Jordan River into a series of canals for subsequent diversion to irrigated land: four parallel canals traverse the west side of the valley and three parallel canals traverse the east side. Most of the canals were in operation by 1910. Parallel distribution systems allow for runoff from higher altitude irrigated areas to be collected and distributed by lower altitude canals. Canal companies generally start delivery of water for irrigation in May and end in October.

Surface water from local streams draining the Wasatch Range and imported from outside of the local drainage basin provided about 70 percent of the public supply in 2000 in Salt Lake Valley. This water is chlorinated and distributed for use across the valley. Under modern conditions, about 68,000 acre-ft/yr of water from local Wasatch Range streams is used for public supply, which is about 40 percent of the average streamflow rate (Utah State Water Plan Coordinating Committee, 1997, p. 9-7; table 5-4). About 75 percent (130,000 acre-ft) of the annual flow comes during the spring snowmelt runoff period from mid-April to mid-July. Most of this water ultimately discharges to the Great Salt Lake because of limited reservoir storage and treatment plant capacity. The feasibility of constructing surface reservoirs on the mountain streams is limited mainly because of environmental and safety concerns. The average annual flow for streams draining the Oquirrh Mountains on the west side of Salt Lake Valley is only about 4,400 acre-ft (Utah State Water Plan Coordinating Committee, 1997, table 5-4). Water rarely flows in these stream channels all the way to the Jordan River.

Water from the Weber and Duchesne Rivers is imported into the Utah Lake drainage basin as part of the Provo River Project and the Central Utah Project (CUP) to supplement surface-water supplies in Salt Lake and Utah Counties. The Salt Lake Aqueduct began conveying water from the Provo River drainage to Salt Lake Valley for public supply in 1951. The CUP consists of numerous diversions, dams, and conveyance systems that allow Utah to use a portion of its allotted share of Colorado River water under the Colorado River Compact. An average of about 111,000 acre-ft/yr was imported to the valley for public supply from these surface-water sources from 1997–2003 based on information provided by Isaacson (2004) and the Utah Division of Water Rights.

Richardson (1906, p. 35) speculated that the first flowing well was drilled in Salt Lake Valley in about 1878. Marine and Price (1964, p. 49) estimated that 7,700 flowing wells supplied about 35,000 acre-ft of water in 1957, mainly for domestic use. Many of the flowing wells have since been capped or abandoned and replaced by municipal water systems. Lowering the hydraulic head in the confined aquifer has caused a small decrease in the area of artesian conditions with time.

Large-yielding wells used for public supply in Salt Lake Valley were first installed in 1931 to supplement surface-water supplies. The estimated withdrawal of water from wells in the valley in 2000 was about 144,000 acre-ft: 93,800 acre-ft for public supply, 25,000 acre-ft for domestic and stock, 23,400 acre-ft for industry, and only 2,200 acre-ft for irrigation (Burden and others, 2001, table 2). Groundwater withdrawal from wells in 2000 was about 28 percent of that used for public supply. Springs and tunnels in the Wasatch Range provided about 2 percent of the water used in the valley for public supply.

Artificial recharge of some of the spring runoff water from mountain-front streams and from imported surface water to the basin-fill aquifer in the southeastern part of the valley is being done through injection wells. About 6,000 acre-ft/yr of water is planned to be injected (Utah Division of Water Rights, written commun., January 5, 2010) for use during periods of peak demand in the summer months. Potential future sources of water to supply the municipal needs of Salt Lake Valley include treated water from the Jordan River and adjacent shallow aquifer, and surface water imported from other areas outside the hydrogeologic basin, such as the Bear River near the Idaho border. Treated wastewater could be used for municipal irrigation and is another possible future water source in the valley.

Hydrogeology

The basin-fill deposits in the Salt Lake Valley consist of unconsolidated to semiconsolidated Tertiary-age deposits overlain by unconsolidated Quaternary-age deposits. The Tertiary-age sediments that crop out along the western and southern margins of the valley were deposited mainly as alluvial fans, in lakes, and as volcanic ash and are estimated to have a hydraulic conductivity of about 1 ft/d (Lambert, 1995, p. 15). On the basis of geophysical studies by Mattick (1970), the contact between these deposits and underlying consolidated rock is estimated to be as deep as 4,000 ft below land surface in areas near the Great Salt Lake and north of Salt Lake City. The permeable Tertiary-age deposits of sand and gravel yield water to wells in the Kearns area, and near Murray, Herriman, and Riverton (Hely and others, 1971, p. 107).

The unconsolidated sediments of Quaternary age were deposited mainly as alluvial fans, by streams, and as deltas and other lacustrine features associated with Lake Bonneville and older paleolakes that once covered the valley. The hydraulic conductivity of coarser grained deposits is estimated to be about 200 ft/d, compared to a value of about 1 ft/d for shallow lake-deposited clays (Lambert, 1995, p. 14). The Quaternary-age sediments are considerably more permeable than those of Tertiary age, but are thought to be less than 1,000 ft thick across most of Salt Lake Valley based on well data (Arnow and others, 1970; Lambert, 1995, fig. 4). The Quaternary-age deposits are thinnest along the margins of the valley and are less than 150 ft thick in the Kearns area. Nearly all the water wells in the valley are open to the Quaternary-age deposits. Lake-deposited clay layers occur throughout the valley, except near the mountain-front canyons, where coarser grained deposits predominate. Lake Bonneville covered much of the western half of Utah and the southeastern corner of Idaho during the late Pleistocene Epoch, with a water level about 1,000 ft above the present-day altitude of its remnant, Great Salt Lake (which is about 4,200 ft). As Lake Bonneville receded, wave-cut terraces on the lower slopes of the mountains and deposits of sand and gravel within the basin were exposed. Interlayered clay, silt, sand, and gravel were deposited as deltas in the lake by major streams as they flowed out of the mountains and are now deeply incised by modern stream channels emanating from the adjacent mountain blocks.

The consolidated rocks in the Wasatch Range bounding the northeastern part of Salt Lake Valley, from Mill Creek Canyon northward, are dominantly sedimentary Triassic-age shale and mudstone with bedding planes striking approximately perpendicular to the mountain front. The Wasatch Fault is inside of the valley west of the mountain front in this area, resulting in shallow depths to bedrock between the fault and the mountain front. This position of the fault is in contrast to that farther south, where it bounds the mountain front. The mountain block along the southeastern part of the valley consists of Precambrian-age quartzite and Tertiary-age intrusive rocks (quartz monzonite) that Hely and others (1971, plate 1) characterized as “rocks of lowest permeability.”

The Oquirrh and Traverse Mountains are made up mostly of Late Paleozoic-age carbonate and quartzite (Oquirrh Formation) and Tertiary-age volcanic rocks. Tertiary-age igneous rocks intruded the Oquirrh Formation in the Oquirrh Mountains, forming deposits of copper and other metals that have been extracted in the Bingham mining district. Consolidated volcanic rocks crop out along the base of the Oquirrh Mountains and underlie the basin-fill deposits on the west side of the valley. The transmissivity of these rocks is dependent on the presence or absence of fractures and is highly variable. Hely and others (1971, plate 1) characterized the volcanic rocks as “rocks of lowest permeability.”

Conceptual Understanding of the Groundwater System

The groundwater system in Salt Lake Valley’s basin-fill deposits includes a shallow aquifer that is separated from a deeper aquifer by discontinuous layers or lenses of fine-grained sediment. A generalized model of the deeper basin-fill aquifer shows an unconfined part near the mountain fronts that becomes confined toward the center of the valley by clay lenses and layers (fig. 3). The extent of the unconfined part of the aquifer corresponds to that of the primary recharge area in the valley (fig. 4) and includes the area near the mountain fronts where no substantial layers of fine-grained materials impede the downward movement of water. The depth to water in the unconfined part of the deeper basin-fill aquifer is typically from 150 to 500 ft below land surface. The transmissivity of the basin-fill deposits is generally highest near the mountains where streams entering the valley deposit the coarsest-grained materials.

Ground water moves laterally from the unconfined part of the basin-fill aquifer to the adjacent confined part, and from the overlying shallow aquifer to the deeper basin-fill aquifer, where the hydraulic gradient is downward and the confining layers are discontinuous. The latter conditions can exist in the secondary recharge area and were mapped by Anderson and others (1994, p. 6). The distinction between the shallow and deeper basin-fill aquifers is not clear in some parts of the valley. Many domestic wells and some public-supply wells are in the secondary recharge area, where water levels are about 100 ft below land surface.

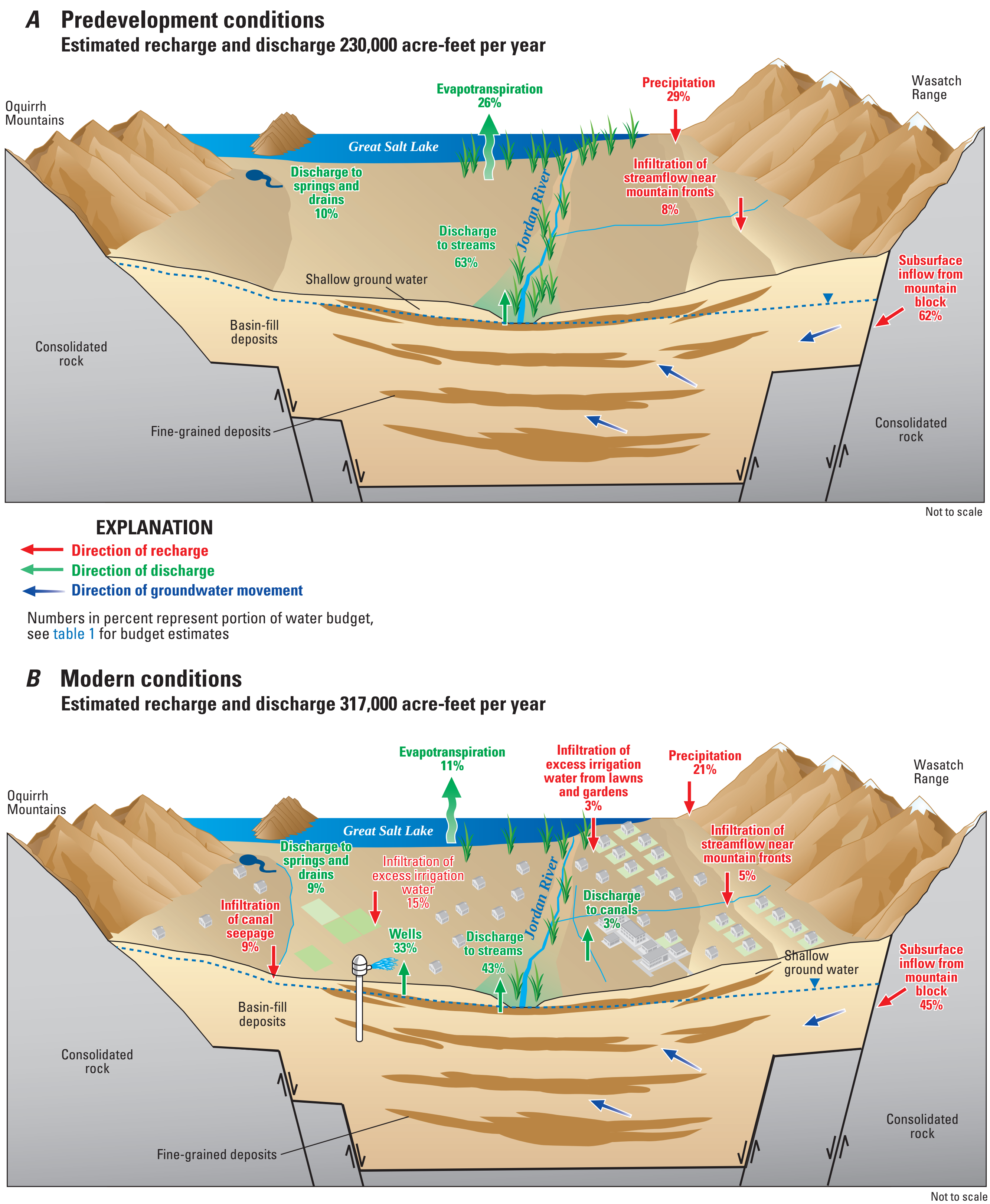


Figure 3. Generalized diagrams for Salt Lake Valley, Utah, showing the basin-fill deposits and components of the groundwater flow system under (*A*) predevelopment and (*B*) modern conditions.

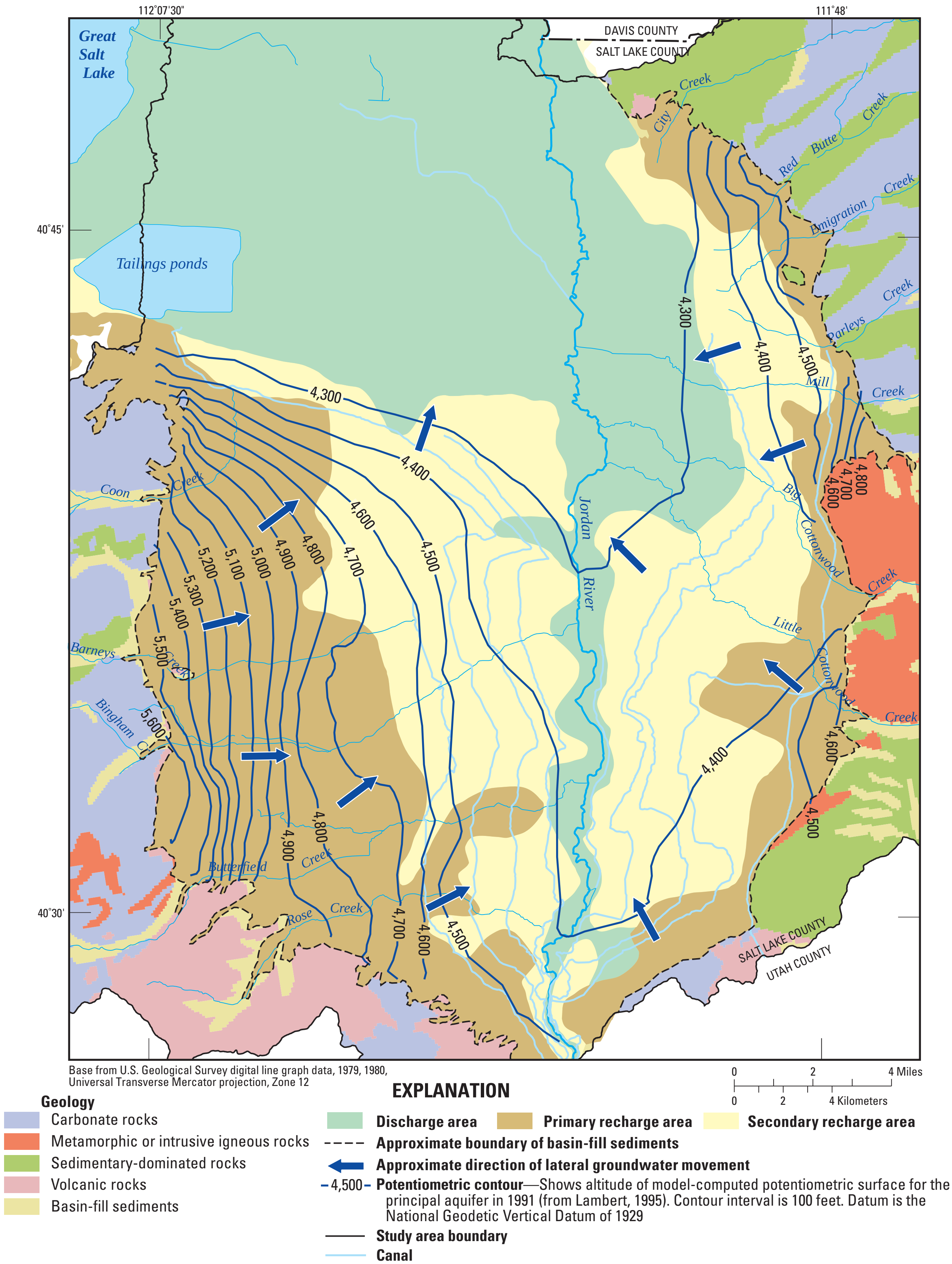


Figure 4. Location of groundwater recharge and discharge areas and approximate direction of lateral groundwater flow in Salt Lake Valley, Utah.

Groundwater discharges in areas where there is an upward hydraulic gradient from the confined part of the deeper aquifer toward the overlying shallow aquifer; such areas are generally in the center of the valley along the Jordan River and in the topographically lowest parts of the valley. This upward gradient and the presence of confining layers prevent water with relatively high concentrations of dissolved solids or other contaminants from moving downward. The confined part of the aquifer can still be susceptible to contamination where the confining layers are discontinuous or where the hydraulic gradient has been reversed (is downward), allowing water from the shallow aquifer to move downward to the confined aquifer. This reversal can result from withdrawals from wells (pumpage) over time and can permit the downward movement of water around an improperly completed well or over a larger area. Both the confined and unconfined parts of the deeper basin-fill aquifer are important sources of drinking water for Salt Lake Valley.

Shallow groundwater is either local in extent because it is perched on fine-grained materials or is laterally continuous and forms a more extensive aquifer. Perched groundwater can occur near the mountains where saturated discontinuous strata of sand and gravel are underlain by fine-grained material and lie above the regional water table. The shallow aquifer is typically present within the upper 50 ft of basin-fill deposits and therefore is vulnerable to contamination because of its proximity to human activities at the land surface. Low yields and poor quality (unacceptable for intended use) limit the use of shallow groundwater in Salt Lake Valley at the present time.

Water Budget

Recharge to and discharge from the basin-fill aquifer system in Salt Lake Valley has been estimated in studies by Hely and others (1971), Waddell and others (1987a), and Lambert (1995). Lambert used a steady-state numerical model to specify or compute an average annual recharge rate of about 317,000 acre-ft to the basin-fill groundwater system under modern conditions (table 1). Estimates of the total groundwater budget have decreased with each successive study. The amount of recharge to the groundwater system affects the amount of water that can be withdrawn from wells without affecting other types of discharge and places a greater emphasis on recharge that originates at the valley surface and therefore is vulnerable to contamination.

Although the amount of water that was recharged and discharged from the basin-fill aquifer before water development began in the valley is not known, estimates were made on the basis of the conceptual model of the system. Mountain-front recharge is estimated to have comprised about 70 percent of recharge to the basin-fill aquifer under predevelopment conditions and includes subsurface inflow from consolidated rocks in the adjacent mountains, underflow in channel fill at the mouths of canyons, and infiltration of streamflow and precipitation runoff near the mountain front (fig. 3). Information is not available to distinguish between water entering the basin-fill aquifer in the subsurface and precipitation runoff at the mountain front, but environmental tracers indicate that subsurface inflow from the mountain blocks may be a substantial component of recharge. Inflow from consolidated rock along the mountain front and from precipitation on the valley floor was specified at 142,000 and 67,000 acre-ft/yr, respectively, in the steady-state simulation by Lambert (1995, table 5), and these rates are assumed to be representative of predevelopment conditions.

Infiltration of excess irrigation water from croplands, lawns, and gardens, and seepage from canals became major sources of recharge to the groundwater system (about 27 percent of estimated average annual recharge) under modern conditions (Lambert, 1995, table 5; table 1). Groundwater discharge to the Jordan River and other streams (about 43 percent of estimated average annual discharge), withdrawals from wells (about 33 percent), and evapotranspiration (about 11 percent) are the main components of discharge under modern conditions. Groundwater discharge to the Jordan River and its tributaries and by evapotranspiration has been reduced from that under predevelopment conditions as a result of lowered groundwater levels caused by withdrawals from wells (table 1).

Recharge to the basin-fill aquifer as subsurface inflow from the mountain block on the east side of Salt Lake Valley is greater than that to the west side, primarily because the west face of the Wasatch Range receives greater amounts of precipitation than does the east side of the Oquirrh Mountains. Infiltration of precipitation in the primary recharge areas of the valley has likely decreased with time because of urban development and the installation of storm drains. Excess irrigation water applied to lawns and gardens is now a major source of infiltration to the basin-fill aquifer in the recharge areas, and much of this water is imported from outside the drainage basin. Losses from major canals diverting water from the Jordan River were estimated to be about 21,000 acre-ft/yr in the southwestern part of the valley (Lambert, 1996, p. 8) out of about 30,000 acre-ft/yr estimated valley wide (Lambert, 1995, table 5). Seepage losses from canals can recharge both the shallow and deeper parts of the basin-fill aquifer because the canals flow mainly through secondary recharge areas. Groundwater recharge has increased by almost one-third from that of predevelopment conditions, primarily due to the addition of canal seepage and excess irrigation water (table 1).

The recharge of excess irrigation water and canal losses has greatly modified the groundwater flow system in the southwestern part of Salt Lake Valley, where there was relatively little recharge prior to irrigation. Canals in this area transport water primarily from the Jordan River, resulting in water with higher concentrations of dissolved solids being recharged to the basin-fill aquifer. Stable isotope data indicate that the shallow unconfined aquifer (Thiros, 1995, p. 51; Thiros, 2003, p. 35) and parts of the deeper basin-fill aquifer (Thiros and Manning, 2004, p. 36) receive substantial recharge from water diverted for irrigation from the Jordan River.

Table 1. Estimated groundwater budget for the basin-fill aquifer in Salt Lake Valley, Utah, under predevelopment and modern conditions.

[All values are in acre-feet per year and are rounded to the nearest thousand. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from Hely and others (1971); a steady-state numerical simulation of the basin-fill aquifer (Lambert, 1995); or were estimated as described in the footnotes. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Subsurface inflow from mountain blocks	[1]142,000	[1]142,000	0
Infiltration of precipitation on valley floor	[1]67,000	[1]67,000	0
Infiltration of streamflow and underflow in channel fill near mountain fronts	[2]18,000	[1]16,000	[3]-2,000
Underflow at Jordan Narrows	[1]2,000	[1]2,000	0
Infiltration of streamflow in valley	[1]1,000	[1]1,000	0
Canal seepage	0	[1]30,000	30,000
Infiltration of excess irrigation water	0	[1]47,000	47,000
Infiltration of excess irrigation water from lawns and gardens	0	[1]10,000	10,000
Infiltration from reservoirs	0	[1]2,000	2,000
Total recharge	**[4]230,000**	**317,000**	**87,000**
Budget component		**Estimated discharge**	
Discharge to streams	[5]145,000	[1]137,000	-8,000
Well withdrawals	0	[1]105,000	105,000
Evapotranspiration	[6]60,000	[1]36,000	-24,000
Discharge to springs	[1]19,000	[1]19,000	0
Discharge to drains	[7]5,000	[1]10,000	5,000
Subsurface outflow to Great Salt Lake	[1]1,000	[1]1,000	0
Discharge to canals	0	[1]9,000	9,000
Total discharge	**230,000**	**317,000**	**87,000**
Change in storage (total recharge minus total discharge)	0	0	0

[1] Estimates from steady-state numerical simulation of the basin-fill aquifer described by Lambert (1995).

[2] Hely and others (1971, p. 56) evaluated the relation of channel loss in Wasatch Range streams to runoff during 1964–68. They noted that the magnitude of losses changed with fluctuations in runoff and generally ranged from 8 to 16 percent of runoff. Recharge from streams and underflow in channel fill near the mountain fronts under predevelopment conditions was estimated to be 10 percent of an average streamflow of about 178,000 acre-feet per year for 1940–80 (Utah State Water Plan Coordinating Committee, 1997, p. 5-4).

[3] The change from predevelopment to modern conditions may be the result of the different methods used to estimate the component rather than an actual change over time.

[4] Hely and others (1971, p. 143) estimated that natural recharge was about 234,000 acre-feet per year.

[5] Hely and others (1971, p. 84) estimated average annual groundwater discharge to the Jordan River from 1943–68 to be 154,000 acre-feet. About 147,000 acre-feet per year of the gross gain in river flow during this period is assumed to be from the confined part of the deeper aquifer because it is unaffected by seasonal changes (Hely and others, 1971, p. 136). The estimate used for groundwater discharge to the Jordan River under predevelopment conditions in this table is the residual amount needed to balance the other recharge and discharge components.

[6] Hely and others (1971, p. 179) estimated evapotranspiration from areas of natural and cultivated vegetation and from bare ground in 1964–68 at about 60,000 acre-feet per year. It is assumed here that natural vegetation would have grown in cultivated areas and discharged a similar amount of groundwater under predevelopment conditions.

[7] Hely and others (1971, p. 179) estimated groundwater discharge from the shallow part of the aquifer to drains in the northwestern part of the valley from measurements of low flows during water years 1964–68. It is assumed here that this shallow groundwater discharged under predevelopment conditions also.

This water is isotopically heavier because of evaporation. Richardson (1906, p. 41) reported that groundwater levels in the area downgradient from the Utah and Salt Lake Canal (completed in 1882) on the west side of Salt Lake Valley had risen as a result of canal seepage. Several wells in the area were reported to have water levels 30-65 ft nearer to the land surface then before the construction of the canal. Richardson stated that "… the quality of groundwater in the area has deteriorated in recent years, containing now much more alkali than formerly. So marked has this change been that surface wells are but little valued, and generally water for domestic use is obtained from deep wells." Recharge from excess irrigation water and canal seepage also affected water levels in the discharge area south-southeast of Salt Lake City that is traversed by Parleys, Mill, Big Cottonwood, and Little Cottonwood Creeks. Taylor and Leggette (1949, p. 23) noted local reports of increasing flow from artesian wells nearest to the recharge area soon after irrigation on higher altitude lands began.

Under present-day conditions, the groundwater system in Salt Lake Valley is greatly affected by withdrawals from wells, which has ranged from about 38,000 acre-ft in 1938 to 165,000 acre-ft in 1988. Withdrawals from wells are about one-third of the total estimated discharge from the modern groundwater system (table 1). Most of the pumping occurs on the east side of the valley because of higher yields and lower concentrations of dissolved solids. In some areas of the valley, groundwater is blended with water from other sources to improve its quality.

In 2000, utilized water rights and approved applications for rights show approximately 400,000 acre-ft/yr of potential groundwater withdrawal from the deeper basin-fill aquifer compared to an estimated "safe yield" of 165,000 acre-ft/yr (Robert Morgan, Utah Division of Water Rights, written commun., May 17, 2000, http://nrwrt1.nr.state.ut.us/meetinfo/m051700/slvplan.pdf). As a result, the Utah Division of Water Rights has implemented a groundwater management plan for the valley that provides guidelines on withdrawal limits in order to protect existing water rights and water quality.

Groundwater Movement

The potentiometric surface for the basin-fill aquifer indicates that groundwater generally moves from recharge areas near the mountain fronts toward the Jordan River and Great Salt Lake (fig. 4). Groundwater moves downward in the primary and secondary recharge areas from the land surface to the shallow unconfined aquifer (where it exists) and then to the deeper basin-fill aquifer. Groundwater moves upward in the discharge area through the confined aquifer, into and through overlying confining layers, and into the shallow unconfined aquifer, where it can discharge to the Jordan River, to drains, or by evapotranspiration or seepage to Great Salt Lake, which is minor. The steeper slope of the potentiometric surface on the west side of the valley indicates less recharge and lower transmissivities due to thinner saturated deposits or less permeable material when compared to the less steep surface on the east side. Faults within and bounding the basin-fill deposits may affect the hydraulic gradient and groundwater movement, and water from wells near faults in the northwestern part of the valley generally is warmer than water more distant from faults, indicating movement from greater depths. Most measured water levels in the deepest parts of the basin-fill aquifer have declined from spring 1975 to spring 2005 (Burden and others, 2005, fig. 14), with the largest decline of about 53 ft in a well in the southeastern part of the valley. This is an area with large withdrawals for public supply because of high yields and good water quality from the wells.

An approximate recharge rate was derived for the southeastern part of Salt Lake Valley from the mouth of Mill Creek Canyon southward to about 2 mi south of the mouth of Little Cottonwood Canyon. The typical age gradient of about 7.5 years/mi (along the groundwater flow path) in this area corresponds to an average linear groundwater flow velocity of 1.9 ft/d (Thiros and Manning, 2004, p. 54). Assuming a porosity of 0.2 (20 percent), an average saturated thickness of 330 ft (generally ranges from 150 to 500 ft), and a north-south cross-section length of 10 mi, the approximate recharge rate for the southeastern part of the valley is about 55,000 acre-ft/yr. Results of age dating using chlorofluorocarbons indicate an average groundwater flow velocity of between 1.4 and 1.8 ft/d in the southwestern part of the valley (Kennecott Utah Copper, 1998, p. 3-18).

Apparent tritium/helium-3 ages determined for water from 64 public-supply wells completed in the basin-fill aquifer in Salt Lake Valley range from 3 years to more than 50 years (Thiros and Manning, 2004, fig. 22) (fig. 5). See Section 1 of this report for a discussion of groundwater age and environmental tracers. Because public-supply wells generally have long open (screened or perforated) intervals (typically 150-500 ft), the samples likely contain mixtures of water with different ages. Water recharged before large amounts of tritium were introduced into the atmosphere by nuclear testing in the early 1950s is considered to be pre-bomb water. Interpreted-age categories were determined from the initial tritium concentration for each sample (measured tritium plus measured tritiogenic helium-3) and its relation to that of local precipitation at the apparent time of recharge (Thiros and Manning, 2004, fig. 21). Water sampled from the public-supply wells was divided into dominantly pre-bomb, modern or a mixture of pre-bomb and modern, or dominantly modern interpreted-age categories.

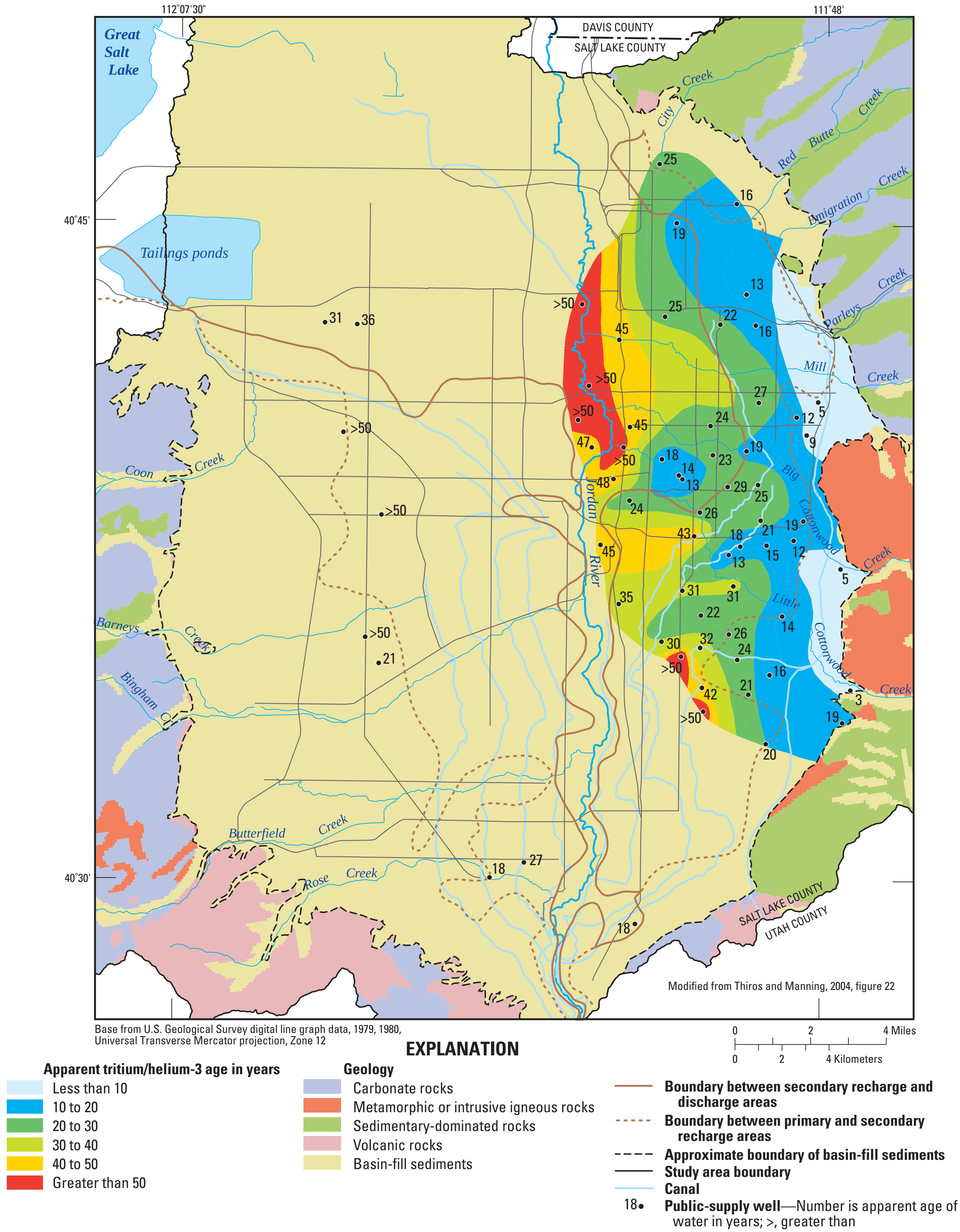


Figure 5. Distribution of apparent tritium/helium-3 ages for water sampled from the deeper basin-fill aquifer in Salt Lake Valley, Utah, 2000–01.

Tritium concentrations in water sampled from the shallow part of the basin-fill aquifer in secondary recharge areas within Salt Lake Valley indicate that most or all of the water was recently recharged from the land surface with little or no mixing with older groundwater. The apparent tritium/helium-3 age for water sampled from 24 monitoring wells ranged from 1 year or less to 38 years (Thiros, 2003, table 5). Water from most of the monitoring wells was contaminated with chlorofluorocarbons, which also indicates that the water has been in contact with human-derived compounds at the land surface.

Ages of groundwater in the primary and secondary recharge areas are generally younger on the east side of the valley than on the west side (Thiros and Manning, 2004, fig. 24), indicating that recharge rates are generally greater on the east side. Groundwater on the east side of the valley generally becomes older with distance from the mountain front, the oldest water being that in the discharge area. On the west side of Salt Lake Valley, the median apparent age of water from wells in the secondary recharge and discharge areas is younger than that of water from wells in the primary recharge area. This age difference is probably affected by the primary recharge area on the west side of the valley being upgradient from two major components of recharge in the area under modern conditions: losses from canals and infiltration from irrigated fields.

Effects of Natural and Human Factors on Groundwater Quality

The occurrence and concentrations of contaminants in water within the basin-fill aquifer system in Salt Lake Valley are influenced by the locations and sources of recharge, the vertical hydraulic gradient, and aquifer properties. Water that enters the basin-fill aquifer in the valley (valley recharge) is more susceptible to transporting man-made chemicals than is subsurface inflow from the adjacent mountains (mountain-block recharge) and surface flow at the mountain front and in major mountain streams. Seepage of excess water from irrigated crops, lawns, gardens, parks, and golf courses; and from leaking canals, water distribution pipes, sewer lines, storm drains, and retention basins are modern-day sources of groundwater recharge in many parts of the valley.

Data were collected as part of three National Water-Quality Assessment (NAWQA) Program studies in Salt Lake Valley to characterize and determine the effects of natural and human factors on groundwater quality. A study to evaluate the occurrence and distribution of natural and human-related chemical constituents and organic compounds in shallow groundwater underlying recently developed (post-1963) residential and commercial areas in the valley was done in 1999 (Thiros, 2003). Thirty monitoring wells were installed and sampled in areas where there was a downward gradient between the shallow and deeper aquifers. Although the aquifers are separated by layers of fine-grained deposits, there is potential for water in these wells to move deeper to parts of the basin-fill aquifer used for public supply. The occurrence and distribution of natural and human-related compounds in groundwater used for drinking and public supply in Salt Lake Valley were evaluated by analyzing water-quality data collected from 31 public-supply wells in 2001 (Thiros and Manning, 2004). An additional 19 wells completed in the primary and secondary recharge areas, mostly used for domestic and public supply, also were sampled to characterize water quality in the deeper basin-fill aquifer in the valley.

General Water-Quality Characteristics and Natural Factors

The inorganic chemical composition of groundwater largely depends on its recharge source, the type of rocks and associated minerals it has contacted, and how long the water has been in contact with the aquifer material. Generally, the most mineralized groundwater is in the northwestern part of the valley near the Great Salt Lake. This area is at the downgradient end of the overall Salt Lake Valley groundwater flow path, and on the basis of stable isotope data (Thiros, 1995, p. 51), the water is possibly thousands of years old. Stable isotope data also indicate that evaporation is not a factor contributing to mineralization of the deeper aquifer; sulfate-reducing conditions and the presence of sodium and chloride ions in pore water left from the desiccation of paleolakes contribute to chemical processes that result in a sodium-chloride-type groundwater. Dissolved-solids concentrations in groundwater from this part of the valley are generally greater than 1,000 mg/L (fig. 6).

Groundwater in the northeastern part of the valley generally has more dissolved sulfate relative to bicarbonate than water in upgradient areas and from local mountain-front streams. Dissolved-solids concentrations there are greater than 500 mg/L (fig. 6), primarily as a result of the contact of the water with easily eroded Triassic-age shale and mudstone in the mountain block and in the basin-fill deposits in the area.

Basin-fill deposits in the southeastern part of the valley are derived from rocks such as quartzite and quartz monzonite, which are more resistant to weathering and include less easily soluble material than the rocks further north. The groundwater in this area is predominantly a calcium-bicarbonate type, similar to that of water in local mountain-front streams, and concentrations of dissolved solids are generally less than 500 mg/L (fig. 6). A relatively large area of groundwater with concentrations of dissolved solids less than 250 mg/L extends northwestward from the mountain front toward the Jordan River following regional flow paths. Age-dating of this water indicates that it moves rapidly through coarse-grained deposits near the mountain front (Thiros and Manning, 2004, fig. 23).

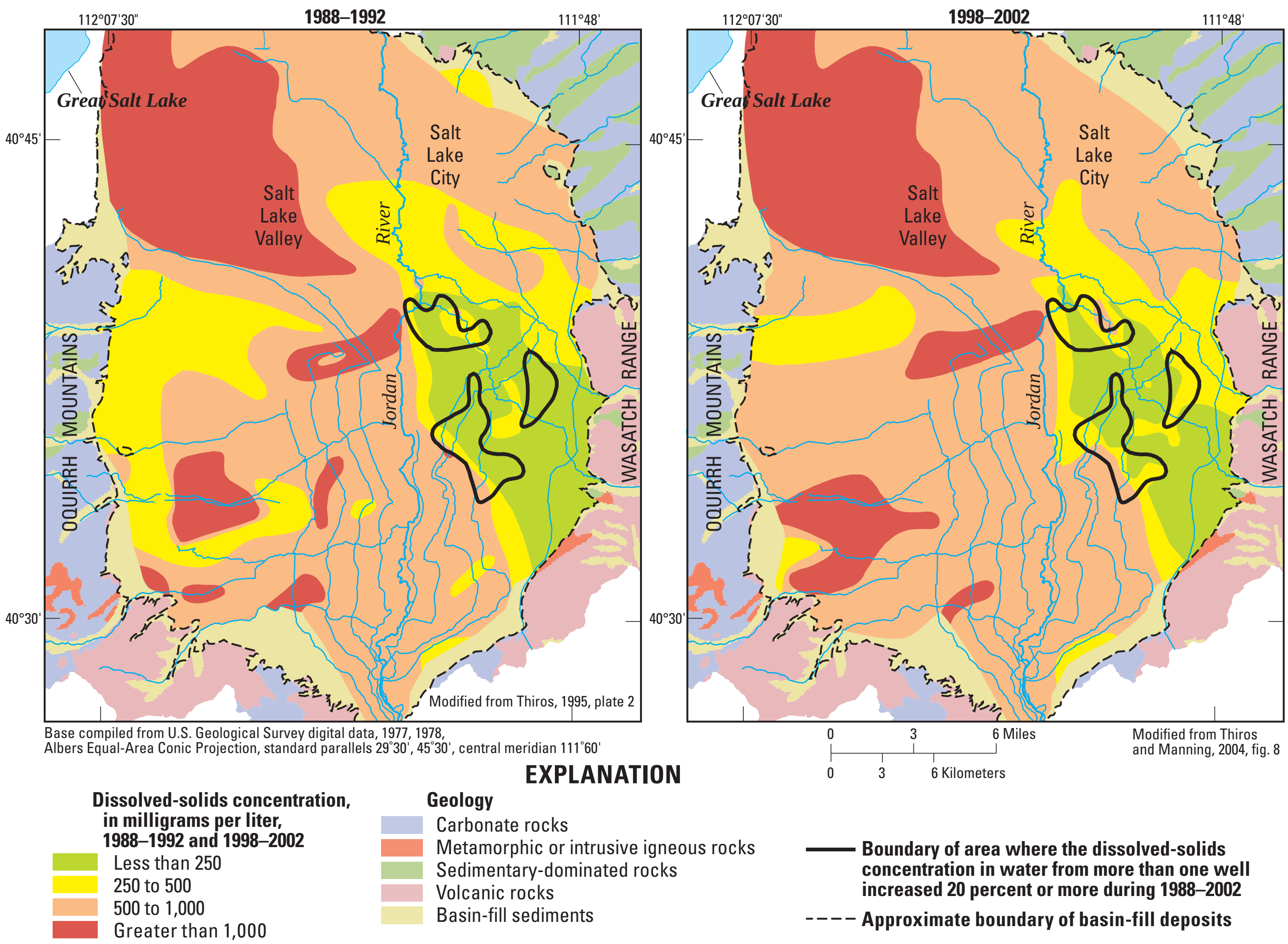


Figure 6. Dissolved-solids concentration in water sampled from parts of the deeper basin-fill aquifer in Salt Lake Valley, Utah, in 1988–92 and 1998–2002.

Groundwater quality in the southwestern part of Salt Lake Valley, Utah, is influenced by reactions between the basin-fill deposits derived from rocks of the Oquirrh Mountains and the different types of water recharged in the area. The Oquirrh Mountains are composed primarily of carbonate rocks that locally have undergone sulfide mineralization. Prior to development in the valley, the main source of recharge to the basin-fill aquifer was subsurface inflow from the mountain block along with seepage from the mountain-front streams and infiltration of precipitation on the valley floor. Geochemical reactions between the basin-fill deposits and the naturally recharged water probably resulted in groundwater with dissolved-solids concentrations less than 1,000 mg/L. Under modern conditions, canal seepage and infiltration of excess irrigation water have contributed to higher concentrations of dissolved solids (greater than 1,000 mg/L) in some areas in this part of the valley (fig. 6). Infiltration of mine drainage and wastewater (most seepage from mining related sources was stopped in 1992) has resulted in an area with high concentrations of sulfate in groundwater downgradient from the Bingham Canyon mining operations (Waddell and others, 1987b, p. 16).

Concentrations of dissolved oxygen in groundwater sampled as part of the NAWQA studies ranged from 0.3 to 11.6 mg/L, and pH ranged from 6.8 to 8.0 standard units. Dissolved-oxygen concentrations in pre-bomb era water from the deeper part of the aquifer in the discharge area indicate reducing conditions; otherwise, groundwater in the valley is generally oxic (contains dissolved oxygen).

Concentrations of dissolved arsenic in groundwater sampled as part of the NAWQA studies ranged from 0.4 to 23 µg/L, with a median value of 2.0 µg/L, in the deeper part of the basin-fill aquifer, and from less than 1.0 to 19.6 µg/L, with a median of 7.3 µg/L, in the shallower part. The drinking-water standard for arsenic is 10 µg/L (U.S. Environmental Protection Agency, 2008). Arsenic concentrations in water from wells in most of the western part of the valley generally were higher than in groundwater from other areas (Thiros and Manning, 2004, fig. 9). Human-related factors in addition to natural factors may be affecting arsenic concentrations in this area. More arsenic-bearing minerals associated with the sulfide-mineralized zone in the Oquirrh Mountains may be present in the fine-grained basin-fill deposits coupled with less recharge available to transport arsenic through the system. Groundwater sampled from near the water table that contained arsenic at concentrations greater than 10 µg/L may be affected by dissolved organic carbon and oxygen present in recharge water from excess irrigation and canal losses. This source of recharge may have mobilized arsenic from the aquifer material through the dissolution of pyrite or by desorption from iron oxides bound to the basin-fill sediments in the western part of the valley. The proximity of faults, and the potential for geothermal water from deep sources to move into the basin-fill deposits also is a potential factor in the elevated concentrations of arsenic in groundwater in some areas.

Concentrations of dissolved uranium in groundwater sampled as part of the NAWQA studies ranged from 0.04 to 15.1 µg/L in the deeper part of the basin-fill aquifer and from less than 1.0 to 93 µg/L in the shallower part, with a composite median value of 4.9 µg/L. The drinking-water standard for uranium is 30 µg/L (U.S. Environmental Protection Agency, 2008). Uranium is soluble under oxic conditions and is concentrated in the sediment in reducing environments as a result of mineral precipitation. The highest concentrations of dissolved uranium were measured in water from wells in the southeastern part of the valley and may result from proximity to uranium-rich intrusive rocks in the Wasatch Range coupled with oxic conditions. Uranium ore processed from 1951 to 1964 at a site in the central part of the valley and its mill tailings were a source of contamination to the basin-fill aquifer (Waddell and others, 1987b, p. 29). Withdrawals from wells in the area are small, so that the naturally upward hydraulic gradient is not affected. A reversal in the gradient could allow contaminated shallow water to move downward to the deeper confined part of the aquifer.

Potential Effects of Human Factors

Agricultural and urban development in the Salt Lake Valley has brought additional sources and processes of recharge to and discharge from the basin-fill aquifer system, which together have acted to accelerate the movement of water from the land surface to parts of the system. This results in the aquifer being more susceptible to activities at the land surface and more vulnerable to contaminants if their sources are present in the valley.

Comparison of analyses of groundwater from the deeper basin-fill aquifer in the valley sampled during 1988–92 and again during 1998–2002 shows a reduction (during the latter period) in the extent of the area with dissolved-solids concentrations of less than 500 mg/L (fig. 6). Dissolved-solids concentrations increased more than 20 percent in some areas near the Jordan River and on the east side of the valley between the two periods (Thiros and Manning, 2004, p. 22). Withdrawals from wells may have caused the vertical and(or) lateral groundwater flow gradients to change, which could allow water with higher concentrations of dissolved solids from the shallow aquifer or from other parts of the deeper aquifer, both from the west and from greater depths, to reach the wells in these areas.

A long-term trend of increasing concentration of dissolved solids, mainly in the form of chloride, approximately corresponds with rising water levels through time at a flowing well in the northeastern part of the valley (fig. 7). Most valley wells show a declining water-level trend over time (Burden and others, 2005, fig. 10) that is related to groundwater pumping. Although in a discharge area, this well is near urbanized recharge areas. New sources of water and contaminants used in the recharge area likely have moved downgradient along the groundwater flow path to this well on the basis of the occurrence of human-related compounds in water from the well and a modern tritium/helium-3 determined age. Waddell and others (1987b, p. 11) suggested that a possible cause for the increase in chloride is the storage and use of road salt in recharge areas along the east side of the valley.

Although nitrate can occur naturally in groundwater, concentrations greater than an estimated background level of about 2 mg/L are generally thought to be related to human activities (Thiros and Manning, 2004, p. 24). Nitrate (as nitrogen) concentrations in water sampled from 26 of the 30 monitoring wells (87 percent) completed in the shallow aquifer in residential/commercial land-use areas were greater than 2 mg/L, indicating a likely human influence. Concentrations ranged from less than 0.05 to 13.3 mg/L with a median value of 6.85 mg/L. The drinking-water standard for nitrate is 10 mg/L (U.S. Environmental Protection Agency, 2008). Nitrate (as nitrogen) concentrations in water from 12 of the 31 public-supply wells sampled for the drinking-water study (39 percent) also were greater than 2 mg/L. The source of nitrate at concentrations above the background level may be the application of fertilizers, other agricultural activities, and leaking or improperly functioning septic systems and sewer pipes in the valley.

Pesticides and (or) VOCs were detected, mostly at very low concentrations, in water from 23 of the 31 public-supply wells sampled for the drinking-water study (Thiros and Manning, 2004). Produced and used exclusively by humans, pesticides and VOCs are known as human-related compounds. Although the measured concentrations of these compounds are not a health concern, their widespread occurrence indicates the presence of water young enough to be affected by human activity in much of the deeper basin-fill aquifer in Salt Lake Valley. Detection of these compounds in water from a well indicates the possibility that water with higher concentrations may enter the well in the future.

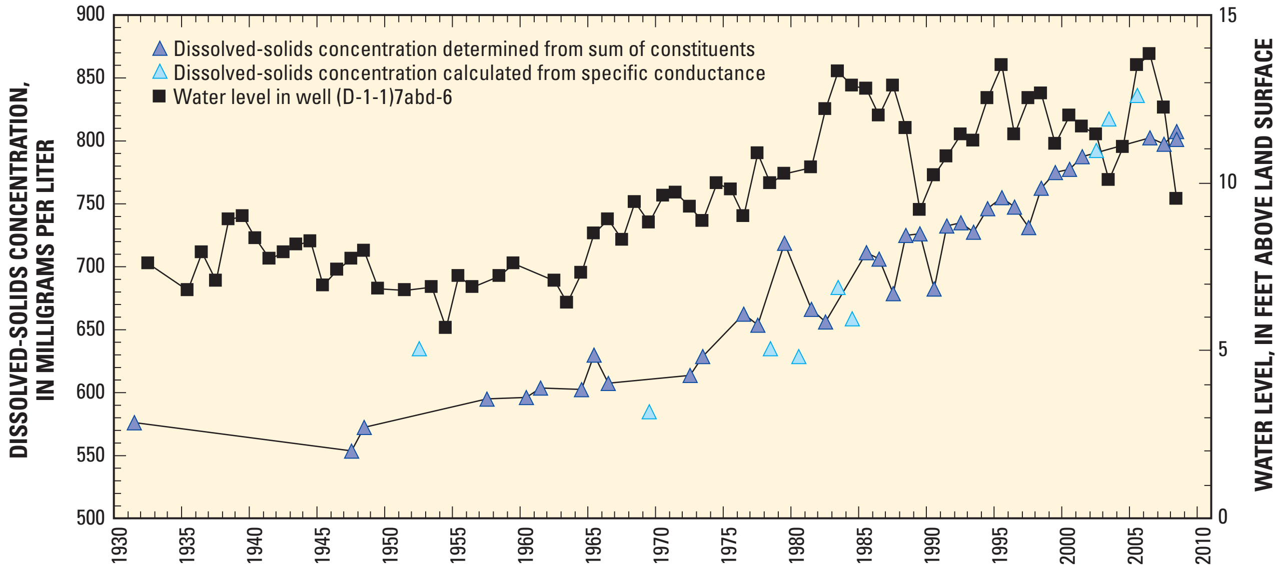


Figure 7. Relation of dissolved-solids concentration to water levels in a flowing well in the northeastern part of Salt Lake Valley, Utah.

At least one pesticide or pesticide degradation product was detected in water from 28 of the 30 monitoring wells completed in the shallow aquifer in residential/commercial land-use areas. The herbicide atrazine and its degradation product deethylatrazine were the most frequently detected pesticides in the NAWQA land-use and drinking-water studies (Thiros, 2003, p. 26, and Thiros and Manning, 2004, p. 27), detected in samples from 23 and 21 of the 30 monitoring wells, respectively, and in 7 and 10 of the 31 public-supply wells, respectively. Atrazine is a restricted-use pesticide that is used primarily on corn and along roads, railroads, other right-of-ways, utility substations, and industrial lots to control weeds and undesired vegetation. It is not intended for household use. The high detection frequency of atrazine in shallow groundwater in residential areas on the west side of the valley may be the result of its application in formerly agricultural or industrial areas that have been converted to residential uses, or the herbicide was applied to agricultural or industrial land upgradient from the residential areas and was transported to these areas in groundwater.

Eleven of the 85 VOCs for which water samples collected for the drinking-water study were analyzed were detected in one or more of the samples. The most frequently detected VOCs were chloroform (54.8 percent of the samples), bromodichloromethane (35.5 percent), and 1,1,1-trichloroethane (19.4 percent). These compounds, along with tetrachloroethylene (PCE, a solvent), also were the most frequently detected VOCs in shallow groundwater in the valley. Chloroform and bromodichloromethane are byproducts of chlorinated groundwater and surface water that has reacted with organic material in the water and aquifer material. Widespread occurrence of these compounds in both shallow and deeper basin-fill aquifers is likely a result of recharge of chlorinated public-supply water used to irrigate lawns and gardens in residential areas of Salt Lake Valley.

Leaking underground gasoline storage tanks commonly are a source of shallow groundwater contamination from the VOCs benzene, toluene, ethylbenzene, and xylene (BTEX compounds). These gasoline-derived compounds typically were not detected in water samples from the shallow aquifer monitoring wells or the public-supply wells in the valley. Natural attenuation enhanced by oxygen-rich (oxic) conditions likely removes most of the BTEX compounds before they reach the deeper aquifer.

Drinking-water study wells in which low levels of VOCs (mainly chloroform) and pesticides (mainly atrazine and (or) its degradation products) were measured at concentrations greater than laboratory or minimum reporting levels (LRLs or MRLs) are shown in figure 8. Also shown are wells that contain water with nitrate concentrations greater than an estimated background level of 2 mg/L. Wells with water that contain human-related compounds above reporting levels and (or) nitrate concentrations above 2 mg/L are referred to as "affected wells." Wells that meet these criteria thus have a reasonably high level of susceptibility to receive water that has been affected by human activities. Eighteen of the 31 public-supply wells (58 percent) sampled for the drinking-water study are considered affected wells.

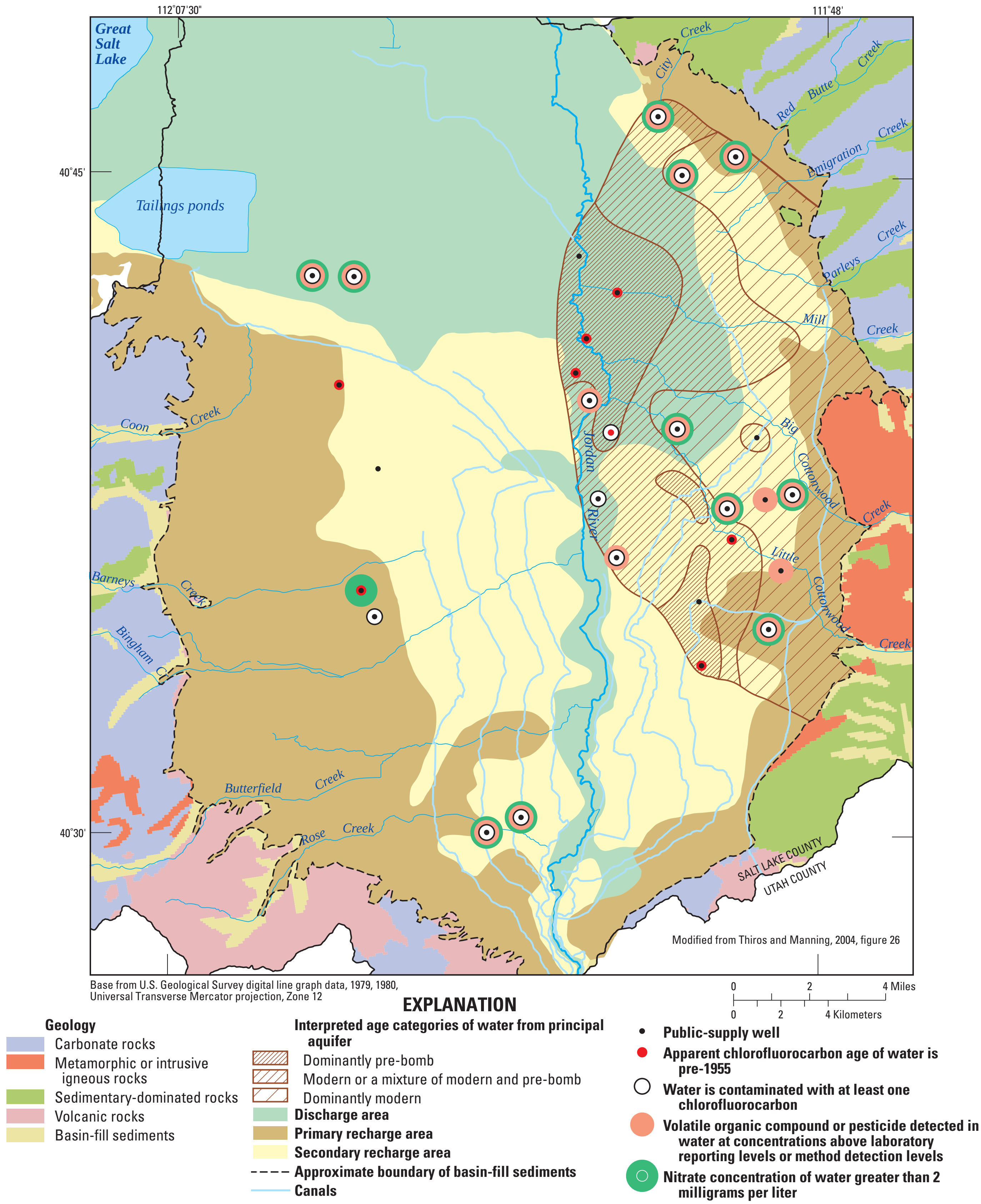


Figure 8. Interpreted-age category, chlorofluorocarbon, human-related compounds, and nitrate information for water sampled from 31 public-supply wells in Salt Lake Valley, Utah, 2001.

The presence of human-related compounds and elevated concentrations of nitrate in the deeper basin-fill aquifer is strongly correlated with the distribution of interpreted-age categories (fig. 8). Nearly all of the affected wells (17 of 18) have either dominantly modern water (generally water less than 20 years old) or a mixture of modern and pre-bomb era waters (Thiros and Manning, 2004, p. 63). Most of the unaffected wells (10 of 13) contain dominantly pre-bomb era water and thus contain little modern water. All of the wells (10 of 10) with dominantly modern water were affected while only 1 of the 11 wells with dominantly pre-bomb era water was affected. These results indicate that most of the modern groundwater in Salt Lake Valley contains human-related compounds at concentrations above reporting levels and (or) has nitrate concentrations greater than the estimated background level of 2 mg/L, and that pre-bomb era water generally is free of these human effects.

The relation between chloroform and atrazine and prometon in water from the shallow aquifer monitoring wells, although not statistically significant, was opposite for the two herbicides. The three highest concentrations of chloroform detected corresponded to three of the four highest concentrations of prometon, likely because of the presence of both of these compounds in residential areas (Thiros, 2003, p. 42). Prometon is registered for use by homeowners to control vegetation. Relatively low concentrations of chloroform corresponded to the four highest concentrations of atrazine and its degradation products; this may be a result of atrazine use on agricultural or nonirrigated industrial and vacant land.

The number of human-related compounds detected in water sampled from the drinking-water study public-supply wells is inversely correlated with the apparent tritium/helium-3 age. This dataset includes concentrations that are considered estimates because they are less than the reporting limit for the analytical method and therefore have a greater relative uncertainty, but have met the identification critera for the compound. Human-related compounds were not detected in water with an apparent age older than 50 years, with one exception. Concentrations of nitrate in water from the 31 sampled public-supply wells is correlated with many factors. Generally, nitrate concentration in water from the sampled wells increased as the depth to the top of the well's open interval became shallower; as the delta oxygen-18 ratio became heavier (more evaporated); as the apparent age of the water became younger; and as the number of human-related compounds detected in water per well increased (Thiros and Manning, 2004, p. 65). On the basis of these correlations, the concentration of nitrate in water from many of the public-supply wells is related to the occurrence of modern valley recharge, which has the potential of being influenced by human activity.

Water-quality data for 80 wells sampled in Salt Lake Valley as part of the NAWQA studies were separated into 8 classes of wells and compared to hydrogeology, water use, and land use (table 2). The well classes represent major components of the conceptual groundwater flow system: the shallow aquifer in the secondary recharge area divided into east and west sides of the valley, the deeper aquifer in the primary and secondary recharge areas divided into east and west sides of the valley, and the deeper aquifer in the discharge area divided into pre-bomb era and modern or mixed-age groundwater.

Groundwater sampled from the shallow basin-fill aquifer on the east side of the valley (class A) is recharged mainly by seepage from mountain-front streams, from canals originating at the Jordan River, and from the infiltration of imported surface water and pumped groundwater used for public supply. The major source of recharge to the shallow aquifer on the west side of the valley (class B) is seepage from canals and fields irrigated with water from the Jordan River. Water from class A (east side) wells had lower median concentrations of dissolved solids, nitrate, and arsenic than did water from the class B (west side) wells. Although most of the class B wells are in residential areas, the detection of agricultural or industrial use pesticides in all of the wells likely indicates groundwater movement from upgradient areas.

Water samples from wells in the unconfined aquifer in primary recharge areas generally had modern or a mixture of modern and pre-bomb era ages, and VOCs were detected in samples from many of the wells. The greatest median depth to water was in wells in the primary recharge area on the east side of the valley (class C), but the surrounding land use is mostly urban, and the groundwater is dominantly modern. VOCs were detected in water from all five wells sampled in this class. Pesticides or VOCs were detected at a higher frequency and median concentrations of nitrate were higher in Class C wells than in wells in the primary recharge area on the west side of the valley(class D), which includes undeveloped range, agricultural, and urban land. Nine of the 10 class D wells are west (upgradient) of any irrigation canal and therefore are not subject to recharge derived from that source. The thicker unsaturated zones in the primary recharge areas (where class C and D wells are located) lessen the susceptibility of the aquifer to the movement of contaminants from the land surface, but the presence of contaminant sources associated with urban land use increases the aquifer's vulnerability to contamination.

Wells completed in the deeper aquifer in secondary recharge areas of the valley (classes E and F) had shallower median depths to water than did wells in the primary recharge areas (classes C and D), and contained water of modern or mixed age. Water from wells in the secondary recharge area on the east side of the valley (class E) had lower

Table 2. Summary of physical and water-quality characteristics for eight classes of wells sampled in Salt Lake Valley, Utah.

[per mil, parts per thousand; TU, tritium units; mg/L, milligrams per liter; µg/L, micrograms per liter; %, percent; pesticide and volatile organic compound (VOC) detections include estimated values below the laboratory reporting level]

Well class	A	B	C	D	E	F	G	H
Number of wells	11	19	5	10	15	9	5	6
Part of basin-fill aquifer	Shallow	Shallow	Deeper	Deeper	Deeper	Deeper	Deeper	Deeper
Recharge or discharge area	Secondary recharge area	Secondary recharge area	Primary recharge area	Primary recharge area	Secondary recharge area	Secondary recharge area	Discharge area	Discharge area
Aquifer confinement	Unconfined	Unconfined	Unconfined	Unconfined	Confined	Confined	Confined	Confined
Head gradient	Downward	Downward	Downward	Downward	Downward	Downward	Upward	Generally upward
General location	East side of valley	West side of valley	East side of valley	West side of valley	East side of valley	West side of valley	Near the Jordan River	East and west sides of valley and near the Jordan River
Interpreted age category of water	Dominantly modern	Dominantly modern	Dominantly modern	Dominantly pre-bomb era with some modern or mixed age	Modern or mixed age	Modern or mixed age	Dominantly pre-bomb era	Modern or mixed age
Land use	Mostly urban areas	Mostly urban areas	Mostly urban areas	Mostly agricultural areas	Mostly urban areas	Urban and agricultural areas	Urban and industrial areas	Mostly urban areas
Dominant sources of water used for irrigation of crops, lawns, and gardens in area	Mountain-front streams, Jordan River, groundwater	Jordan River	Mountain-front streams, groundwater	Groundwater	Mountain-front streams, Jordan River, groundwater	Jordan River	Jordan River	Mountain-front streams, Jordan River, groundwater
Physical characteristics								
Median well depth, feet	73.5	67.5	510	306	544	440	935	318
Median depth to top of well screen, feet	62.5	57	266	208	265	290	395	115
Median depth to water, feet	58.7	49.7	194	162	136	105	5	1
Median deuterium concentration, per mil	[1]-112.9	-102.1	-117.0	-118.6	-120.4	-111.2	-124.2	-113.5
Median tritium concentration, TU	12.4	12.3	21.3	1.0	7.6	10.7	0.2	11.7
Water-quality characteristics								
Median pH, standard units	7.3	7.3	7.1	7.2	7.5	7.4	7.7	7.4
Median dissolved-oxygen concentration, mg/L	5.3	5.3	7.4	7.9	[1]5.8	[1]5.6	0.5	4.8
Median dissolved-solids concentration, mg/L	414	1,300	562	696	316	615	345	675
Median nitrate concentration, mg/L	4.45	7.05	3.34	2.96	1.21	3.06	0.04	3.14
Median arsenic concentration, µg/L	1.1	11.7	0.9	[1]5	0.5	5	1.9	1.5
Number of different pesticides detected	14	10	2	3	3	4	0	3
Number of pesticide detections	23	100	5	5	4	11	0	7
Percentage of wells where pesticides were detected	82%	100%	60%	30%	20%	56%	0%	83%
Number of different VOCs detected	13	18	12	4	7	8	0	5
Number of VOC detections	42	73	22	10	[2]25	[1, 3]12	0	12
Percentage of wells where VOCs were detected	91%	95%	100%	67%	80%	67%	0%	100%

[1] One well in this classification was not sampled for this constituent or constituent group.
[2] Two samples in this classification were analyzed for a smaller set of compounds.
[3] One sample in this classification was analyzed for a smaller set of compounds.

median concentrations of dissolved solids and nitrate and a lower frequency of pesticide or VOC detections compared to upgradient wells in the unconfined part of the aquifer (class C). This is likely due to fine-grained beds impeding the downward flow of water in the aquifer in the secondary recharge area. Water from wells completed in the deeper aquifer in the secondary recharge area on the west side of the valley (class F) had more frequent pesticide detections and an isotopically heavier median concentration of deuterium, indicating that it has undergone some evaporation, than water from wells in classes D and E. The area of class F wells includes the last large parcels of agricultural land in the valley and receives a significant amount of recharge from water diverted from the Jordan River for irrigation.

Water samples from deeper wells in the discharge area that were composed predominantly of pre-bomb era water (class G) had no pesticide or VOC detections and a very low median concentration of nitrate. Although the wells in class G are generally surrounded by urban or industrial land, they have the deepest median depth to the top of the well screen (open interval) and are in areas with a dominantly upward hydraulic gradient. Water from three of the five wells had dissolved oxygen concentrations equal to or less than 0.5 mg/L, indicative of reducing conditions. In contrast, wells completed in the deeper aquifer in a discharge area, but with modern or mixed age water (class H), had higher median concentrations of dissolved solids and nitrate and pesticides and VOCs were frequently detected. This indicates that class H wells produce a component of water recharged in the valley. The median depth to the top of the interval open to the aquifer in class H wells was the shallowest of the well classes representing the deeper basin-fill aquifer in the valley. These wells were probably completed in the upper part of the confined aquifer because of the artesian conditions present when they were drilled. Changes in the vertical hydraulic gradient at and in the area of class H wells have likely occurred as a result of pumping, so that some water recharged at the land surface has moved downward past the confining layers and into the deeper aquifer.

Summary

Changes in land use and water use in Salt Lake Valley, Utah have affected groundwater quality through changes in the sources, amount, and quality of water that recharges the basin-fill aquifer system. Water that enters the aquifer in the valley (basin or valley recharge) is more susceptible to receiving man-made chemicals than is both surface flow and subsurface inflow from the adjacent mountains. Seepage of excess water from irrigated cropland, lawns, gardens, parks, and golf courses; and from leaking canals, water distribution pipes, sewer lines, storm drains, and retention basins are now sources of recharge to the basin-fill aquifer. The diversion of water from Jordan River/Utah Lake to the east side of the valley began in 1882. Water from the Jordan River is acceptable for irrigation, but not for potable uses because of turbidity and mineral content. Surface water from local streams draining the Wasatch Range and imported from outside of the local drainage basin provided about 70 percent of the public supply in 2000. This water is chlorinated and distributed for use across the valley. Groundwater withdrawal from wells in 2000 was about 28 percent of the total used for public supply.

The basin-fill deposits in the valley consist of unconsolidated to semiconsolidated Tertiary-age deposits overlain by unconsolidated Quaternary-age deposits. The groundwater system in the valley includes a shallow aquifer that is separated from a deeper aquifer by discontinuous layers or lenses of fine-grained sediment. The deeper basin-fill aquifer consists of an unconfined part near the mountain fronts that becomes confined toward the center of the valley. Groundwater discharges in areas where there is an upward gradient from the confined part of the deeper aquifer to the overlying shallow aquifer, generally in the center of the valley along the Jordan River and in the topographically lowest parts of the valley. Both the confined and unconfined parts of the aquifer are important sources of drinking water for Salt Lake Valley.

Under predevelopment conditions, recharge occurred along the mountain fronts and from the infiltration of precipitation. Mountain-front recharge is estimated to have comprised more than 70 percent of recharge to the basin-fill aquifer system under predevelopment conditions, and includes subsurface inflow from consolidated rocks in the adjacent mountains (mountain-block recharge) and seepage from major streams and precipitation runoff near the mountain front. Under modern conditions, infiltration of excess irrigation water from croplands, lawns, and gardens, and seepage from canals became major sources of recharge to the groundwater system (about 27 percent of estimated average annual recharge). Groundwater recharge has increased by almost one-third from that of predevelopment conditions, primarily due to the addition of canal seepage and excess irrigation water.

The inorganic chemical composition of groundwater depends largely on its recharge source, the type of rocks and associated minerals it has contacted, and how long the water has been in contact with the aquifer material. Major factors related to the occurrence of contaminants within the basin-fill aquifer include the locations and sources of recharge, vertical direction of groundwater movement, and aquifer properties. Water that enters the basin-fill aquifer in the valley (valley or basin recharge) is more susceptible to receiving man-made chemicals than is subsurface inflow from the adjacent mountains (mountain-block recharge). Widespread occurrence of chloroform and bromodichloromethane in both the shallow and deeper basin-fill aquifers is likely a result of recharge of chlorinated public-supply water used to irrigate lawns and gardens in residential areas of Salt Lake Valley.

The presence of human-related compounds and elevated concentrations of nitrate in the deeper basin-fill aquifer is strongly correlated with the distribution of interpreted-age categories. Nearly all of the public-supply wells where a VOC or pesticide was detected or where the nitrate concentration was greater than 2 mg/L, have either dominantly modern water (water less than 20 years old) or a mixture of modern and pre-bomb era (pre-1950) waters. With one exception, human-related compounds were not detected in groundwater with an apparent age of older than 50 years.

References Cited

Anderson, P.B., Susong, D.D., Wold, S.R., Heilweil, V.M., and Baskin, R.L., 1994, Hydrogeology of recharge areas and water quality of the principal aquifers along the Wasatch Front and adjacent areas, Utah: U.S. Geological Survey Water-Resources Investigations Report 93–4221, 74 p.

Arnow, Ted, Van Horn, Richard, and LaPray, Reed, 1970, The pre-Quaternary surface in the Jordan Valley, Utah: U.S. Geological Survey Professional Paper 700–D, p. D257–D261.

Burden, C.B., and others, 2001, Ground-water conditions in Utah, spring of 2001: Utah Department of Natural Resources, Division of Water Resources Cooperative Investigations Report No. 42, 120 p.

Burden, C.B., and others, 2005, Ground-water conditions in Utah, spring of 2005: Utah Department of Natural Resources, Division of Water Resources Cooperative Investigations Report No. 46, 120 p., accessed September 29, 2008 at http://ut.water.usgs.gov/publications/GW2005.pdf.

Calkins, F.C., Butler, B.S., and Heikes, V.C., 1943, Geology and ore deposits of the Cottonwood-American Fork area, Utah: U.S. Geological Survey Professional Paper 201, 152 p.

Fenneman, N.M., 1931, Physiography of the Western United States: New York, McGraw-Hill, 534 p.

Hely, A.G., Mower, R.W., and Harr, C.A., 1971, Water resources of Salt Lake County, Utah: Utah Department of Natural Resources Technical Publication No. 31, 244 p.

Hintze, L.F., Willis, G.C., Laes, D.Y.M., Sprinkel, D.A., and Brown, K.D., 2000, Digital geologic map of Utah: Utah Geological Survey, Map 179DM, scale 1:500,000.

Isaacson, A.E., 2004, Water use trends and supply along Utah's Wasatch Front: Utah Economic and Business Review, Bureau of Economic and Business Research, University of Utah, v. 64, nos. 7 and 8, 14 p.

Kennecott Utah Copper, 1998, Final draft remedial investigation report for Kennecott Utah Copper South Facilities groundwater plume, southwestern Jordan Valley, Utah: March 16, 1998, version B.

Lambert, P.M., 1995, Numerical simulation of ground-water flow in basin-fill material in Salt Lake Valley, Utah: Utah Department of Natural Resources Technical Publication No. 110–B, 58 p.

Lambert, P.M., 1996, Numerical simulation of the movement of sulfate in ground water in southwestern Salt Lake Valley, Utah: Utah Department of Natural Resources Technical Publication No. 110–D, 44 p.

LandScan Global Population Database, 2005, Oak Ridge, Tennessee: Oak Ridge National Laboratory, accessed May 17, 2010 at http://www.ornl.gov/landscan/.

Mabey, D.R., 1992, Subsurface geology along the Wasatch Front, *in* Gori, P.L., and Hayes, W.W., eds., Assessment of regional earthquake hazards and risk along the Wasatch Front, Utah: U.S. Geological Survey Professional Paper 1500, p. C1–C16.

Marine, I.W., and Price, Don, 1964, Geology and ground-water resources of the Jordan Valley, Utah: Utah Geological and Mineralogical Survey Bulletin 7, 68 p.

Mattick, E.R., 1970, Thickness of unconsolidated to semiconsolidated sediments in Jordan Valley, Utah: U.S. Geological Survey Professional Paper 700–C, p. C119–C124.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239. Available at http://pubs.usgs.gov/sir/2008/5239/.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM), Oregon State University, accessed on May 17, 2010 at http://www.ocs.oregonstate.edu/prism/.

Richardson, G.B., 1906, Underground water in the valleys of Utah Lake and Jordan River, Utah: U.S. Geological Survey Water-Supply Paper 157, 81 p.

Taylor, G.H., and Leggette, R.M., 1949, Ground water in the Jordan Valley, Utah: U.S. Geological Survey Water-Supply Paper 1029, 357 p.

Thiros, S.A., 1995, Chemical composition of ground water, hydrologic properties of basin-fill material, and ground-water movement in Salt Lake Valley, Utah: Utah Department of Natural Resources Technical Publication No. 110–A, 59 p.

Thiros, S.A., 2003, Quality and sources of shallow ground water in areas of recent residential development in Salt Lake Valley, Salt Lake County, Utah: U.S. Geological Survey Water-Resources Investigations Report 03–4028, 74 p. Available at http://pubs.water.usgs.gov/wri034028/.

Thiros, S.A. and Manning, A.H., 2004, Quality and sources of ground water used for public supply in Salt Lake Valley, Salt Lake County, Utah, 2001: U.S. Geological Survey Water-Resources Investigations Report 03–4325, 95 p. Available at http://pubs.water.usgs.gov/wri034325/.

U.S. Census Bureau, 2002, Quick facts for Salt Lake County, Utah, accessed December 12, 2002 at http://quickfacts.census.gov/qfd/states/49/49035.html.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

Utah Department of Natural Resources, Division of Water Resources, 1999, A water-related land use summary report of the State of Utah, accessed May 12, 2010 at http://www.water.utah.gov/planning/landuse/wrlui.pdf.

Utah Department of Natural Resources, Division of Water Resources, 2007, Water related land use map data layer, accessed May 12, 2010 at http://www.water.utah.gov/planning/landuse/Metadata.pdf.

Utah Governor's Office of Planning and Budget, 2008, Demographic and economic projections, 2008 baseline projections, accessed March 30, 2009 at http://governor.utah.gov/dea/projections.html.

Utah State Water Plan Coordinating Committee, 1997, Utah State water plan—Jordan River Basin: Utah Division of Water Resources, Salt Lake City, September 1997.

Waddell, K.M., Seiler, R.L., Santini, M., and Solomon, D.K., 1987a, Ground-water conditions in Salt Lake Valley, Utah, 1969–83, and predicted effects of increased withdrawals from wells: Utah Department of Natural Resources Technical Publication No. 87, 69 p.

Waddell, K.M., Seiler, R.L., Santini, M., and Solomon, D.K., 1987b, Chemical quality of ground water in Salt Lake Valley, Utah, 1969–85: Utah Department of Natural Resources Technical Publication No. 89, 55 p.

This page intentionally left blank.

Section 3.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in Truckee Meadows, Nevada

By Jena M. Huntington

Basin Overview

Truckee Meadows is a north-south trending basin covering about 94 mi^2 in western Nevada that is undergoing the urbanization of its rangeland and irrigated agricultural areas. Groundwater quality in the basin is influenced by both natural and human-induced factors. Truckee Meadows is bordered on the west by the Carson Range, a spur of the Sierra Nevada Range; on the east by the Virginia Range; on the north by volcanic hills related to the Carson and Virginia Ranges; and on the south by the Steamboat Hills and Pleasant Valley (fig. 1). While the average elevation of the basin is 4,500 ft, Mount Rose to the west soars to 10,778 ft, Peavine Mountain to the north rises to 8,266 ft, and the Steamboat Hills to the south reach an elevation of 6,181 ft.

The Truckee River, which originates at Lake Tahoe in the Sierra Nevada Range, flows from west to east across Truckee Meadows and exits the valley through a deeply incised canyon within the Virginia Range. Steamboat Creek, which has the Truckee River's largest tributary area (Stockton, 2003), flows northward from Pleasant Valley. The basin experiences the "rain shadow" effect due to it's location on the leeward side of the Sierra Nevada Range. This effect, coupled with the elevation of the valley floor, generates an arid desert climate with low humidity (Gates and Watters, 1992). Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an estimated average annual precipitation of about 10.4 in. over the alluvial basin as a whole (McKinney and Anning, 2009). Up to about 40 in. of precipitation falls each year in the adjacent mountains, mostly as snow.

Truckee Meadows is home to the cities of Reno and Sparks and expanding suburbs. Analysis of LandScan population data for 2005 (Oak Ridge National Laboratory, 2005) indicated a population of about 263,000 for the alluvial basin as a whole (McKinney and Anning, 2009) and a population density of about 2,750 people/mi^2. Land cover for the alluvial basin in 2001 was about 3 percent agricultural, 55 percent urban, 24 percent range, and about 18 percent for other uses (fig. 1; U.S. Geological Survey, 2003).

The movement of water through the geologic materials of the basin, coupled with anthropogenic activities and recharge from the land surface to the aquifer, results in elevated concentrations of some chemical constituents and organic compounds in groundwater. Groundwater-quality issues identified in Truckee Meadows and described later in this section include naturally occurring arsenic and elevated concentrations of other dissolved constituents, and the presence of nitrate, volatile organic compounds, and pesticides associated with human activities and land-use practices in the basin.

Water Development History

The Washoe Native American tribes were the first people to inhabit the Truckee Meadows area. Fur trading expeditions arrived in the basin in the 1820s and army expeditions began coming through Truckee Meadows en route to Sacramento, California in the 1840s. It was then that a Paiute Indian guide whose name sounded like "Truckee" became the namesake of the Truckee River (Rowley, 1984; Gates and Watters, 1992). Wagon trains followed the Truckee River Trail to California over what was to be called Donner Pass in the Sierra Nevada Range after the Donner Party starvation tragedy during the winter of 1846–47 (Gates and Watters, 1992). Gold was discovered in the Comstock Lode to the southeast of Truckee Meadows in 1859, and the town of Reno was formed to provide supplies (Rowley, 1984; Land and Land, 1995).

During the 1860s, livestock production and agriculture spread in the basin and Reno became the crossroads for the Transcontinental and Virginia and Truckee Railroads (Land and Land, 1995). Several irrigation ditches were constructed to divert water from the Truckee River to the western, southern, and northern parts of the basin. Electric companies also diverted water from the river into wooden aqueducts that hugged the canyon walls until they reached turbines downstream.

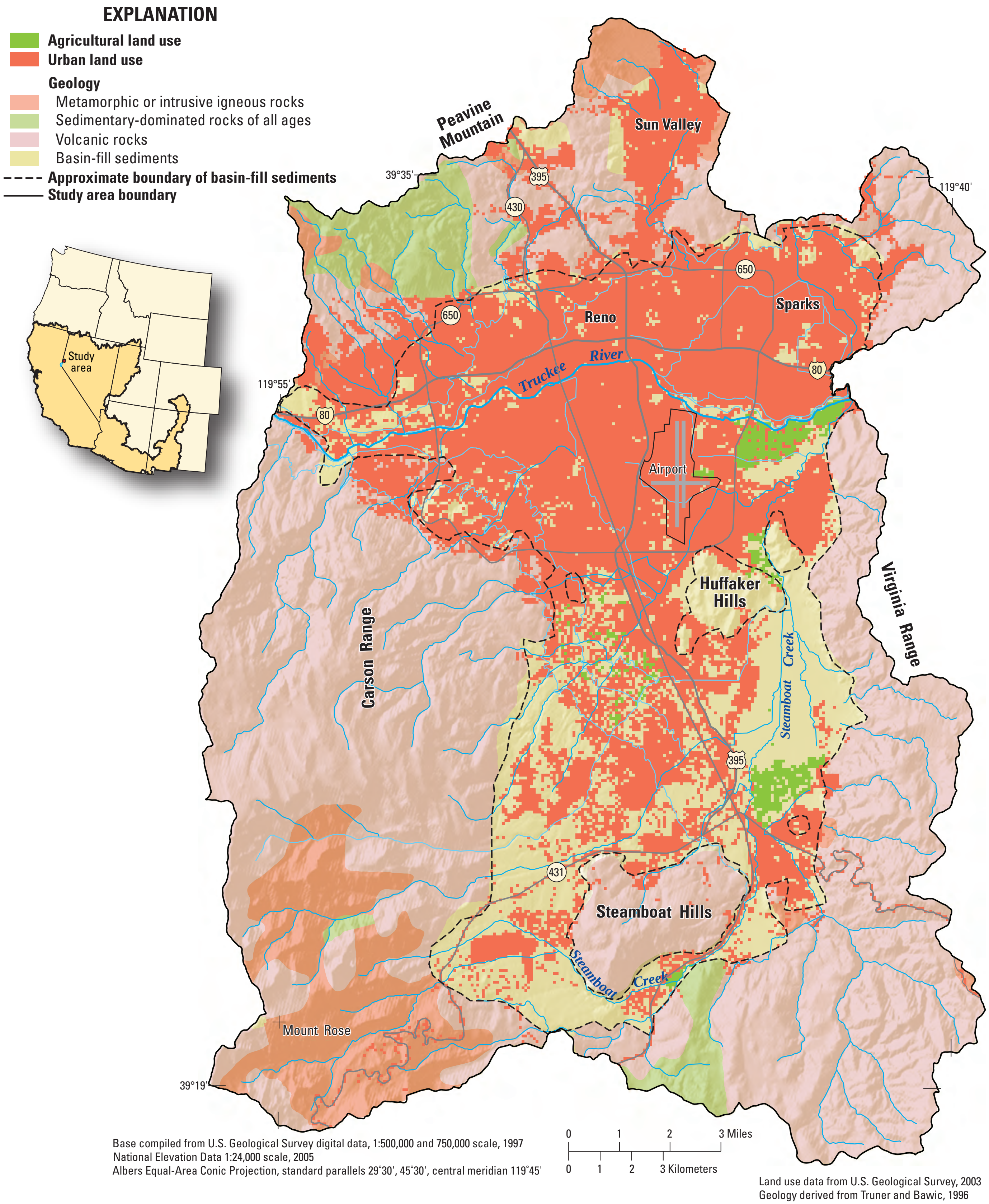


Figure 1. Physiography, land use, and generalized geology of Truckee Meadows, Nevada.

Drinking water used in Truckee Meadows historically came from the Truckee River, although contamination problems started as early as development did. Raw sewage was discharged directly into the river, and during the late 1880s, upstream sawmills began dumping sawdust into the river. Although the Truckee River served as the sole source of drinking water through the turn of the 20th century, the population in the basin grew quickly, and groundwater pumping was initiated in the late 1950s for municipal supply when a focused effort to provide a back-up source for surface water was implemented (Christopher Benedict, Washoe County Department of Water Resources, written commun., 1999). Most of the land previously used for agriculture in the basin has been urbanized. Currently, very little land is used for raising livestock or growing crops.

Hydrogeology

Truckee Meadows, like most basins in the Basin and Range Physiographic Province, is a structural depression bounded by fault-block mountains. The Carson Range to the west is made up of diverse metavolcanic and metasedimentary rocks that were intruded by granitic rocks. This sequence of rocks was mostly covered by thick flows of Tertiary volcanic rocks that include rhyolite and andesite. The geology of the Virginia Range is similar, although extrusive rocks almost completely cover the granitic base rocks (Bateman and Scheibach, 1975). Most of the consolidated rocks bordering Truckee Meadows are of low permeability and do not store or transmit appreciable amounts of water (Cohen and Loeltz, 1964, p. S8). Volcanic rocks protrude from within the basin at the Huffaker Hills and Steamboat Hills. Normal faults, generally trending north, northwest, and northeast, have been mapped through much of the basin. Geothermal water occurs in association with these faults in the Reno area and in the Steamboat Hills area (Bateman and Scheibach, 1975).

Basin-fill deposits in the Truckee Meadows basin have been divided into three general units—sedimentary rocks of Tertiary age, older alluvium of Quaternary age, and younger alluvium of Quaternary age (Cohen and Loeltz, 1964, p. S11). The Tertiary material was deposited mainly in a fluvial environment and consists of unconsolidated to partly consolidated diatomaceous sediments interbedded with coarse-grained sandstones, shales, gravels, and tuffs (Bonham and Rogers, 1983; and Trexler and Cashman, 2006). Tertiary sedimentary rocks are considered to be relatively close to land surface, within 1,150 ft, especially in the eastern parts of the basin, and are thickest in the northwest, where sediment thickness is in places more than 2,000 ft (Widmer and others, 2007). On the basis of well yield data, these rocks are considered to be of low permeability, but recent research has indicated the presence of intervals within the Tertiary sediments that are capable of transmitting appreciable volumes of water. This is particularly true in the eastern parts of the basin, where it is likely that several municipal water-supply wells have been completed in these sediments (Widmer and others, 2007; and Trexler and others, 2000).

During the Quaternary period, glacial outwash—silt, sand, gravel, and boulders—from the mountains to the west was deposited in the Truckee Meadows basin along with poorly sorted pediment and alluvial-fan deposits. This alluvium unconformably overlies the Tertiary sediments and is exposed on the Mount Rose alluvial fan complex in the southwestern part of the basin and along the Truckee River. Younger alluvial deposits are present mostly in the valley lowlands along the floodplains of the Truckee River and Steamboat Creek, along the stream channels of tributary drainages entering the basin, and along the base of alluvial fans as thin, sheet-like aprons of reworked sediment (Bonham and Rogers, 1983). Compared to the thick deposits on the west side of the basin, a relatively thin section of Quaternary age alluvial-fan deposits skirts the base of the Virginia Range, and in the central part of the basin the maximum thickness of Quaternary age deposits is thought is be less than about 650 ft (Abbott and Louie, 2000).

Deposits of highest hydraulic conductivity to transmit water lie to the north of the Huffaker Hills (Cohen and Loeltz, 1964, p. S14). Hydraulic conductivity of the basin-fill material estimated from pumping-test data ranges from about 12 to 28 ft/d. Estimates of transmissivity for the basin-fill aquifer from pumping-test data listed by Cohen and Loeltz (1964, table 4) range from 200 to 7,000 ft^2/day.

Conceptual Understanding of the Groundwater System

Truckee Meadows is an open basin drained by the Truckee River. The basin-fill aquifer system is made up primarily of unconsolidated Quaternary deposits and Tertiary sediments, although fractured bedrock influences groundwater flow and quality in some areas. Both semiconfined and unconfined conditions exist in the basin-fill aquifer. Relatively thick unsaturated zones underlie the alluvial fans to the south and north and become thinner toward the basin lowlands. Fine-grained flood plain deposits are interbedded with coarser grained stream channel deposits in the lower parts of the basin. In general, the occurrence of fine-grained sediment increases with depth due to the much lower depositional energy that was present prior to the uplift of the Sierra Nevada approximately 2 million years ago (Christopher Benedict, written commun., 2009). Because of aggradation and erosion, confining layers can be discontinuous, of variable thickness, and interbedded with more permeable deposits. Discontinuous, fine-grained fluvial deposits create confined conditions mostly in the northeastern part of the basin north of the Huffaker Hills, where they overlie saturated coarse-grained deposits. Flowing wells are present in this area, although more recent municipal well pumpage has reduced the number of flowing wells and (or) their discharge rates.

The aquifer is recharged naturally by the infiltration of precipitation falling on the surrounding mountains and basin margins and from human-related sources in the valley, such as seepage from surface-water diversions, excess irrigation water, and pumped groundwater from municipal wells that is discharged to the Truckee River and subsequently infiltrates (fig. 2). Groundwater generally flows from recharge areas in the west and south toward the Truckee River and Steamboat Creek, which may receive relatively minor amounts of groundwater seepage, and to discharge areas in the center and eastern parts of the valley, where evapotranspiration (ET) occurs. Geothermal water also enters the basin-fill aquifer along faults within the valley.

Water Budget

Recharge to the basin-fill aquifer in Truckee Meadows is from the infiltration of precipitation on the surrounding mountains and alluvial slopes (mountain-front recharge), infiltration of precipitation on the basin floor, seepage of excess irrigation water, losses from the Truckee River and ditches that divert water from the river onto the margins of the basin, artificial recharge through injection wells, and by subsurface inflow from adjacent basins (table 1). Van Denburgh and others (1973, table 12) estimated recharge from precipitation along the mountain fronts above 5,000 ft to be about 24,400 acre-ft/yr using the Maxey-Eakin method ((Maxey and Eakin, 1949; Eakin and others, 1951), which applies a percentage of average annual precipitation within specified altitude zones to estimate recharge. Most of the natural recharge originates as precipitation at high altitudes on the western part of the drainage area and enters the basin-fill aquifer as seepage from snowmelt runoff. An unknown fraction of the precipitation eventually enters the basin-fill aquifer as subsurface inflow from the surrounding consolidated rocks where they are permeable or fractured. Recharge from the infiltration of precipitation on the basin floor was estimated to be 5 percent of the average precipitation, or about 2,100 acre-ft/yr (Van Denburgh and others, 1973, table 12).

Seepage from the Truckee River to the basin-fill aquifer was estimated by Cohen and Loeltz (1964, p. S21) to be about 4,000 acre-ft/yr. Subsurface inflow to Truckee Meadows from adjacent basins was estimated by Van Denburgh and others (1973, table 13) to be about 300 acre-ft/yr from Pleasant Valley to the south and 700 acre-ft/yr through the Truckee Canyon area to the west. Rush and Glancy (1967, p. 37) estimated about 100 acre-ft/yr from Spanish Springs Valley and 25 acre-ft/yr from Sun Valley to the north. Thus the total subsurface inflow from adjacent basins is estimated to be about 1,125 acre-ft/yr, and all of the natural recharge to the Truckee Meadows basin-fill aquifer, including that from infiltration of precipitation and inflow from adjacent basins, totals about 31,600 acre-ft/yr (table 1).

Groundwater discharges naturally by ET and by seepage to the Truckee River and Steamboat Creek (both to the north and south of the Huffaker Hills). Under predevelopment conditions, the relative quantity of discharge equaled that of recharge because the system was assumed to be in equilibrium (no change in the average volume of storage). Although the quantities associated with the components of recharge to the basin-fill aquifer listed in table 1 are based on several assumptions and few data, they are considered to be within the correct order of magnitude and thus indicate the degree to which each component recharged the groundwater system.

Human related changes to the groundwater flow system beneath Truckee Meadows began in the late 1800s, when water diverted from the Truckee River for irrigation began recharging the aquifer (fig. 2). Inflow to Truckee Meadows from the Truckee River and its principal diversions averaged about 530,000 acre-ft/yr from 1919–69 (Van Denburgh and others, 1973, p. 30). Cohen and Loeltz (1964, p. S20) estimated that about 88,000 acre-ft/yr of Truckee River water was diverted and applied to 22,000 irrigated acres during the 1950s and early 1960s and that about 6,000 acre-ft/yr of canal losses recharged the basin-fill aquifer. They assumed that 25 percent of the applied irrigation water (mainly by flooding) recharged the aquifer, about 25,000 acre-ft/yr during that time. The recharge from excess irrigation water and canal losses to the groundwater system almost doubled the quantity of recharge from that of predevelopment conditions. This additional recharge resulted in a rise in groundwater levels, an increase in the volume of water stored in the aquifer, and an increase in groundwater discharge from ET and seepage to streams (Cohen and Loeltz, 1964, p. S27).

The area of irrigated agriculture in the basin has decreased since the 1960s in response to the expansion of urban land. An estimated 7,800 acre-ft/yr of water was applied to approximately 2,120 acres of irrigated fields in 2001 (McKinney and Anning, 2009). Assuming 25 percent of this amount infiltrates past the root zone, about 2,000 acre-ft/yr of excess irrigation water recharges the aquifer in agricultural areas under modern conditions. This estimate is less than one-tenth of the recharge from excess irrigation to the groundwater system in the 1960s. Only a fraction of the once expansive irrigated land remained in 2001 and even less acreage is irrigated today (2010), although diversions to ditches in the western and northern parts of the basin still averaged about 67,000 acre-ft/yr for the period from 1989 to 2002 (Regional Water Planning Commission, 2005, fig. 2-11 and p. 2-22). Many of these ditches are now lined (Christopher Benedict, written commun., 2009) and therefore the 6,000 acre-ft/yr of ditch losses, as estimated by Cohen and Loeltz, (1964) is likely less than 500 acre-ft/yr.

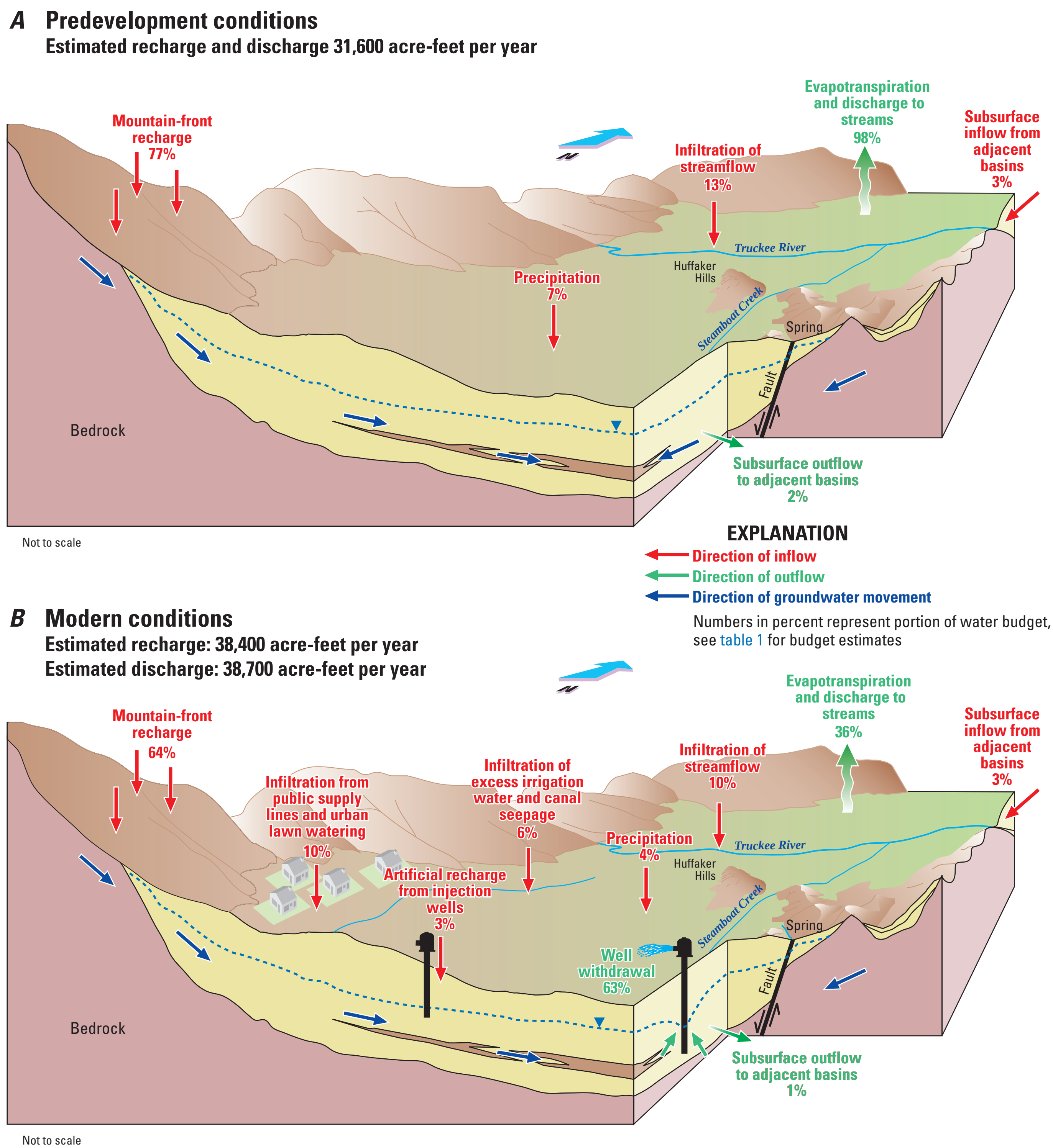


Figure 2. Generalized diagrams for Truckee Meadows, Nevada, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin-fill aquifer in Truckee Meadows, Nevada, under predevelopment and modern conditions.

[All values are in acre-feet per year and are rounded to the nearest hundred. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 2. <, less than]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Mountain-front recharge	[1]24,400	[1]24,400	0
Infiltration of precipitation on alluvial basin	[1]2,100	[9]1,600	-500
Infiltration of streamflow from the Truckee River	[2]4,000	[2]4,000	0
Subsurface inflow from adjacent basins	[1,3]1,100	[1,3]1,100	0
Infiltration of excess irrigation water and canal seepage	0	[11, 8]2,500	2,500
Infiltration from public supply lines	0	[12]2,100	2,100
Infiltration of excess urban lawn water	0	[11]1,700	1,700
Artificial recharge from injection wells	0	[4, 10]1,000	1,000
Total recharge	**31,600**	**38,400**	**6,800**
Budget component		**Estimated discharge**	
Evapotranspiration and discharge to streams	[5]31,100	[6]13,800	-17,300
Subsurface outflow to adjacent basins	[2]< 500	[2]< 500	0
Well withdrawals	0	[7]24,400	24,400
Total discharge	**31,600**	**38,700**	**7,100**
Change in storage (total recharge minus total discharge)	0	-300	-300

[1] Van Denburgh and others (1973).

[2] Cohen and Loeltz (1964).

[3] Rush and Glancy (1967).

[4] Regional Water Planning Commission, Washoe County Department of Water Resources (2005).

[5] Assumed to equal the recharge total for predevelopment conditions minus estimated subsurface outflow.

[6] Assumed to be the residual between total recharge and discharge from wells under modern conditions minus estimated subsurface outflow.

[7] Lopes and Evetts (2004).

[8] Written communication from Christopher Benedict, Washoe County Department of Water Resources, 2009.

[9] Estimated as 75 percent of predevelopment conditions, due to 49 percent urban land use in 2001 (McKinney and Anning, 2009).

[10] Truckee Meadows Water Authority (2009).

[11] Calculated from McKinney and Anning (2009).

[12] CDM and Bouvette Consulting (2002).

In 2000, about 68,000 acre-ft of water from streams and wells was supplied for public use to about 29,500 acres of urban land in Truckee Meadows (McKinney and Anning, 2009). Some of this water is used to irrigate vegetation in urban/residential areas, and depending on how efficiently the water is used, a small fraction likely infiltrates to the aquifer. Assuming that one half of the publicly supplied water is used for irrigation by sprinklers and that 5 percent of this water infiltrates into the subsurface past the root zone, then recharge from excess irrigation water in urban areas is estimated to be about 1,700 acre-ft/yr. Water also leaks from the pipes used to distribute water throughout the urban area and about 2,100 acre-ft/yr was estimated to recharge the aquifer in the central part of the basin (CDM and Bouvette Consulting, 2002).

Since 1993, chlorinated surface water has been injected during the winter months into several public-supply wells in the central part of the basin. The aquifer is used to store the injected water until it is needed during summer months when demand is highest, or during drought, when the groundwater is pumped. The total amount of water artificially recharged through injection wells from 1993 (81 acre-ft) to 2003 (2,400 acre-ft) was 10,800 acre-ft, and averaged about 980 acre-ft/yr (Regional Water Planning Commission, 2005, p. 2-13; Truckee Meadows Water Authority, 2009, p. 69).

The extraction and artificial recharge of groundwater for geothermal production in the basin is not listed in table 1. Geothermal water is pumped for power generation and then reinjected after use to approximately the same depth from which it was removed. Typically there is little or no loss between extraction and reinjection. In 2000, about 39,600 acre-ft of geothermal water was pumped and 37,700 acre-ft was reinjected (Lopes and Evetts, 2004, table 1). Steamboat Creek receives natural discharge from the Steamboat Springs geothermal area that is not included in this groundwater budget.

The Truckee River is the main source of water for public supply to the central part of Truckee Meadows. Groundwater is used to supplement the surface-water supply in the basin with about 21,200 acre-ft pumped from public-supply wells and about 2,800 acre-ft from domestic wells in 2000 (Lopes and Evetts, 2004, table 1). Withdrawals for irrigation and stock watering under modern conditions are minimal (about 380 acre-ft in 2000).

Groundwater Movement

Groundwater moves from topographically high recharge areas to lower areas of Truckee Meadows, where under natural conditions the water discharges. The general direction of groundwater flow in the basin is from southwest to northeast, toward the Truckee River (fig. 3) (Covey and others, 1996, p. 58). Groundwater naturally discharges to the Truckee River and Steamboat Creek and in areas north of the Huffaker Hills and near the Reno International Airport. Water-level contours indicate that the consolidated rock of the Huffaker Hills may transmit water through fractures.

The presence of chlorofluorocarbons (CFCs) in groundwater is used as an indicator of young water and as a tool for estimating specific groundwater ages. CFCs are man-made organic compounds that are used in industrial processes and in the home. After their introduction in the 1930s, atmospheric concentrations increased nearly exponentially until the 1990s (Plummer and Busenberg, 2000). Chlorofluorocarbons were detected in samples collected from ten wells in the Truckee Meadows from 2002 to 2008 (table 2). The ten wells ranged in depth from 14 to 760 ft, and all contained CFCs at a concentration(s) that indicates a fraction of modern water recharged less than about 50 years ago. A more specific age date is not reported because no other forms of age-dating (i.e. tritium, carbon-14) were conducted to interpret a more refined recharge date.

Additional sources and paths of recharge and discharge under modern conditions have had an effect on groundwater levels in the Truckee Meadows. Water-level declines in domestic wells in the southern part of the basin have been attributed to diminished recharge from excess irrigation in the area as agricultural land is urbanized (Regional Water Planning Commission, 2005, p. 2-19). Groundwater pumping for public supply has resulted in several changes to the groundwater flow system, primarily on a local scale. Vertical hydraulic gradients near pumping centers seasonally change from an upward to a downward gradient. Municipal well pumping has also likely caused water-level declines in parts of the basin-fill aquifer and consequently increased water loss from the Truckee River to the groundwater system (Christopher Benedict, written commun., 2009), although the volume of such loss has not been quantified.

Although the estimated groundwater budget for modern conditions (table 1) does not show a significant overall change in storage in the basin, the reduction in recharge from excess irrigation water and the increase in discharge from well withdrawals would result in the removal of water from storage in areas where other processes of discharge or recharge have not changed. Under these conditions, groundwater levels would decline, and in areas of natural discharge, ET and seepage to streams would be reduced before any water would be removed from storage. These effects are observed more frequently in areas further away from the Truckee River, as the river tends to buffer changes in storage near its channel (Christopher Benedict, written commun., 2009).

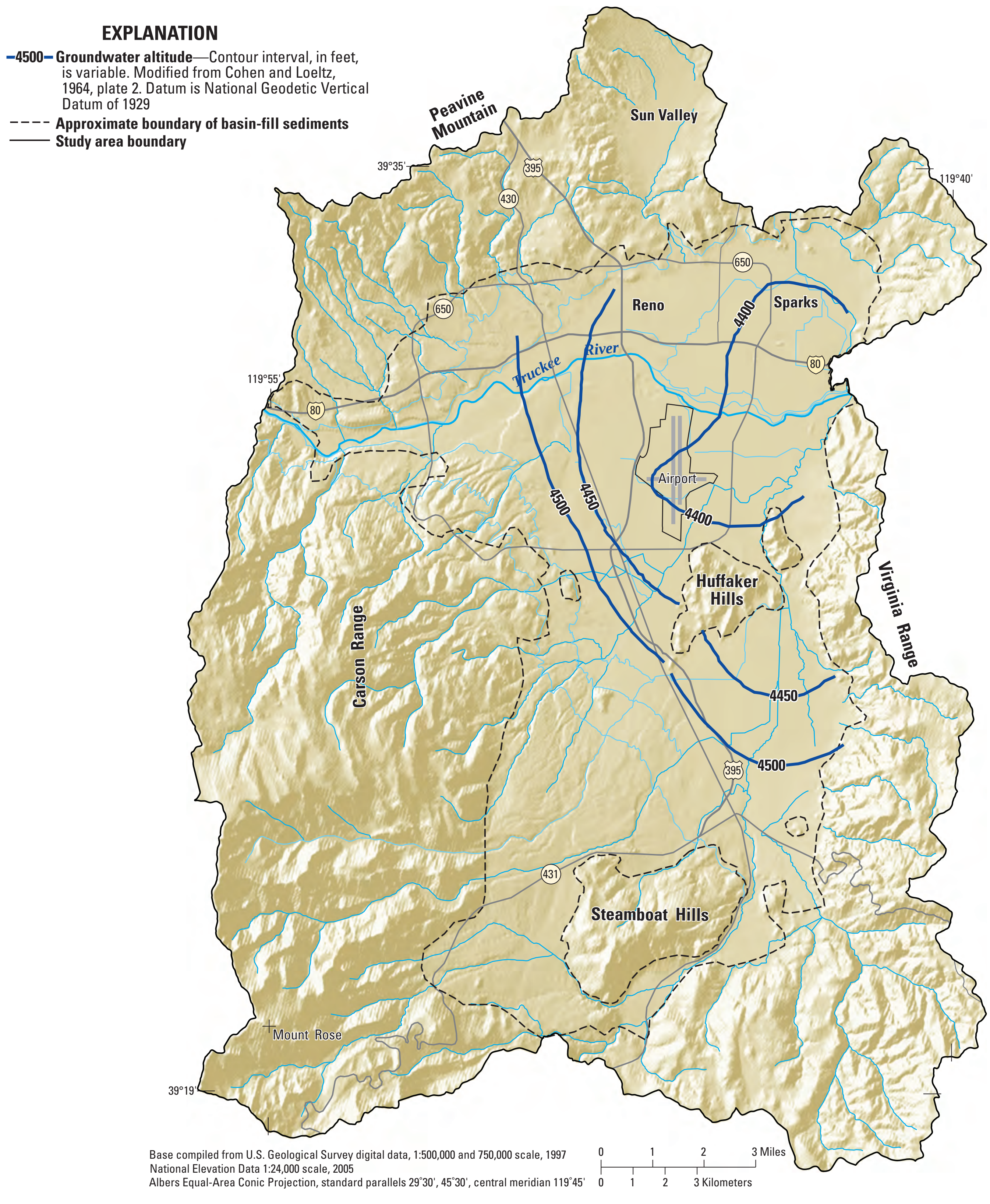


Figure 3. Generalized groundwater levels in 1962 in Truckee Meadows, Nevada.

Table 2. Designation of groundwater age in selected wells in Truckee Meadows, Nevada.

[Modern, groundwater sample contained chlorofluorocarbons at concentrations that indicate a fraction of modern water recharged less than about 50 years ago]

Station identifier	Well depth (feet)	Sample date	Water age
392837119485901	159	06-03-2002	Modern
392918119464901	21	06-27-2002	Modern
392944119440301	20	06-06-2002	Modern
392937119452601	14	06-12-2002	Modern
393023119513701	49	08-26-2006	Modern
393108119415102	26	08-30-2006	Modern
392506119462201	530	10-29-2003	Modern
392414119474701	760	11-13-2003	Modern
392231119501901	236	11-30-2003	Modern
393053119445601	191	04-08-2008	Modern

Effects of Natural and Human Factors on Groundwater Quality

Groundwater quality in the Truckee Meadows basin is influenced by both natural and human-induced factors. The movement of water through the geologic materials in the basin, coupled with the movement of water from the land surface to the aquifer, results in elevated concentrations of some constituents and compounds in groundwater. The addition of recharge sources at the land surface and increased pumping from wells facilitates the movement of water and contaminants to parts of the aquifer used for water supply. Areas in the basin most susceptible to movement of water between the land surface and the aquifer are in the western part and near the Truckee River, where confining layers are likely to be thin, discontinuous, or not present. Groundwater withdrawals also can induce the lateral movement of poor quality water to parts of the basin-fill aquifer used for water supply in the basin.

The following description of groundwater quality in Truckee Meadows is based mainly on results of the analyses of samples collected in 1994 and 1995 from 28 shallow monitoring wells and 18 water-supply (principal) wells as part of the NAWQA Program (Covay and Bevans, 1997; Bevans and others, 1998) and from other water-quality data from the basin reported by Cohen and Loeltz (1964, table 5) and Van Denburgh and others (1973, table 18). The shallow monitoring wells, which range in depth from 15 to 78 ft, were in an urban setting. The water-supply wells were from 185 to 760 ft deep. Many of these wells were resampled in 2002 and 2003, and in addition to a few wells sampled by the program for the first time, are shown on figure 4 (data listed in Berris and others, 2003; Stockton and others, 2003).

General Water-Quality Characteristics and Natural Factors

The general water-quality characteristics as well as the occurrence and concentrations of individual chemical constituents and organic compounds of groundwater in Truckee Meadows varies areally across the basin. Groundwater near the Truckee River and other streams entering the basin is generally a calcium bicarbonate type with dissolved-solids concentrations typically less than 300 mg/L. Away from streams and upland recharge areas, dissolved-solids concentrations increase and sodium bicarbonate becomes the dominant water type. Sulfate-rich groundwater is associated with hydrothermally altered consolidated rocks at several places along the margins of the basin. Sodium-chloride groundwater with high dissolved-solids concentrations occurs in the geothermal area near Steamboat Hills in the southern part of the basin. Radon concentrations (or activities) in water from the public-supply wells ranged from 300 to 1,500 pCi/L, with a median of 760 pCi/L, and uranium concentration ranged from less than 1.0 to 7.0 µg/L (Covay and Bevans, 1997). Other natural contaminants, such as iron, manganese, boron, and antimony have been detected in both shallow monitoring wells and in several public-supply wells, although concentration data are not yet available (John Hulett, Washoe County Department of Water Resources and Paul Miller, Truckee Meadows Water Authority, written commun., 2009).

Concentrations of dissolved-oxygen concentrations in water from the water-supply wells sampled by NAWQA ranged from 0.4 to 5.5 mg/L, with a median of 3.8 mg/L; pH ranged from 7.0 to 8.1, with a median of 7.6; and dissolved-solids concentrations ranged from 149 to 548 mg/L, with a median of 228 mg/L. Dissolved-oxygen concentrations in water from the shallow monitoring wells ranged from 0.1 to 6.6 mg/L, with a median of 0.3 mg/L; pH ranged from 6.5 to 8.1, with a median of 7.1; and dissolved-solids concentrations ranged from 137 to 1,460 mg/L, with a median of 420 mg/L (Covay and Bevans, 1997).

Geothermal activity in the Truckee Meadows area has an effect on the temperature and chemistry of the groundwater. The temperature of water in the sampled public-supply wells ranged from 58°F to 104°F, with the higher temperatures probably owing to geothermal activity. Geothermal systems in Truckee Meadows have contributed to naturally high concentrations of arsenic in water from the basin-fill deposits. Arsenic in groundwater (and springs) also can come from the volcanic rocks bounding the basin and from the sediment derived from these rocks. Arsenic in water samples from wells in the basin has been reported at concentrations as high as 640 µg/L (Bateman and Scheibach, 1975).

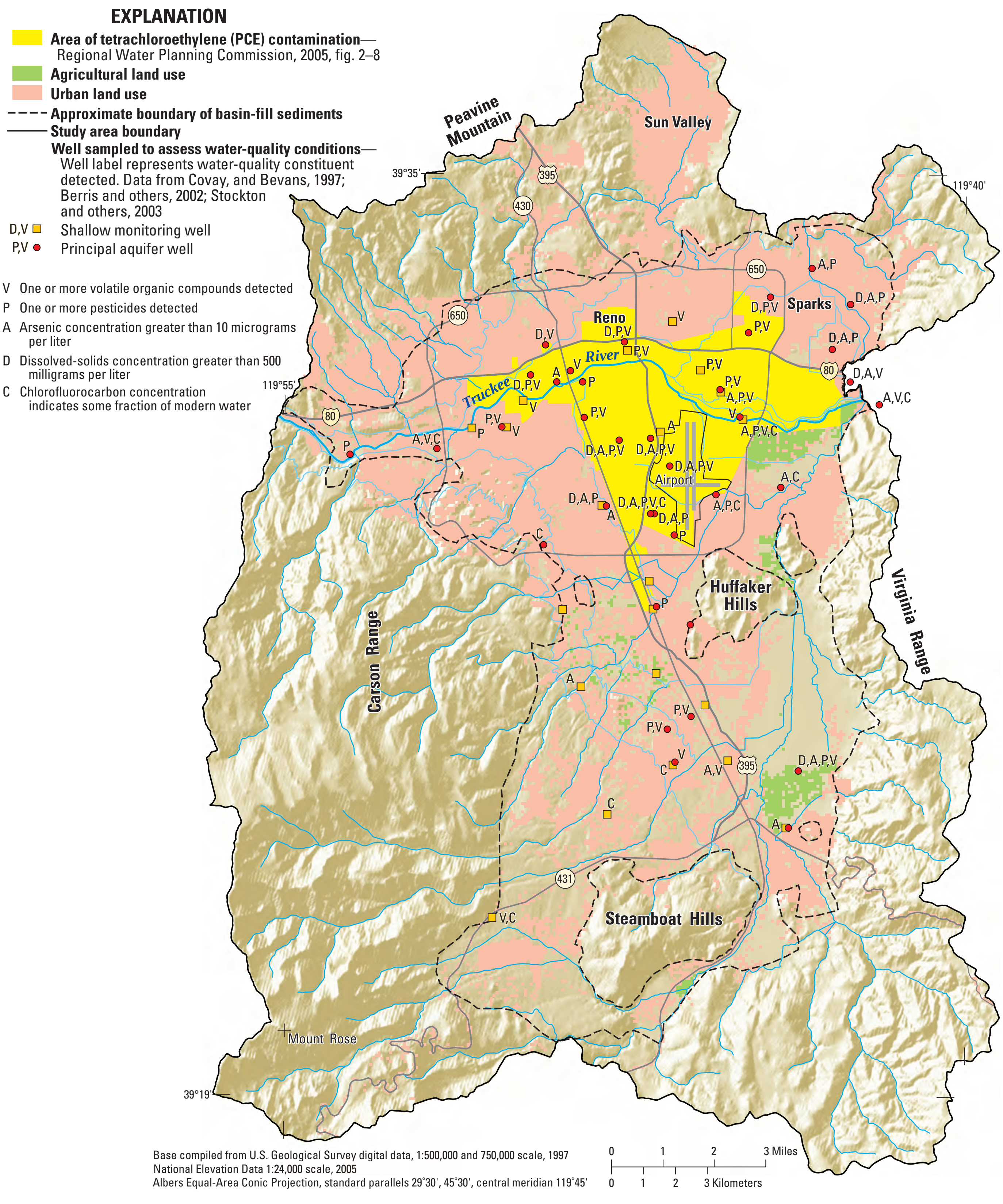


Figure 4. Location of wells in Truckee Meadows, Nevada, sampled by the NAWQA Program.

In samples collected by NAWQA investigators, concentrations of dissolved arsenic ranged from less than 1 to 92 µg/L, with a median of 5.5 µg/L, in water from 18 water-supply wells, and from less than 1 to 230 µg/L, with a median of 7.0 µg/L, in water from the 28 shallow monitoring wells (Covay and Bevans, 1997). Arsenic concentrations in water from 23 wells (7 used for water supply and 16 used for monitoring), primarily in the central and northeastern parts of the basin and sampled by the NAWQA Program between 1994 and 2003, exceeded the U.S. Environmental Protection Agency's (USEPA) maximum contaminant level (MCL) of 10 µg/L for arsenic in drinking water (U.S. Environmental Protection Agency, 2008; each time "MCL" is mentioned in this chapter, it denotes the citation USEPA, 2008). Pumping from the basin-fill groundwater system may change flow directions and gradients, resulting in the potential for movement of such arsenic enriched geothermal water to supply wells in the basin.

Potential Effects of Human Factors

The major chemical constituents and organic compounds detected in groundwater in Truckee Meadows and the processes or sources that affect their presence and concentrations are summarized in table 3. Concentrations of nitrate plus nitrite (as nitrogen) ranged from less than 0.05 to 3.6 mg/L with a median of 0.88 mg/L in water from the water-supply wells, and from less than 0.05 to 10 mg/L with a median of 1.85 mg/L in water from the shallow monitoring wells. The MCL for nitrate plus nitrite as nitrogen in drinking water is 10 mg/L, which is enforceable only in public-supply systems (USEPA, 2008). Reducing conditions caused by denitrification likely affect nitrate concentrations in some of these wells. Elevated concentrations of nitrate measured in groundwater in the southern part of Truckee Meadows were attributed to the recharge of septic system effluent (Regional Water Planning Commission, 2005, p. 2-3).

At least one pesticide was detected in 68 percent (19 of 28) of the shallow monitoring wells and in 44 percent (8 of 18) of the water-supply wells in the basin sampled by NAWQA (Covay and Bevans, 1997). The herbicide atrazine was detected in 10 monitoring wells and 3 supply wells and its degradation product deethylatrazine was detected in 11 monitoring wells and 6 supply wells, all at concentrations one to three orders of magnitude smaller than the MCL for atrazine (3 µg/L). The herbicides prometon and simazine also were detected at small concentrations in water from 5 and 7 shallow monitoring wells, respectively. Although the concentrations of these compounds are very small and not currently a health concern, their presence in the aquifer indicates the potential for their movement from the land surface and the possibility that higher concentrations may occur in the future. It is not known what proportion of pesticide contamination is residual from agricultural activities that have since decreased in extent compared to domestic and municipal landscaping activities.

Table 3. Summary of selected constituents in groundwater in Truckee Meadows, Nevada, and sources or processes that affect their presence or concentration.

[mg/L, milligrams per liter]

Constituent	General location	Median value or detections	Possible sources or processes
		Shallow aquifers	
Dissolved solids	Mostly in the north	420 mg/L	Evapotranspiration and dissolution.
Sulfate	Basin margins	61 mg/L	Associated with altered consolidated rocks.
Nitrate	Highest in the south	1.85 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems.
Volatile organic compounds	Basin wide	19	Point sources including underground gasoline tanks & solvents from repair & dry cleaners.
Pesticides	Mostly in the north	19	Lawn fertilizer.
		Principal aquifers	
Dissolved solids	Mostly in the north	228 mg/L	Evapotranspiration and dissolution.
Sulfate	Central	21 mg/L	Associated with altered consolidated rocks.
Nitrate	Highest in the south	0.88 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems.
Volatile organic compounds	Mostly in the north	10	Potential downward movement from shallow aquifers.
Pesticides	Mostly in the north	8	Potential downward movement from shallow aquifers.

Volatile organic compounds (VOCs) were detected in water from 68 percent of the shallow monitoring wells and 55 percent of the water-supply wells sampled by NAWQA in the basin (Covay and Bevans, 1997). These compounds originate near land surface, usually in urban areas, such as at gasoline stations with leaking underground-storage tanks and at dry cleaners that use solvents. The most commonly detected compounds were chloroform, tetrachloroethylene (PCE), and methyl *tert*-butyl ether (MTBE). Chloroform, a trihalomethane, was detected in samples from 6 shallow monitoring wells and 5 water-supply wells. Its presence in the aquifer is most likely from the recharge of chlorinated water used for public supply through seepage from distribution lines and infiltration of excess landscape irrigation water. PCE was detected at concentrations ranging from 0.8 to 20 ug/L in samples collected as part of NAWQA studies in 1994–95 from 4 shallow monitoring wells and from 3 water-supply wells; the MCL for PCE is 5 µg/L. Studies conducted as part of a study by the Central Truckee Meadows Remediation District have documented PCE in groundwater (fig. 4) to depths greater than 350 ft in an area of about 16 mi^2 (Regional Water Planning Commission, 2005, p. 2-17). Remediation plans and treatment facilities are in place to remove the PCE from the water supply. MTBE, a gasoline additive that is water-soluble and therefore can readily reach the water table through permeable sediments, was detected in samples from 6 shallow monitoring wells. Although MTBE is an unregulated compound, the U.S. Environmental Protection Agency (1997) advises that concentrations of MTBE in drinking water should be less than 20 to 40 µg/L to avoid an unpleasant taste and odor as well as the potential for adverse health effects. Water samples from two shallow monitoring wells had MTBE concentrations of 140 and 220 µg/L, but MTBE was not detected in samples collected from the deeper water-supply wells (Covay and Bevans, 1997).

Summary

The Truckee Meadows basin in western Nevada is undergoing the urbanization of its rangeland and irrigated agricultural areas. The Truckee River provided most of the water used in the basin in 2000 while groundwater supplied about 27 percent. The complex basin-fill aquifer system, consisting of both unconsolidated Quaternary and Tertiary sediments, is under both leaky-confined and unconfined conditions. The aquifer is recharged naturally by the infiltration of Truckee River water, precipitation falling on the surrounding mountains and basin margins, and by human-related sources of water in the valley, such as seepage from surface-water diversions and excess irrigation water. Natural recharge to the basin-fill aquifer in Truckee Meadows is estimated at about 31,600 acre-ft/yr. Groundwater generally flows from recharge areas in the west and south toward the Truckee River and Steamboat Creek, which may receive relatively minor amounts of groundwater seepage, and to discharge areas in the center of the valley where the water is lost to ET.

Human-related changes to the Truckee Meadows groundwater flow system began in the late 1800s when water diverted from the Truckee River for irrigation increased recharge to the basin-fill aquifer. By the early 1960s, seepage from excess irrigation water and canal losses to the groundwater system almost doubled the quantity of recharge from that of predevelopment conditions and resulted in a rise in groundwater levels, an increase in the volume of water stored in the aquifer, and an increase in groundwater discharge through ET and seepage to streams. The area of irrigated agriculture in the basin has decreased since the 1960s, resulting in a decrease in recharge associated with irrigation, although this decrease was accompanied by an increase in municipal groundwater pumping since the late 1950s.

Groundwater quality in the Truckee Meadows basin is influenced by both natural and human-induced factors. The addition of recharge sources at the land surface and increased pumping from wells facilitates the movement of water and contaminants to parts of the aquifer used for water supply. Areas most susceptible to the movement of water and any included contaminants from land surface are in the western part of the basin and near the Truckee River, where the confining layers are likely to be thin, discontinuous, or not present.

Groundwater near the Truckee River and other streams entering the basin typically has dissolved-solids concentrations less than 300 mg/L. Sodium-chloride groundwater with high dissolved-solids concentrations occur in geothermal areas within the basin. These geothermal systems have contributed to naturally high arsenic concentrations in water from the basin-fill deposits and from springs issuing from volcanic rock. Arsenic in groundwater also can come from the volcanic rocks bounding the basin and from the sediment derived from these rocks. Concentrations of arsenic in water from 23 wells sampled by the NAWQA Program, mostly in the central and northeastern parts of the basin, exceeded the drinking-water standard for arsenic of 10 µg/L.

Reducing conditions in the aquifer likely affect nitrate concentrations through denitrification, although elevated concentrations measured in groundwater in the southern part of Truckee Meadows were attributed to the recharge of septic system effluent. At least one pesticide was detected in 68 percent of the shallow monitoring wells and in 44 percent of the water-supply wells in the basin sampled by NAWQA. Volatile organic compounds were detected in water sampled from 50 percent of the shallow monitoring wells and 39 percent of the supply wells sampled. Remediation plans and treatment facilities are in place to remove tetrachloroethylene from groundwater in the central part of the basin.

References Cited

Abbott, R.E., and Louie, J.N., 2000, Depth to bedrock using gravimetry in the Reno and Carson City, Nevada, area basins: Geophysics, v. 65, no. 2, p. 340–350.

Bateman, R.L. and Scheibach, R.B., 1975, Evaluation of geothermal activity in the Truckee Meadows, Washoe County, Nevada: Nevada Bureau of Mines and Geology Report 25, 38 p.

Berris, S.N., Crompton, E.J., Joyner, J.D., Ryan, Roslyn, 2003, Water resources data, Nevada, water year 2002: U.S. Geological Survey Water-Data Report NV–02–1. Available at http://pubs.usgs.gov/wdr/WDR-NV-02-1/.

Bevans, H.E., Lico, M.S., and Lawrence, S.J., 1998, Water quality in the Las Vegas Valley area and the Carson and Truckee River Basins, Nevada and California, 1992–96: U.S. Geological Survey Circular 1170, 47 p. Available at http://pubs.usgs.gov/circ/circ1170/.

Bonham, H.F., and Rogers, D.K., 1983, Geologic map, Mt. Rose NE quadrangle: Nevada Bureau of Mines and Geology, Map 4Bg.

CDM and Bouvette Consulting, 2002, Central Truckee Remediation District Remediation Management Plan vol. 2: prepared for Washoe County Department of Water Resources, October 28, 2002.

Cohen, P.M., and Loeltz, O.J., 1964, Evaluation of hydrogeology and hydrogeochemistry of Truckee Meadows area Washoe County, Nevada: U.S. Geological Survey Water Supply Paper 1779–S, 63 p.

Covey, K.J., Banks, J.M., Bevans, H.E., and Watkins, S.A., 1997, Data on ground-water quality, Reno-Sparks area, Nevada, 1994 and 1995: U.S. Geological Survey Open-File Report 97–222, 1 map.

Covay K.J., and Bevans, H.E., 1997, Data on ground-water quality, Reno-Sparks area, Nevada, 1994 and 1995: U.S. Geological Survey Open-File Report 97–222, 1 plate.

Eakin, T.E., Maxey, G.B., Robinson, T.W., Fredericks, J.C., and Loeltz, O.J., 1951, Contributions to the hydrology of eastern Nevada: Nevada State Engineer, Water Resources Bulletin 12, 171 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Gates, W.C.B., and Watters, R.J., September 1992, Geology of Reno and Truckee Meadows, Nevada: Bulletin of the Association of Engineering Geologists, v. 29, no. 3, p. 229–298.

Land, B.N., and Land, M.E., 1995, A short history of Reno, University of Nevada Press, Reno, Nevada.

Lopes, T.J., and Evetts, D.M., 2004, Ground-water pumpage and artificial recharge estimates for calendar year 2000 and average annual natural recharge and interbasin flow by hydrographic area, Nevada: U.S. Geological Survey Scientific Investigations Report 2004–5239, 88 p. Available at http://pubs.usgs.gov/sir/2004/5239/.

Maxey, G.B., and Eakin, T.E., 1949, Ground water in White River Valley, White Pine, Nye, and Lincoln Counties, Nevada: Nevada State Engineer, Water-Resources Bulletin 8, 59 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239. Available at http://pubs.usgs.gov/sir/2008/5239/.

Oak Ridge National Laboratory, 2005, LandScan Global Population Database, accessed September, 2006 at http://www.ornl.gov/landscan/.

Plummer, L.N. and Busenberg, E., 2000, Chlorofluorocarbons *in* Cook, P.G., and Herczeg, A.L., eds., Environmental tracers in subsurface hydrology: Boston, Kluwer Academic Publishers, p. 441–478.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM): Oregon State University, accessed on April 15, 2005 at http://www.ocs.oregonstate.edu/prism/.

Regional Water Planning Commission, 2005, 2004–2025 Washoe County comprehensive regional water management plan: Washoe County Department of Water Resources, accessed on December 22, 2008 at http://www.co.washoe.nv.us/water/rwmp/index/.

Rowley, W.D., 1984, Reno, hub of the Washoe Country: Windsor Publications, Inc., Woodland Hills, California.

Rush, F.E., and Glancy, P.A., 1967, Water-resources appraisal of the Warm Springs-Lemmon Valley area, Washoe County, Nevada: Nevada Department of Conservation and Natural Resources, Water Resources Reconnaissance Report 43, 70 p. Available at http://water.nv.gov/home/publications/recon.cfm.

Solomon, D.K., Cook, P.G., and Sanford, W.E., 1998, Dissolved gases in subsurface hydrology *in* Kendall, C., and McDonnell, J.J, eds., Isotope tracers in catchment hydrology: Amsterdam, Elsevier Science, p. 291–318.

Stockton, E.L., Jones, C.Z., Rowland, R.C., and Medina, R.L., 2003, Water resources data, Nevada, water year 2003: U.S. Geological Survey Water-Data Report NV–03–1. Available at http://pubs.usgs.gov/wdr/wdr-nv-03-1/.

Trexler, J.H., Jr., Cashman, P.H., Henry, C.D., Muntean, T.W., Schwartz, K., TenBrink, A., Faulds, J.E., Perlins, M., and Kelly, T.S., 2000, Neogene basins in western Nevada document tectonic history of the Sierra Nevada-Basin and Range transition zone for the last 12 Ma *in* Lageson, D.R., Peters, S.G., and Lahren, M.M., eds., Great Basin and Sierra Nevada: Boulder, Colorado, Geological Society of America Field Guide 2, p. 97–116.

Trexler, J.H. Jr., and Cashman, P.H., 2006, Informal narrative: a brief geologic history of the Reno, Nevada area, with emphasis on implications for the Reno basin subsurface geology and hydrogeology: prepared for the Washoe County Department of Water Resources, May.

Truckee Meadows Water Authority, 2009, 2010–2030 Water resources plan, accessed May 2009 at http://www.tmh2o.com/water_system/resources/2030wrp.

Truner, R.M., and Bawic, W.J., 1996, Digital map of Nevada at scale of 1:500,000, digital dataset for Nevada geology.

U.S. Environmental Protection Agency, 1997, Drinking water advisory: Consumer acceptability advice and health effects analysis on methyl tertiary-butyl ether (MTBE), Fact Sheet EPA-882-F-97-009, December 1997, 4 p.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed February 2007 at http://www.mrlc.gov/.

Van Denburgh, A.S., Lamke, R.D., and Hughes, J.L., 1973, A brief water-resources appraisal of the Truckee River Basin, western Nevada: Nevada Division of Water Resources, Reconnaissance Report 57, 122 p. Available at http://water.nv.gov/home/publications/recon.cfm.

Widmer, M.C., Cashman, P.H., Benedict, F.C. Jr. and Trexler, J.H. Jr., 2007, Neogene through Quaternary stratigraphy and structure in a portion of the Truckee Meadows Basin: a recorded recent tectonic history: GSA Abstracts with Programs, v. 39, no. 4, Cordilleran Section Meeting.

Section 4.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifers in Eagle and Carson Valleys, Nevada

By Jena M. Huntington

Basin Overview

Eagle Valley is a small valley about 30 mi south of Reno, Nevada that has undergone rapid urban development. The valley is bounded to the north by the Virginia Range, to the east by Prison Hill, and to the west by the Carson Range (fig. 1). These mountains rise to altitudes of about 8,000 ft, 5,700 ft, and greater than 9,200 ft, respectively (Maurer and others, 1996). To the south, the boundary between Eagle Valley and Carson Valley is marked by a subtle alluvial divide (Welch, 1994). The Eagle Valley floor has an area of about 15,000 acres (23 mi^2) and lies at an altitude of about 4,700 ft (Maurer and Berger, 1997).

Carson Valley is adjacent to and south of Eagle Valley (fig. 1). The Pine Nut Mountains bound the valley to the east and rise gradually to altitudes of about 8,000 to 9,000 ft. Like Eagle Valley, the Carson Range borders Carson Valley to the west rising abruptly to altitudes between 9,000 and 11,000 ft. The valley floor is oval-shaped with an area of about 104,000 acres or 162 mi^2, and slopes northward from an altitude of about 5,000 ft at its southern end to about 4,600 ft at its northern end (Maurer and others, 2004).

Eagle and Carson Valleys have a semiarid climate as a result of their location within the rain shadow of the Sierra Nevada Range. Annual precipitation on the floor of Eagle Valley averages about 10 in., while along the crest of the Carson Range precipitation averages about 38 in/yr. The Virginia Range receives much less precipitation than the Carson Range—slightly more than 14 in/yr (Schaefer and others, 2007). Annual precipitation on the floor of Carson Valley averages 8.4 in. (period of record 1971–2000; National Oceanic and Atmospheric Administration, 2002, p. 12). However, the Carson Range in this area receives 25.5 in. of precipitation per year (period of record 1971–2000, Western Regional Climate Center, 2003) and precipitation on the Pine Nut Mountains averages 15.7 in/yr (period of record 1984–2002; Dan Greenlee, Natural Resources Conservation Service, written commun., 2003). In both mountain ranges, most precipitation falls as rain or snow during November through April. Snow in the Carson Range accumulates to depths of many feet during most winters and melts in early spring to early summer. Other climatic characteristics of Eagle and Carson Valleys are prevailing westerly winds, large daily temperature fluctuations, and infrequent, but severe storms (Garcia and Carman, 1986).

Urban land occupies more than half of Eagle Valley while irrigated agricultural land and rangeland makes up nearly half of Carson Valley, according to the National Land Cover Database (NLCD) dataset for 2001 (U.S. Geological Survey, 2003). Analysis of LandScan population data for 2005 (Oak Ridge National Laboratory, 2005) indicated a population for the alluvial basin as a whole to be about 48,000 for Eagle Valley and 36,000 for Carson Valley (McKinney and Anning, 2009). This equates to a population density of about 2,055 and 220 people/mi^2 for Eagle and Carson Valley, respectively. The increase in population in Eagle Valley beginning in the early 1960s has slowly expanded Carson City's initial city limits in all directions and has caused a shift from a historically agrarian society to a more urban society (Covay and others, 1996). Eagle Valley supports about 1,100 acres of irrigated agricultural land, mostly consisting of pasture. This shift in land use from agriculture to urban will likely affect the basin-fill groundwater system due to changes in sources and quality of recharge. Total water use in the Eagle Valley in 2000 was about 20,000 acre-ft; 81 percent of which was for public supply (McKinney and Anning, 2009). Groundwater provides about 61 percent of public supply. In Carson Valley, diversions from the Carson River, which runs south to north, and pumped groundwater is used to irrigate about 45,000 acres of agricultural land, primarily alfalfa, pasture and flax. Groundwater is the sole source of public supply in Carson Valley.

The movement of water through geologic materials of the basins coupled with movement of water from the land surface to the basin-fill aquifers results in elevated concentrations of some constituents and compounds in groundwater. Groundwater-quality issues identified in Eagle and Carson Valley and described later in this section include naturally occurring uranium and other dissolved constituents, and the presence of nitrate, volatile organic compounds, and pesticides associated with anthropogenic sources in the basins.

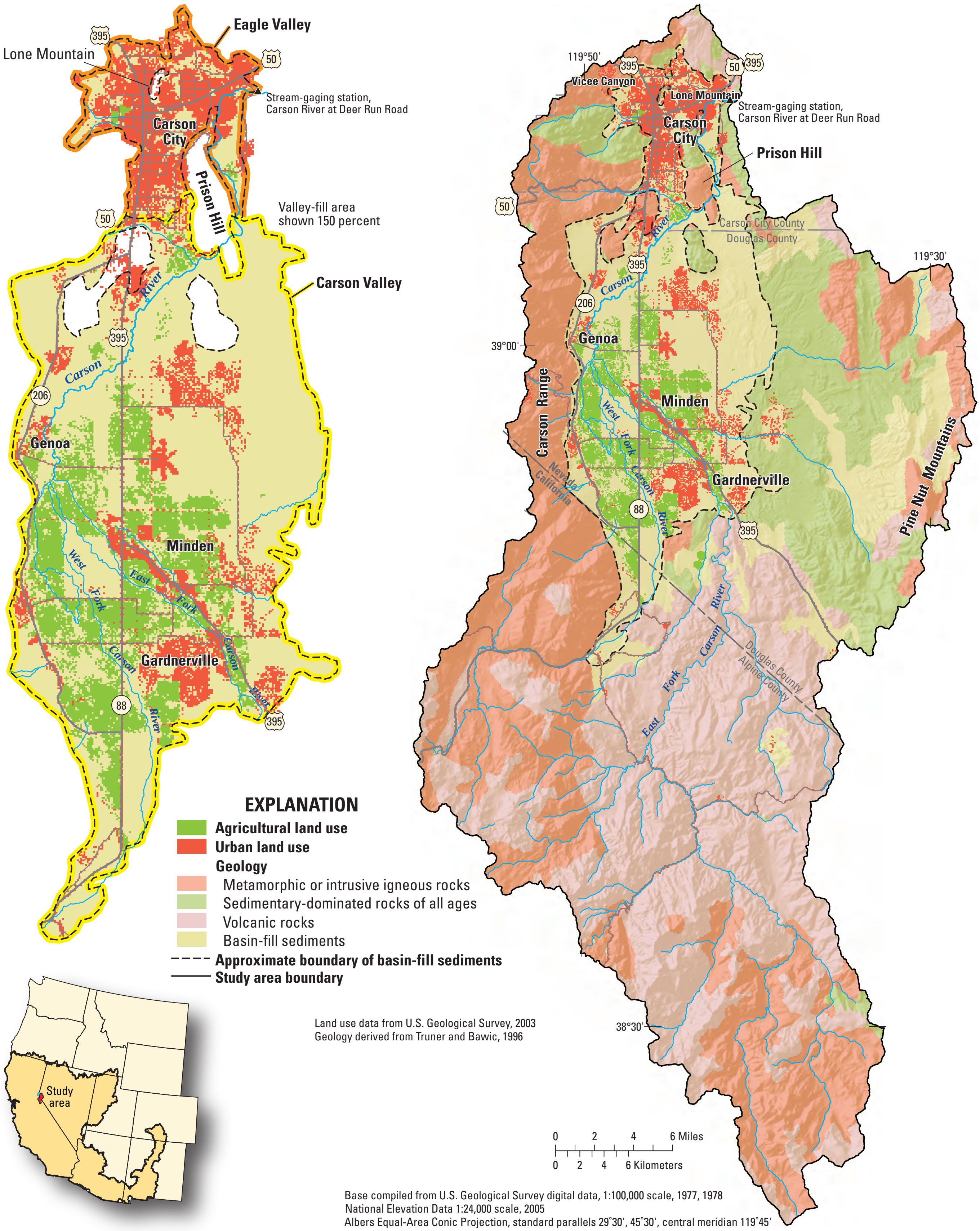


Figure 1. Physiography, land use, and generalized geology of Eagle and Carson Valleys, Nevada.

Water Development History

Eagle Valley

Accounts of early travelers through Eagle Valley in June 1859 describe it as being a small but fertile valley along the towering snow-covered Carson Range. A few acres of green meadows and cultivated fields irrigated with water from a small stream gave an inviting appearance upon entering the valley. Carson City was the only development within Eagle Valley and consisted of about a dozen small houses and two stores at that time (Simpson, 1876). Carson City expanded gradually to serve ranching, irrigated farming, and silver and other mineral mines in the area.

With the increase in population in Eagle Valley (fig. 2), water use has shifted from agricultural to domestic purposes. Historically, surface water was the major source of public supply and groundwater was used only intermittently. Groundwater has since become the major source of municipal supply, accounting for about 80 percent of that supply in 2004 (Kenneth Arnold, Carson City public works, oral commun., 2006) and public-water systems serve most of the population in Eagle Valley. Most homes are served by a wastewater-treatment plant that exports treated effluent to a reservoir in the Pine Nut Mountains (Schaefer and others, 2007). Since 1997, some of the effluent has been returned to Eagle Valley for irrigation of golf courses and alfalfa fields (Maurer and Thodal, 2000).

Carson Valley

Carson Valley was inhabited by the Washoe Indians in 1848, when a small party of Mormons arrived with plans to cut a shorter wagon route from Salt Lake City, Utah to Sacramento, California over the Sierra Nevada Range. The wagon route that they created, otherwise known as the California Trail, the Carson River Route, or the Emigrant Trail, became a highly traveled route that brought immigrants and prosperity to Carson Valley (Dangberg, 1972). In August 1853, a local newspaper reported that in May of that year at least 1,000 wagons and 300,000 cattle and sheep traveled through Carson Valley on the California Trail (Dangberg, 1972).

Diversions from the East and West Forks of the Carson River aided in turning southern Carson Valley into a productive agricultural area. Only 260 acres of land were irrigated in 1852, but more acres were added as an increased number of people traveled through the valley. Large mining operations on the Comstock Lode in Virginia City and Gold Hill to the northeast were accompanied by an increase in population and in irrigated acreage in Carson Valley (Dangberg, 1972).

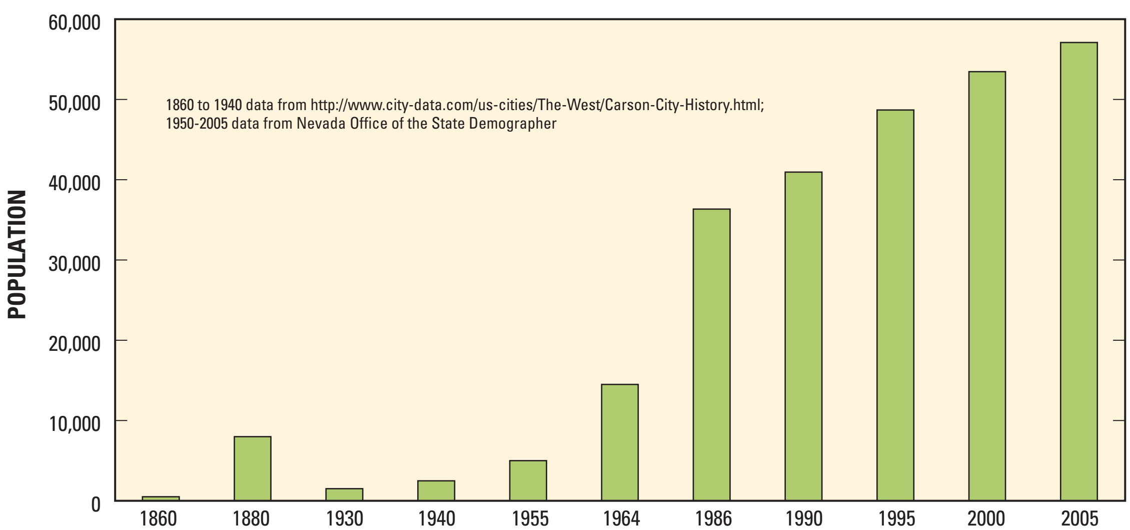


Figure 2. Population in the Carson City area of Eagle Valley, Nevada from 1860 to 2005.

Three main towns lie within Carson Valley: Genoa, the first settlement along the Sierra Nevada front; Gardnerville, established in the 1860s as an agricultural town in the center of the valley; and Minden, adjacent to Gardnerville and established in 1905 as the railroad hub for the valley (Toll, 2008). Captain Simpson of the U.S. Army Corps of Topological Engineers described Carson Valley during a visit to Genoa in 1859. He stated that Carson Valley was beautiful, "fenced off, as it appears, into inclosures, and dotted with cattle" (Simpson, 1876).

Although Carson Valley has been a major agricultural area since the 1850s, urbanization around Gardnerville, Minden, Genoa and subdivisions around Johnson Lane, Indian Hills, and Gardnerville Ranchos have grown steadily (fig. 3). Development is also increasing along the eastern and western sides of the valley. Most of the newly urbanized land was historically agricultural land. Factors responsible for population increases are available residential property, desirable aesthetic qualities, and growth in Nevada's gaming industry (Thodal, 1996).

Surface water, in the form of treated effluent, has been imported to Carson Valley from the Lake Tahoe Basin since the late 1960s and from Eagle Valley since 1988 (Maurer and Berger, 2006; Nevada State Demographer's Office, accessed on September, 11, 2006). Imported effluent is applied as irrigation water and is stored in reservoirs and wetlands (Maurer and Berger, 2006). Groundwater is exported from Washoe Valley to the north into Eagle Valley to supply Carson City's municipal uses (Nevada State Water Plan, 1999).

Hydrogeology

The mountains surrounding Eagle and Carson Valleys were created during Basin-and-Range faulting, which began about 17 million years ago (Stewart, 1980). They consist of consolidated rocks that have been uplifted by extensional tectonics near the base of the mountains while the valley floor was dropped. This faulting formed a basin that is partly filled with sediments eroded from the surrounding mountains during the Quaternary period. Movement along some faults within the last 300 to 12,000 years (Trexler, 1977) indicates that uplift of the mountains is continuing (Maurer and others, 1996).

Mesozoic-age granite and metamorphosed rocks crop out to the north and west of Eagle Valley and near Prison Hill, and most likely underlie most of the valley floor (Moore, 1969). In the Virginia Range, Tertiary sandstone and volcanic rocks consisting mostly of rhyolite, andesite, and basalt flows, flow breccias, and tuffs overlie the granite and metamorphosed rocks (Moore, 1969; Trexler, 1977).

Quaternary sediments of two ages are present in Eagle Valley. The older sediments form fans at the mouths of deeply incised canyons on the western side of the valley. Small individual fans merge into one wide fan extending as much as 1 mi eastward into the valley from the mountain front and are made up of partly consolidated to unconsolidated gravel, sand, and silt, with discontinuous clay layers (Maurer and others, 1996). Similar fans are present at the base of the Virginia Range to the north and Prison Hill to the east (Trexler and

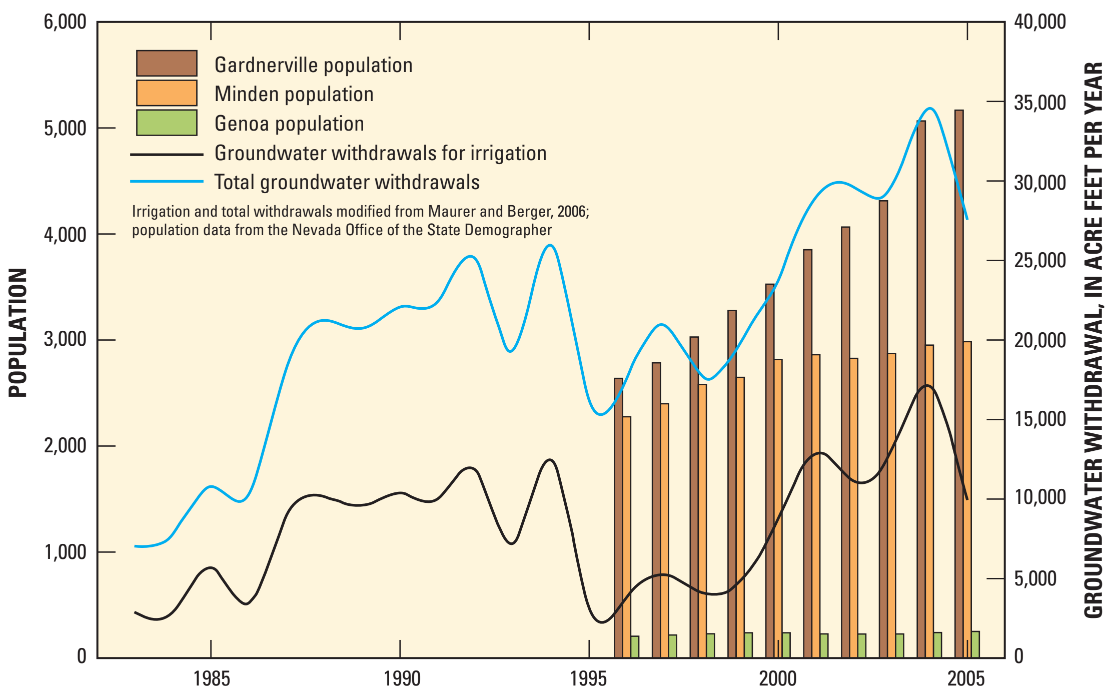


Figure 3. Population in Gardnerville, Minden, and Genoa, Nevada from 1996 to 2005 and annual groundwater pumping in Carson Valley, Nevada and California, 1983-2005.

others, 1980). The discontinuity of clay layers in the central part of the basin enable a direct hydraulic connection from the land surface to the basin-fill aquifer and make the aquifer susceptible to contamination from sources at the surface (Lico, 1998, p. 1). The younger sediments in the valley lowlands consist of fine-grained sands, silty and muddy sands, and clay (Arteaga, 1986; Trexler and others, 1980). Overall, basin-fill sediments are coarse-grained near the base of the mountains and finer grained near the center of the valley. The basin-fill sediments are estimated to be about 1,200 ft thick at a point 1.5 mi west of Lone Mountain, about 400 to 800 ft thick beneath the northeastern and southern parts of Eagle Valley, and about 2,000 ft thick about 1 mi northwest of Prison Hill (Arteaga, 1986). In general, the deepest part of the alluvial basin is in the center of the Eagle Valley (Schaefer and others, 2007).

Similar to the rocks in Eagle Valley, exposed consolidated rocks in Carson Valley are mostly granitic, metavolcanic, and metasedimentary, and make up most of the Carson Range and the Pine Nut Mountains (Covay and others, 1996). These same rocks underlie the floor of Carson Valley (Moore, 1969, p. 18). Volcanic rocks are exposed on the northeastern and southeastern end of the valley; westward dipping, semiconsolidated rocks are exposed on the eastern side of the valley (Maurer and Berger, 2006).

Both semiconsolidated Tertiary sediments and unconsolidated Quaternary basin-fill sediments are present in Carson Valley (Maurer, 1986). Poorly sorted coarse- to fine-grained unconsolidated sediments deposited by tributary streams form alluvial fans at the base of the mountain blocks (Maurer and Berger, 2006). The alluvial aquifer is made up of Quaternary sediments that were deposited on the valley floor by the Carson River and its tributary streams. Most of those sediments are well-sorted sand and gravel, interbedded with fine-grained silt and clay from overbank flood deposits (Maurer, 1986; Maurer and Berger, 2006). Thickness of the basin-fill sediments generally exceeds 1,000 ft (Maurer, 1986). Due to the downward tilting to the west of the Pine Nut Mountains relative to the uplift along the eastern margin of the Carson Range, the thickest section of the basin-fill deposits, more than 5,000 ft, lies west of the valley axis (Moore, 1969; Maurer, 1986).

Estimated hydraulic conductivities of the basin-fill sediments in Eagle Valley, those values used in the most recent groundwater flow model, range from about 1 to 31 ft/d in shallow sediments and from 0.03 to 155 ft/d in the deeper, coarser sediments that constitute the more transmissive part of the aquifer (Schaefer and others, 2007). In Carson Valley, hydraulic conductivity values estimated from pump-test data range from 14.7 to 16.4 ft/d (U.S. Bureau of Reclamation, written commun., 1981). Maurer (1986) calculated hydraulic conductivity values ranging from about 1 to 9 ft/d in sediments between 300 and 500 ft deep and from 86 to 865 ft/d in sediments less than 300 ft deep on the basis of proportions of coarse- and fine-grained material indicated in well logs.

Conceptual Understanding of the Groundwater System

Eagle Valley is small open basin with no surface-water drainage, although the Carson River flows just beyond the southeastern basin boundary (fig. 1). The river acts as both a recharge and discharge boundary to the groundwater system on the south and east sides of the basin, respectively. The mean annual flow in the Carson River from 1979–2001 was 501 ft^3/s at the streamgaging station at Deer Run Road (fig. 1) (Schaefer and others, 2007).

Carson Valley is an open basin drained by the Carson River. The East and West Forks of the river enter Carson Valley from the south, join near Genoa and continue north. A long period of record, dating back to the turn of twentieth century, is available to determine the mean annual inflow of the Carson River (Maurer and others, 2004). The East Fork inflow (period of record 1890–2002) was 276,400 acre-ft and the West Fork inflow (period of record 1901–2002; Berris and others, 2003, p. 178 and 185) was 80,320 acre-ft, which totals to 356,720 acre-ft. Mean annual outflow of the mainstem of the Carson River for the period 1940–2002 (Berris and others, 2003, p. 191) was 296,500 acre-ft.

Water Budgets

Eagle Valley

Prior to agricultural and urban development, recharge to the basin-fill aquifer in Eagle Valley was from the infiltration of precipitation—on the surrounding mountains, the alluvial slopes (mountain-front recharge), and the basin floor—and by infiltration of flow through the channels of canyon creeks entering the valley from the west (fig. 4 and table 1). Worts and Malmberg (1966, table 2) used the method of Maxey and Eakin (1949) to estimate recharge from precipitation along the mountain fronts at about 8,300 acre-ft/yr. The method applies a percentage of the average annual precipitation within specified altitude zones to estimate recharge. The bulk of this natural recharge from precipitation originates at high altitudes on the western part of the drainage area and enters the basin-fill aquifer as seepage from snowmelt runoff. The aquifer is also recharged by an estimated 3,000 to 6,000 acre-ft/yr of snowmelt that infiltrates consolidated rocks where they are permeable or fractured and moves along flow paths into basin fill (Maurer and Berger, 1997, p. 32). Recharge from the infiltration of precipitation on the basin floor was estimated to be about 400 acre-ft/yr (Worts and Malmberg, 1966). Infiltration of water from the channels of canyon creeks to the basin-fill aquifer was estimated by Maurer and Thodal (2000) to be about 2,600 acre-ft/yr based on an estimated average conditions (table 1).

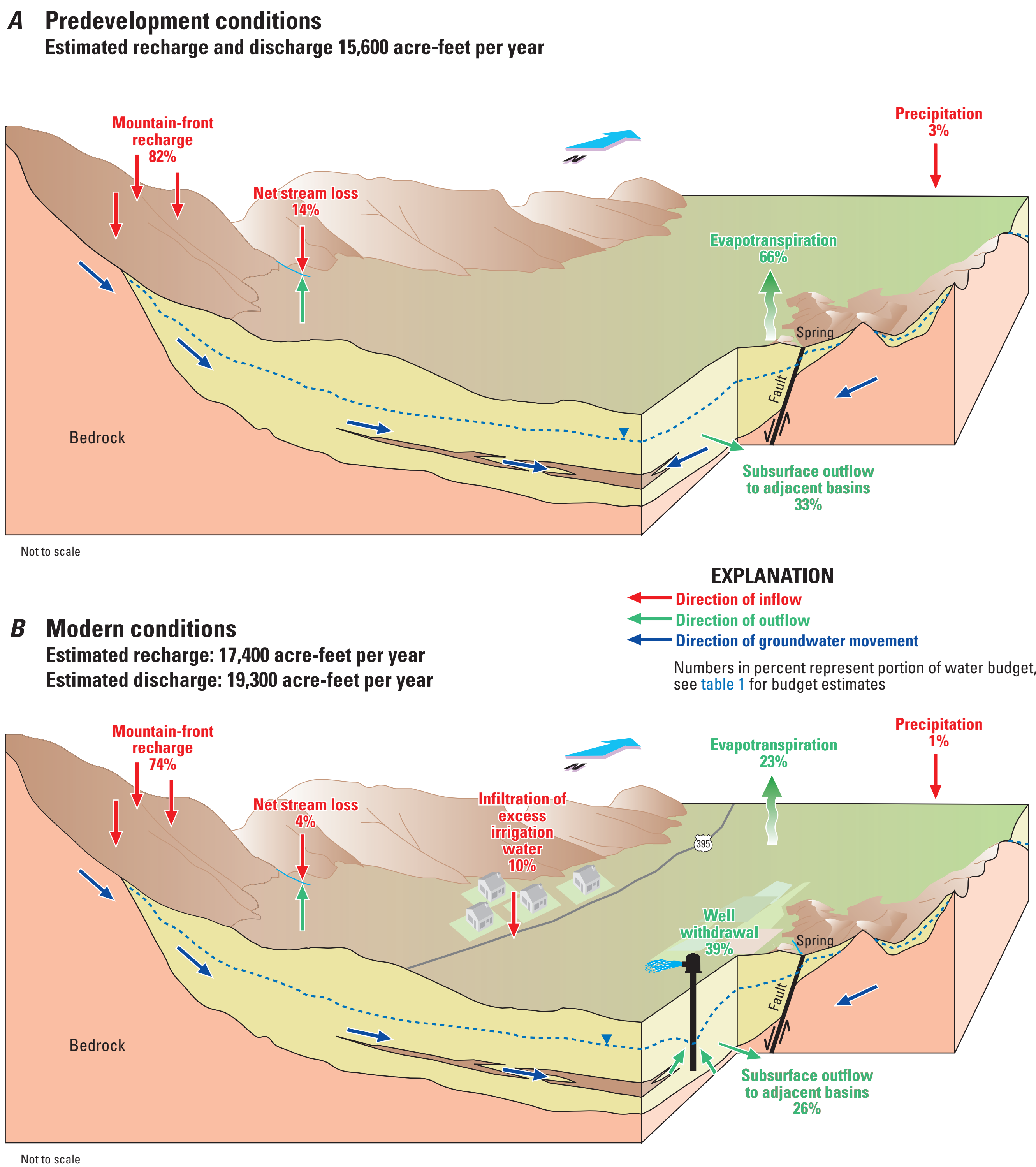


Figure 4. Generalized diagrams for Eagle Valley, Nevada, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment conditions and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin-fill aquifer in Eagle Valley, Nevada, under predevelopment and modern conditions.

[All values are in acre-feet per year (acre-ft/yr) and are rounded to the nearest hundred. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 4]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Mountain-front recharge	[5]12,800	[1]12,900	100
Infiltration of preciptation on basin	[2]400	[4]100	-300
Infiltration of streamflow	[4,6]2,400	[4,6]2,600	200
Infiltration of excess irrigation water	0	[4]1,800	1,800
Total recharge	**15,600**	**17,400**	**1,800**
Budget component		**Estimated discharge**	
Subsurface outflow to adjacent basins	[4]5,100	[4]5,100	0
Evapotranspiration	[3]10,300	[1]4,500	-5,800
Well withdrawals	0	[1]7,500	7,500
Discharge to streams	[4,6]200	[1,6]2,200	2,000
Total discharge	**15,600**	**19,300**	**3,700**
Change in storage (total recharge minus total discharge)	0	-1,900	-1,900

[1] Simulated by calibrated groundwater flow model for 1997-2001 average conditions (Schaefer and others, 2007).

[2] Estimated natural conditions by Worts and Malmberg (1966).

[3] Assumed to equal estimated residual between predevelopment recharge and discharge.

[4] Estimates from Maurer and Thodal (2000), averages are shown here where estimated ranges of values were documented.

[5] Maurer and Berger estimated recharge from snowmelt infiltrating consolidated rock and moving along flow paths into the basin fill from 3,000 to 6,000 acre-ft/yr (1997, p. 32), an average of 4,500 acre-ft/yr was assumed here, in addition to the 8,300 acre-ft/yr estimated by Worts and Malmberg (1966, table 2) using the Maxey-Eakin method.

[6] Net stream loss is represented in figure 4A & B and was calculated as gross stream loss - gross stream gain; under predevelopment conditions 2,400 acre-ft/yr - 200 acre-ft/yr = 2,200 acre-ft/yr net stream loss; under modern conditions 2,600 acre-ft/yr – 2,200 acre-ft/yr = 400 acre-ft/yr net stream loss.

Groundwater discharges in Eagle Valley through subsurface outflow to adjacent basins and by evapotranspiration (ET). Maurer and Thodal (2000) estimated that 2,900 acre-ft/yr of subsurface outflow from Eagle Valley enters Carson Valley to the south—about 400 acre-ft/yr of outflow beneath Clear Creek and, based on water yield deficiencies, an additional 2,500 acre-ft/yr flows out of the valley beneath the upper part of the Clear Creek watershed. Maurer and Berger (1997) also estimated about 2,200 acre-ft/yr of subsurface outflow to Dayton Valley to the east. Under predevelopment conditions, the relative quantity of discharge equaled that of recharge because the system was assumed to be in equilibrium (no change in the average volume of storage).

Human-related changes to the Eagle Valley groundwater flow system first began when mountain creeks were diverted for irrigated agriculture and more recently as a consequence of the conversion of farmlands to urban use. This land-use change resulted in a reduction in ET by phreatophytes (phreatophyte-area reductions from 7.7 mi^2 in 1964 to about 1.7 mi^2 in 2000) and an increase in recharge from irrigated lawns, infiltration of treated waste-water effluent on golf courses, and effluent from septic tanks (Maurer and Thodal, 2000; McKinney and Anning, 2009; Schaefer and others, 2007). Infiltration of excess urban irrigation was estimated by Maurer and Thodal (2000) to range from 1,300 to 2,300 acre-ft/yr. Increases in groundwater pumping since the 1970s, mostly for municipal supply, has diverted groundwater that was historically discharged by phreatophytes or flowed to the Carson River. Therefore the decrease in ET is attributed to both fewer phreatophytes and increases in groundwater pumping (table 1; Schaefer and others, 2007).

Additional groundwater (not indicated in table 1) is imported to Eagle Valley from other basins, including Washoe Valley to the north, Dayton Valley to the east, and Carson Valley to the south (Nevada Division of Water Resources, 1999). Surface-water transfers are received from the Lake Tahoe Basin to the west and from the Carson River in Dayton Valley. All transferred water is used for Carson City municipal supply. Beginning in 1991, artificial recharge (through infiltration beds) was initiated in Vicee Canyon on the northwestern side of Eagle Valley.

Groundwater pumping has caused water-level declines in the northwestern and southern parts of Eagle Valley (Maurer and Thodal, 2000; Schaefer and others, 2007; Arteaga, 1986, fig. 3), whereas water-level fluctuations in the center of the valley reflect variations in annual precipitation. Although water levels have increased in a few wells, no change in hydraulic gradients in the valley have been detected. Of the wells with higher water levels, a few are near golf courses and the increases are probably a response to irrigation, whereas water-level increases in other wells may be a consequence of land-cover changes from native vegetation (phreatophytes) to residential development (Maurer and Thodal, 2000).

Carson Valley

Prior to agricultural and urban development, the basin-fill aquifer in Carson Valley was recharged by subsurface inflow from adjacent basins, the infiltration of precipitation on the surrounding mountains and alluvial slopes (mountain-front recharge), infiltration of precipitation on the basin floor, and infiltration of stream water from the Carson River (fig. 5 and table 2). Maurer and Thodal (2000) estimate approximately 2,900 acre-ft/yr of groundwater inflow from Eagle Valley to the north. Four methods have been used to estimate the amount of recharge to the aquifer from the mountains and alluvial slopes of Carson Valley:

Method	Recharge (acre-ft/yr)	Reference
Water yield	22,000	Maurer and Berger, 2006
Chloride balance	40,000	Maurer and Berger, 2006
Altitude precipitation	25,000	Glancy and Katzer, 1976
Watershed modeling	35,000	Jeton and Maurer, 2007

For the purposes of this report, a value of 30,500 acre-ft/yr, the average of the four estimates, is used to represent mountain-front recharge (table 2). Precipitation that falls near the valley floor is recharged on the western alluvial fans (about 300 acre-ft/yr) and in Quaternary gravels and eolian sand deposits (at an average rate of 500 acre-ft/yr, Maurer and Berger, 2006, table 6). Infiltration of water from the Carson River and other smaller streams is difficult to quantify, as most estimates were made after diversion of streamflow began for irrigation in the basin. Maurer and Berger (2006, table 22) estimate a minimum of 10,000 acre-ft/yr of groundwater recharge by infiltration through stream channels, mostly during summer months, when groundwater levels are low; for the purposes of this report, about one-fourth of that value, or 2,500 acre-ft/yr, is assumed to occur. These components of groundwater recharge to the Carson Valley groundwater system under pre-development conditions total about 36,700 acre-ft/yr (table 2).

Natural groundwater discharge in Carson Valley occurs by means of discharge to streams, ET, and springs (table 2). Very little groundwater, less than 100 acre-ft/yr, flows from Carson Valley into Dayton Valley to the northeast (Glancy & Katzer, 1976). Groundwater discharge to streams from the basin-fill aquifer (mainly to the Carson River), about 15,000 acre-ft/yr, occurs mostly during winter months, when groundwater levels are high (Maurer and Berger, 2006, table 22). Spring discharge was calculated on the basis of flow rates reported in Glancy and Katzer (1976, table 27) as about 1,000 acre-ft/yr. Under predevelopment conditions, the relative quantity of discharge was assumed to equal that of recharge because the system was considered to be in equilibrium; therefore, the estimate of ET calculated here represents the residual of 20,600 acre-ft/yr (table 2).

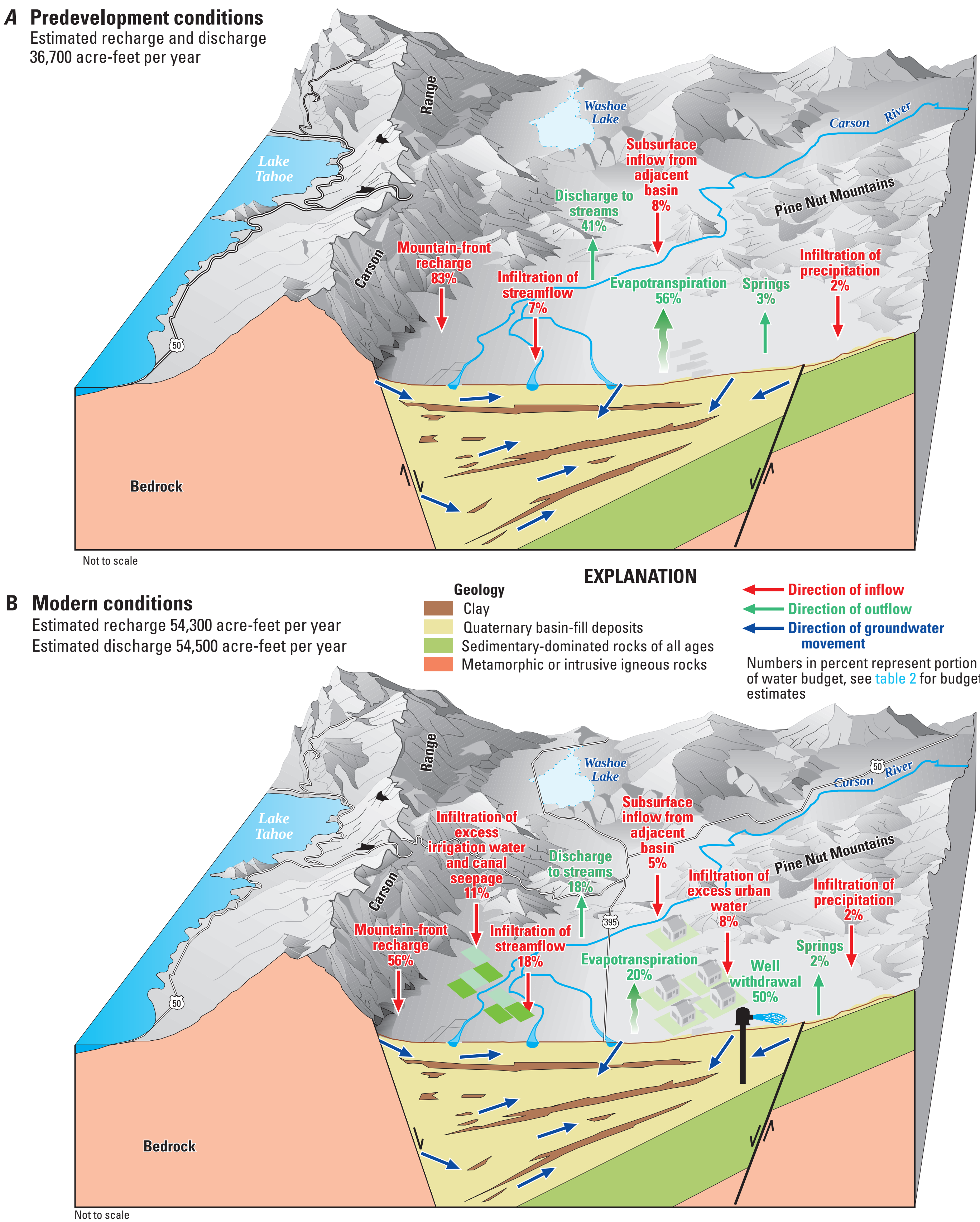


Figure 5. Generalized diagrams for Carson Valley, Nevada, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment conditions and (*B*) modern conditions.

Table 2. Estimated groundwater budget for the basin-fill aquifer in Carson Valley, Nevada, under predevelopment and modern conditions.

[All values are in acre-feet per year (acre-ft/yr) and are rounded to the nearest hundred. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 5. <, less than]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Subsurface inflow from adjacent basin	[1]2,900	[1]2,900	0
Mountain-front and mountain-block recharge	[5]30,500	[5]30,500	0
Infiltration of precipitation on basin	[2]800	[2]800	0
Infiltration of excess irrigation water and canal seepage	0	[2]6,000	6,000
Infiltration of streamflow	[8]2,500	[2]10,000	7,500
Infiltration of excess urban irrigation water and septic tanks	0	[7]4,100	4,100
Total recharge	**36,700**	**54,300**	**17,600**
Budget component		**Estimated discharge**	
Evapotranspiration	[4]20,600	[2]11,000	-9,600
Springs	[6]1,000	[6]1,000	0
Well withdrawals	0	[2]27,400	27,400
Discharge to streams	[2]15,000	[2]15,000	0
Subsurface outflow to adjacent basin	[3]< 100	[3]< 100	0
Total discharge	**36,700**	**54,500**	**17,800**
Change in storage (total recharge minus total discharge)	0	-200	-200

[1] Maurer and Thodal (2000, table 9).

[2] Maurer and Berger (2006, table 22).

[3] Glancy and Katzer (1976).

[4] Assumed to equal estimated residual between predevelopment recharge and discharge.

[5] Averaged value of estimates using different methods from Maurer and Berger (2006), Glancy and Katzer (1976) and Jeton and Maurer (2007).

[6] Calculated from spring discharge estimates (Glancy & Katzer, 1976, table 27).

[7] Average of range given in Maurer and Berger (2006, table 18) for secondary recharge from lawn watering and septic tanks.

[8] Estimated as one-quarter of 10,000 acre-ft/yr published in Maurer and Berger (2006).

Human-related changes to the Carson Valley groundwater flow system started as early as 1850, when the Carson River was first diverted for irrigated agriculture. Maurer and Berger (2006) estimate about 6,000 acre-ft/yr of return flow from irrigation pumping and about 4,100 acre-ft/yr of urban irrigation return from lawn watering and seepage from septic tanks (table 2). Groundwater discharge by ET was estimated to be about 11,000 acre-ft/yr (Maurer and Berger, 2006). This is considerably less than the estimated ET under predevelopment conditions, when areas of natural wetlands, greasewood, and riparian vegetation were more extensive, and prior to construction of the irrigation-ditch system and the clearing of fields. Total groundwater pumping in Carson Valley was about 27,400 acre-ft/yr in 2005 (fig. 3; Maurer and Berger, 2006). Because of the uncertainty in the estimates of these groundwater budget components, a numerical model of groundwater flow in Carson is being developed by the U.S. Geological Survey to help refine the estimates.

The largest change since the early 1900s in Carson Valley that affects the groundwater system has been the conversion of agricultural land or areas of natural phreatophytic vegetation to residential or commercial use. Other changes are those in water use and use patterns, for example, increased application of treated wastewater and groundwater for irrigation, and changes in the configuration of the surface-water irrigation distribution system (Maurer and Berger, 2006). Converting agricultural land to residential or commercial land would have the effect of decreasing ET (table 2) as well as increasing flow in the Carson River—via runoff from impervious surfaces—that subsequently discharges from Carson Valley. Water levels on the eastern side of Carson Valley have declined by nearly 20 ft since the early 1980s due to changes in the configuration of the irrigation distribution system, namely the discontinued use of a reservoir that was active since the early 1900s (Maurer and Berger, 2006). No groundwater gradient reversals have been observed.

Groundwater Movement

Eagle Valley

In the northern part of the Eagle Valley, groundwater flows eastward and southeastward beneath the topographic divide into Dayton Valley (fig. 6; Worts and Malmberg, 1966; Arteaga, 1986; Maurer and Berger, 1997). In the southern part of the Eagle Valley, some groundwater flows northeastward around the northern end of Prison Hill and southeastward beneath the topographic divide into Carson Valley (Worts and Malmberg, 1966; Arteaga, 1986). Unconfined to confined conditions are present in the basin-fill sediments. Clay lenses throughout Eagle Valley separate the shallow water-table aquifer from the one or more deeper confined alluvial aquifers (Arteaga, 1982). The degree of confinement varies spatially through the valley due to the clay lenses being discontinuous at different depths. The area of thickest basin-fill sediment, northwest of Prison Hill, has the most pronounced confined conditions. It is here that groundwater flow from the north, northwest and southwest converge and generally move east toward the Carson River (Welch, 1994).

Modern groundwater (less than about 50 years old) typically indicates an aquifer is susceptible to human activities at the land surface. Chlorofluorocarbons (CFCs), an indicator of young groundwater, were analyzed in samples collected from 13 wells ranging in depth from 20 to 700 ft in Eagle Valley from 2002–2008. Table 3 shows that water from the wells contained concentrations of CFCs such that each has a fraction of modern water recharged less than about 50 years ago. A more specific age date is not reported because no other forms of age-dating (such as tritium, carbon-14) were conducted to interpret a more refined recharge date.

Carson Valley

Depth to groundwater is generally deeper to the east and west, near the mountain ranges, and shallower in the center of Carson Valley. The shallow groundwater table of about 5 ft below land surface along the center of the valley is maintained by infiltration of Carson River water that is diverted across the valley floor through canals, ditches, and flood-irrigated fields (Maurer and Peltz, 1994, sheet 2). Beneath alluvial fans to the west, depth to water is greater than 200 ft within 1 mi of the Carson Range, and groundwater moves eastward (fig. 7). Depth to water beneath alluvial fans to the east is about 200 ft within 3 mi of the Pine Nut Mountains, and groundwater moves westward (Maurer and Peltz, 1994, sheet 2). Groundwater, therefore, moves generally toward the Carson River (fig. 7) and then continues northward parallel to the river (Berger and Medina, 1999).

Samples collected from seven wells in Carson Valley in 2003 were analyzed for CFCs. Samples from all of the wells contained concentrations of CFCs such that each has a fraction of its water recharged less than about 50 years ago (table 3). A more specific age date is not reported because no other forms of age-dating were conducted to interpret a more refined recharge date. The presence of such young groundwater indicates relatively rapid infiltration and downward movement from the land surface, and the potential for any contaminants in the water to move deeper into the aquifer.

Although groundwater exists under both confined and unconfined conditions in Carson Valley, no single confining layer extends across the entire valley (Covey and others, 1996). Rather, the confining layers occur mainly as scattered, discontinuous clay beds, 30 to 70 ft thick, at a depth of 200 to 300 ft. Artesian conditions exist on the west side of the valley, although at shallower depths of about 100 ft (Maurer and Berger, 2006).

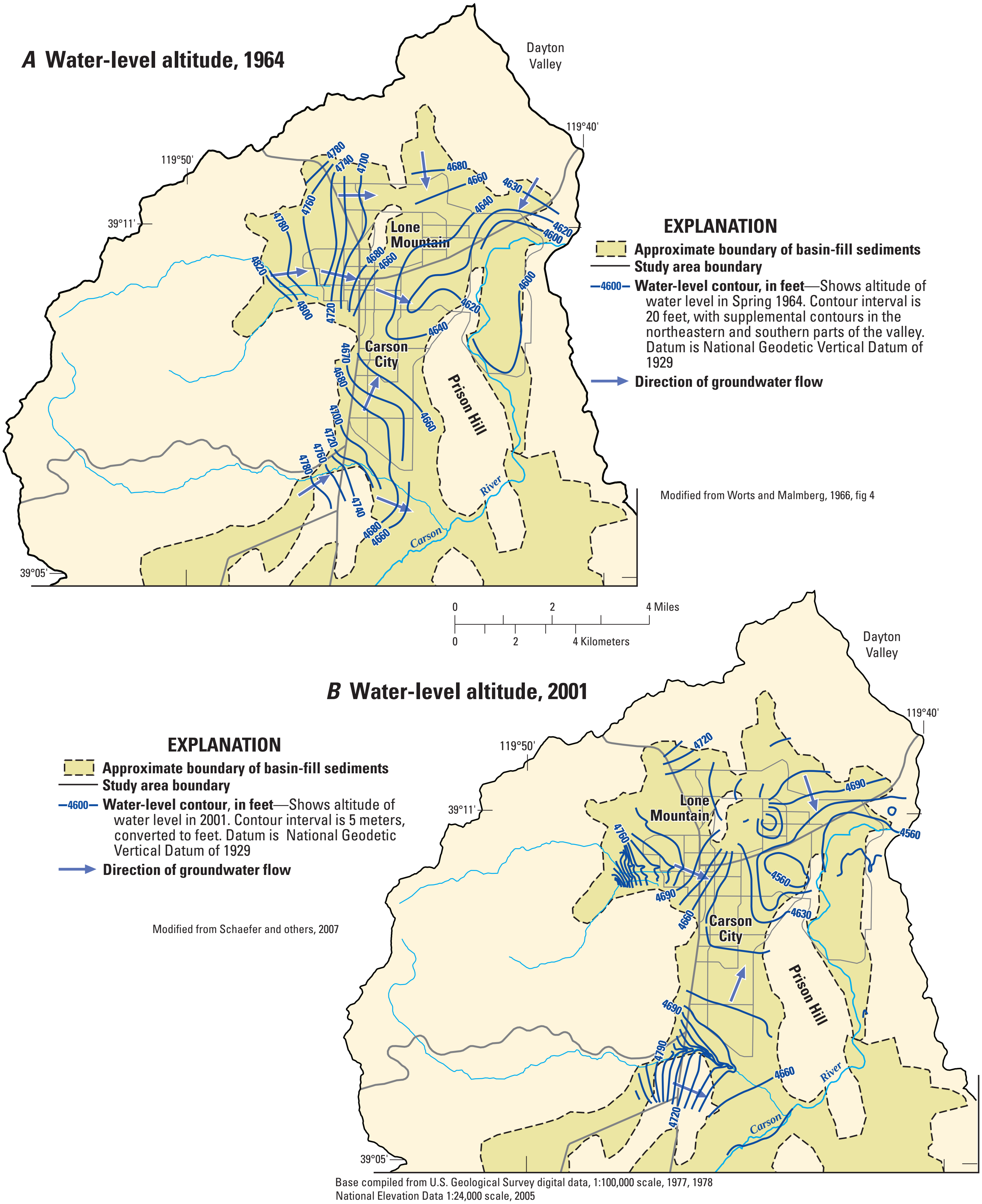


Figure 6. Water-level altitudes and general direction of groundwater flow in Eagle Valley, Nevada, in (*A*) 1964 and (*B*) 2001.

Table 3. Designation of groundwater age in selected wells in Eagle and Carson Valleys, Nevada.

[Modern, groundwater sample contained chlorofluorocarbons at concentrations that indicate a fraction of modern water recharged less than about 50 years ago]

Station identifier	Well depth (feet)	Sample date	Water age
Eagle Valley			
391030119480701	185	05-28-2002	Modern
390943119474801	108	06-26-2002	Modern
391110119460601	98	05-13-2002	Modern
391110119460602	20	05-13-2002	Modern
390834119450701	28	06-11-2002	Modern
390708119450301	140	08-29-2006	Modern
391127119442501	32	08-29-2006	Modern
391231119442901	238	10-15-2003	Modern
391231119442903	130	08-31-2006	Modern
391111119481901	117	07-07-2003	Modern
390637119472301	312	07-02-2003	Modern
390637119472303	120	07-02-2003	Modern
391014119450701	700	07-29-2008	Modern
Carson Valley			
385606119412201	245	07-15-2003	Modern
385304119460601	27	05-30-2003	Modern
385612119464101	20.5	05-30-2003	Modern
385655119413101	200	07-09-2003	Modern
385815119500301	16	05-01-2003	Modern
385816119482401	21	05-02-2003	Modern
390315119403201	64	07-15-2003	Modern

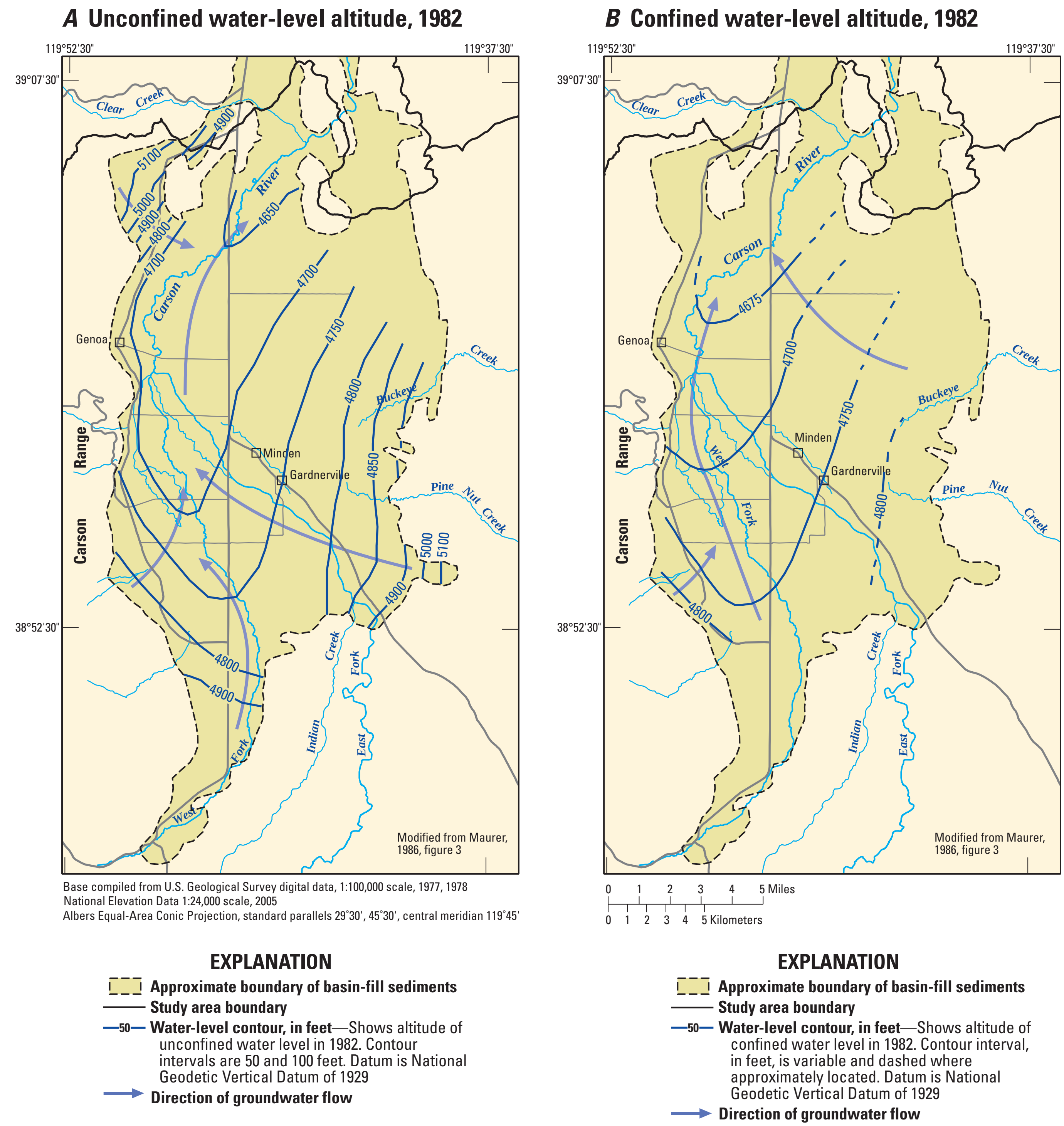


Figure 7. Water-level altitudes and general direction of groundwater flow under (*A*) unconfined conditions and (*B*) confined conditions in Carson Valley, Nevada and California, 1982.

Effects of Natural and Human Factors on Groundwater Quality

The occurrence and concentrations of contaminants in water within the basin-fill aquifer system in the Eagle and Carson Valleys are influenced by both natural and human-related factors. The movement of water through geologic materials of the basin coupled with movement of water from the land surface to the aquifer results in elevated concentrations of some constituents and compounds in groundwater. Water diverted from the Carson River, which enters the groundwater system by infiltration along irrigation canals and ditches and as excess irrigation water, as well as seepage from septic-tank systems, are new sources of recharge to the basin-fill aquifer that accompanied development. Although the shallow aquifer intercepts, stores, and transports some of this water, with a consequent increase in the concentration of nitrate and other dissolved constituents within that aquifer, the concern is for the deeper aquifer, which is a source of drinking-water supply in this growing residential area. Groundwater withdrawals also can induce the movement of poorer quality water laterally and from underlying strata into the area and depth interval of the basin-fill aquifer used for water supply in the valley.

The following description of groundwater quality in Eagle and Carson Valleys is based primarily on the results of analyses of samples collected from about 30 wells (shallower monitoring and domestic wells and deeper wells typically used for public supply) in each valley from 1987 to 1990 as part of the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program (fig. 8; Welch, 1994). Other data used in the interpretation of water quality were collected prior to the NAWQA sampling and can be found in Garcia (1989). A report by Schaefer and others (2007) focuses on the effect of urbanization on water quality in the principal aquifers in Eagle Valley.

General Water-Quality Characteristics and Natural Factors

Generally, the waters in the principal aquifers in Eagle and Carson Valleys are dilute, with dissolved-solid concentrations less than 1,000 mg/L, and are acceptable for drinking on the basis of standards set by the U.S. Environmental Protection Agency (2008; each time a drinking-water standard is mentioned in this section, it denotes this citation). The chemical characteristics of groundwater on the west side of Eagle and Carson Valleys most likely reflect the composition of the minerals in the igneous rocks and the natural geochemical reactions between the water and those minerals. Groundwater in an isolated area in northeastern Carson Valley has elevated concentrations of sulfate (greater than 50 mg/L) and fluoride (0.8 to 1.8 mg/L) and a higher proportion of sodium than does groundwater in the rest of the valley. This may be due to low-temperature reactions of the water with aquifer sediments derived from local metamorphic rocks that include marine evaporites containing gypsum.

Dissolved oxygen was detected at concentrations less than 1 mg/L in 6 of 37 (about 16 percent) wells on the western sides of Eagle and Carson Valleys, and in 9 of 18 wells (50 percent) on the eastern sides of the valleys (Welch, 1994, p. 43). The pH in groundwater in both valleys ranged from approximately 6.5 to greater than 8 pH units. Oxidation-reduction conditions in the basin-fill aquifer in Eagle Valley generally are controlled by the chemistry of the water entering the aquifer from the surrounding mountain blocks, with the most oxygenated water near recharge areas around the edges of the basin and less oxygenated water near the center of the basin (fig. 9; Schaefer and others, 2007). Chloride concentrations in groundwater along the Carson Range were lower (4 to 6 mg/L) than in water at sites farther east into the valleys (11 to 64 mg/L) (Welch, 1994, p. 41). This higher range in chloride to the east may be due to the interaction of groundwater with weathered granitic bedrock in that area.

Few groundwater samples collected in Carson and Eagle Valleys by NAWQA in 1988–89 exceeded the drinking-water standard of 30 µg/L for uranium (1 of 26 wells in Carson Valley and 4 of 23 wells in Eagle Valley) (Welch, 1994, table 11). The highest measured concentrations generally were along the western edges of Eagle and Carson Valleys. In these areas, uranium-222 seems to be concentrated on iron and manganese oxides that coat grains and fractures in granitic bedrock and in organic matter within the basin-fill sediments. Arsenic exceeded the drinking-water standard in less than 1 percent of samples collected from wells completed in the principal aquifer throughout Eagle and Carson Valleys (Welch, 1994, p. 58). Water samples from most of the sites exceeded the proposed drinking-water standard for radon of 300 pCi/L (97 of 103 sites; Welch, 1994, p. 72).

Potential Effects of Human Factors

Selected chemical constituents and organic compounds detected in groundwater in Eagle and Carson Valleys and the processes or sources that affect their presence and concentrations are summarized in table 4. Concentrations of dissolved solids in water in Eagle Valley's principal aquifer range from about 100 mg/L to more than 500 mg/L, with an average of about 270 mg/L (Anning and others, 2007). The use of treated sewage effluent to irrigate a golf course in the northeastern part of Eagle Valley has caused locally higher concentrations of dissolved solids in groundwater in that part of the valley (Anning and others, 2007). Sewage effluent used as recharge was found to be one of the most likely sources of groundwater contamination among all sources of recharge in Eagle Valley (Maurer and Thodal, 2000, p. 42).

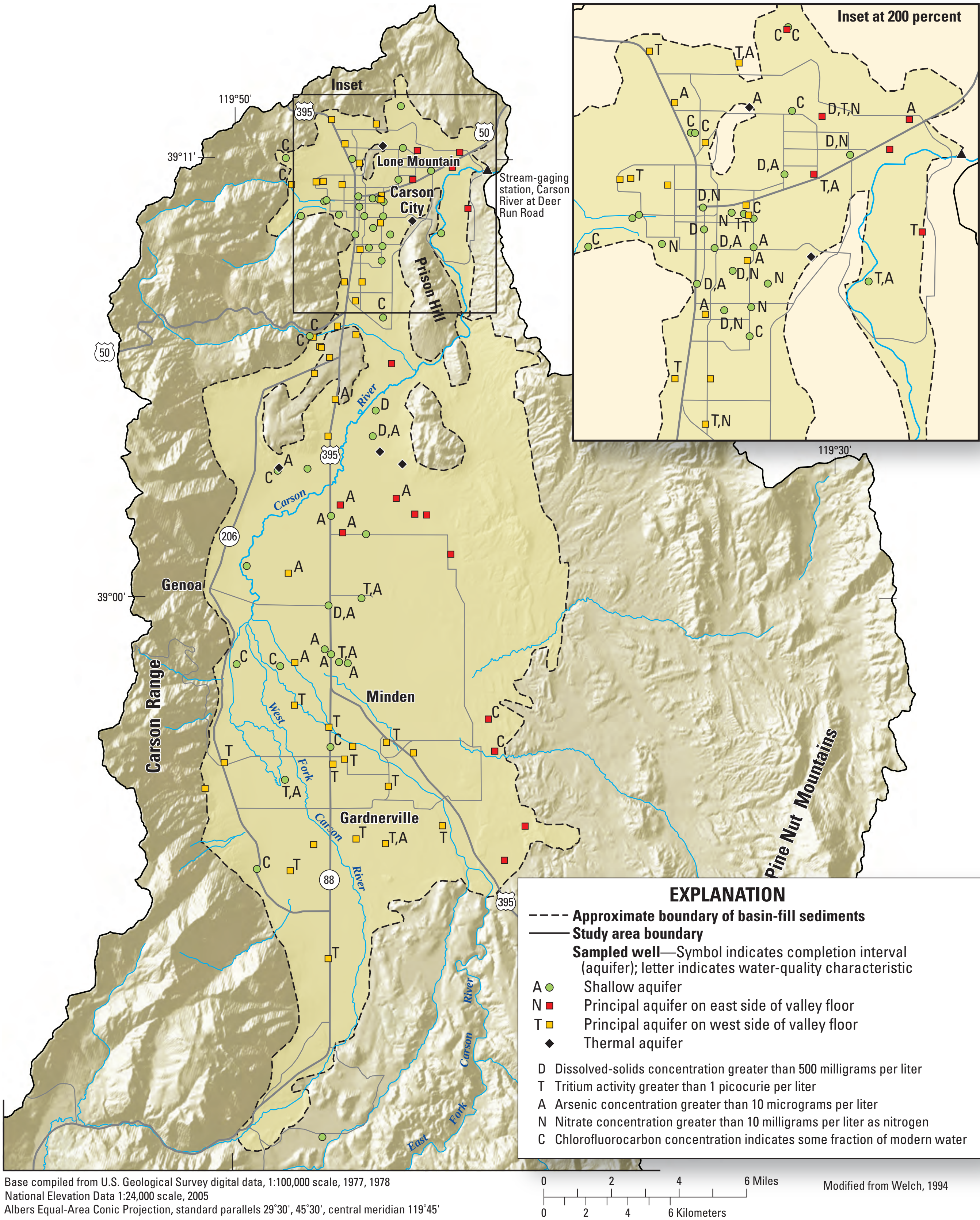


Figure 8. Location and completion interval (aquifer) of wells sampled in Eagle and Carson Valleys, Nevada, by the NAWQA Program.

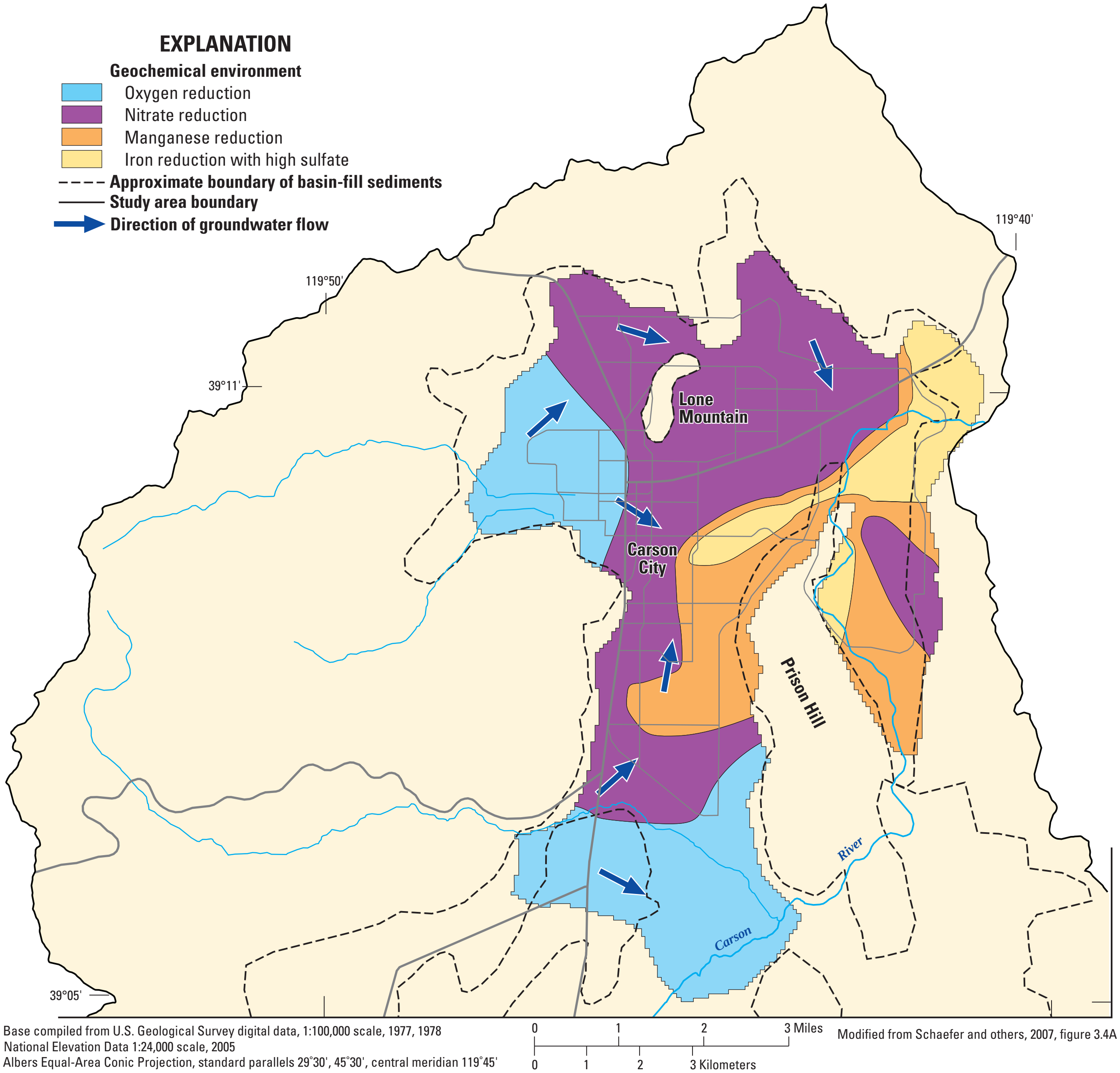


Figure 9. Oxidation-reduction classification zones in Eagle Valley, Nevada.

Table 4. Summary of selected constituents in groundwater in Eagle and Carson Valleys, Nevada, and sources or processes that affect their presence or concentration.

[All data from Welch (1994) unless otherwise noted. mg/L, milligrams per liter; n/a, not applicable]

Constituent	General location	Median value or detections	Possible sources or processes
		EAGLE VALLEY	
		Shallow aquifers	
Dissolved solids	Western and central basin	434 mg/L	Evapotranspiration and dissolution
Sulfate	Western basin	57 mg/L	Associated with altered consolidated rocks
Nitrate	West-central basin	0.17 mg/L	Treated wastewater, leaky sewer pipes, septic systems
Volatile organic compounds	Near urban areas	10	Point sources including underground gasoline tanks and solvents from repair shops and dry cleaners
Pesticides	Near irrigated land	9	Irrigated crop fertilizers
		Principal aquifers	
Dissolved solids	Eastern basin	160 mg/L	Evapotranspiration and dissolution
Sulfate	Eastern basin	10 mg/L	Associated with altered consolidated rocks
Nitrate	North-western basin	0.49 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems
Volatile organic compounds	Northern basin	[1]5	n/a
Pesticides	North-eastern basin	[1]2	n/a
		CARSON VALLEY	
		Shallow aquifers	
Dissolved solids	North-western basin	451 mg/L	Lawn irrigation, agricultural runoff, and sewage effluent
Sulfate	North-western basin	54 mg/L	Associated with altered consolidated rocks
Nitrate	North-western basin	0.36 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems
		Principal aquifers	
Dissolved solids	Eastern basin	179 mg/L	Evapotranspiration and dissolution
Sulfate	Eastern basin	25 mg/L	Associated with altered consolidated rocks
Nitrate	West-central basin	0.97 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems
Volatile organic compounds	n/a	0	n/a
Pesticides	n/a	0	n/a

[1] From Berris and others (2003).

Groundwater contamination as a result of human activity is more common (and commonly detected) in the shallow rather than the deeper (principal) aquifer, although nitrate concentrations exceeded the drinking-water standard in water from 3 percent of sampling sites (wells) in the principal aquifer throughout Eagle and Carson Valleys (Welch, 1994, p. 58). Those sites with elevated nitrate concentrations were in areas in which septic systems were in use and may have been leaking to deeper groundwater (Welch, 1994; Rosen, 2003).

Shallow aquifers in Eagle and Carson Valleys contained arsenic, fluoride, and nitrate concentrations that exceeded drinking-water standards, and concentrations of dissolved solids, iron, manganese, and sulfate all locally exceeded secondary drinking-water standards (Welch, 1994). The drinking-water standard for arsenic was exceeded in samples from 3 of 39 sampling sites, and the standards for fluoride and nitrate were exceeded in samples from 2 of 40 and 41 sites, respectively (Welch, 1994, p. 58–60). Manganese had the most common exceedance of the secondary standard of 0.1 mg/L, in samples from 21 of 40 sites, followed by iron, which exceeded the secondary standard of 0.6 mg/L in samples from 8 of 40 sites (Welch, 1994, p. 60). Elevated concentrations of manganese and iron may be a result of irrigation water wetting previously dry sediments that have oxide coatings. The rise in water level resulting from excess irrigation water may have allowed the dissolution of organic matter, which reacted with oxygen from the recharge water and in turn the oxide coating on the sediments.

Urban development in Eagle and Carson Valleys has been accompanied by an increase in use of, and amounts of, fertilizers, pesticides, and other manmade chemicals applied to the land. These chemicals can enter and degrade the quality of the shallow aquifer and move downward through the groundwater system, particularly in areas with shallow depth to water. Eagle Valley had 10 and 5 detections of a volatile organic compound (VOC) in water from shallow and deep wells, respectively, and 2 and 9 detections of a pesticide in water from shallow and deep wells, respectively (Berris and others, 2003). Volatile organic compounds were detected most frequently in wells near urban areas and pesticides in wells near irrigated areas. The most frequently detected VOC was trichloromethane, better known as chloroform. Chloroform, a byproduct of the reaction of organic material in source water with chlorine added during treatment, can potentially be found in groundwater as a result of infiltration of treated wastewater used to irrigate lawns and golf courses (Rosen and others, 2006). The herbicide atrazine and its degradation product, deethylatrazine, were the most frequently detected pesticide compounds. Atrazine is commonly used to control broadleaf and grassy weeds.

Summary

Eagle and Carson Valleys are hydraulically connected adjacent basins along the eastern front of the Sierra Nevada Range in northwestern Nevada and east-central California. The Carson River bisects Carson Valley from south to north and acts as a groundwater discharge zone for Eagle Valley as the river skirts its southern border. Precipitation that falls mostly as snow in the mountains recharges the basin-fill aquifers by infiltration within the mountain blocks and along the mountain fronts. Under natural conditions, groundwater discharges as evapotranspiration in the central part of the basins. The Carson River acts as both a source and a sink for groundwater in Carson Valley. In both valleys, clay lenses that commonly form confining layers are discontinuous and groundwater occurs under confined and unconfined conditions. Depth to water is typically deeper along the basin margins than near the basin center of the basin.

Both Eagle and Carson Valley have historically been agricultural basins, and although Carson Valley still supports agriculture, urban development has resulted in a reduction in irrigated acreage and a substantial increase in areas of impervious surfaces. Consequently, groundwater discharge by evapotranspiration has been reduced. Limited surface-water supplies have forced the use of groundwater as the main source of municipal supply and groundwater discharge in both valleys.

Water in the principal aquifers in Eagle and Carson Valleys is fairly dilute, and with few exceptions meets established quality standards for drinking water. The effects of urbanization on groundwater quality are most apparent in the shallow aquifer. Wastewater effluent from the Lake Tahoe basin is applied as irrigation water in Carson Valley and treated wastewater in Eagle Valley is used to irrigate golf courses and parks. Chlorine used in the treatment of wastewater can react with organic material in the source water to create chloroform before application to the land surface, and, as a result, chloroform is the most frequently detected volatile organic compound in samples of groundwater. Infiltration of treated wastewater has degraded the quality of water within the shallow aquifer, which poses the risk of consequent downward movement into the principal aquifer. Elevated levels of nitrate also were detected in water in the principal aquifers throughout Eagle and Carson Valleys in areas where septic systems were in use and may have been leaking to the deeper aquifers.

References Cited

Anning, D.W., Bauch, N.J., Gerner, S.J., Flynn, M.E., Hamlin, S.N., Moore, S.J., Schaefer, D.H., Anderholm, S.K., and Spangler, L.E., 2007, Dissolved solids in basin-fill aquifers and streams in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2006–5315, 168 p. Available at http://pubs.usgs.gov/sir/2006/5315/.

Arteaga, F.E., 1982, Mathmatical model analysis of the Eagle Valley ground-water basin, west-central Nevada: U.S. Geological Survey Open-File Report 80–1224, 55 p.

Arteaga, F.E., 1986, Mathematical model analysis of the Eagle Valley ground-water basin, west-central Nevada: State of Nevada Department of Conservation and Natural Resources Division of Water Resources Water Resources Bulletin 45. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Berger, D.L., and Medina, R.L., 1999, Spatial ground-water data base in Carson Valley, Douglas County, Nevada, and Alpine County, California, development and documentation: U.S. Geological Survey Water-Resources Investigations Report 99–4188.

Berris, S.N., Crompton, J.E., Joyner, J.D. and Ryan, R., 2003, Water resources data, Nevada, water year 2002: U.S. Geological Survey Water Data Report NV–02–1. Available at http://pubs.usgs.gov/wdr/WDR-NV-02-1/.

Covay, K.J., Banks, J.M., Bevans, H.E., Watkins, S.A., 1996, Environmental and hydrologic settings of the Las Vegas Valley area and the Carson and Truckee River basins, Nevada and California: U.S. Geological Survey Water-Resources Investigations Report 96–4087, 72 p.

Dangberg, G., 1972, Carson Valley: historical sketches of Nevada’s first settlement: Carson Valley Historical Society, Minden, Nevada, 144 p.

Garcia, K.T., 1989, Ground-water quality in Douglas County, western Nevada: U.S. Geological Survey Water-Resources Investigations Report 87–4269, 107 p.

Garcia, K.T., and Carman, R.L., 1986, Water quality characteristics and nutrient and suspended-sediment loads, Carson River and Truckee Canal, western Nevada, water year 1980: U.S. Geological Survey Water-Resources Investigations Report 85–4147, 107 p.

Glancy, P.A. and Katzer, T.L., 1976, Water-resources appraisal of the Carson River basin, western Nevada: U.S. Geological Survey Open-File Report 75–179, 177 p.

Jeton, A.E, and Maurer, D.K., 2007, Precipitation and runoff simulations of the Carson Range and Pine Nut Mountains, and updated estimates of ground-water inflow and the ground-water budgets for basin-fill aquifers of Carson Valley, Douglas County, Nevada, and Alpine County, California: U.S. Geological Survey Scientific Investigations Report 2007–5205, 57 p. Available at http://pubs.usgs.gov/sir/2007/5205/.

Lico, M.S., 1998, Quality of ground water beneath urban and agricultural lands in Las Vegas Valley and the Carson and Truckee River Basins, Nevada—Implications for water supply: U.S. Geological Survey Water-Resources Investigations Report 97–4259, 24 p.

Maurer, D.K, 1986, Geohydrology and simulated response to groundwater pumpage in Carson Valley, a river-dominated basin in Douglas County, Nevada, and Alpine County, California: U.S. Geological Survey Water-Resources Investigations Report 86–4328, 109 p.

Maurer, D.K. and Peltz, L.A., 1994, Potential for, and possible effects of, artificial recharge in Carson Valley, Douglas County, Nevada: U.S. Geological Survey Water-Resources Investigations Report 94–4126, 87 p.

Maurer, D.K., Berger, D.E., and Prudic, D.E., 1996, Subsurface flow to Eagle Valley from Vicee, Ash, and Kings Canyons, Carson City, Nevada, estimated from Darcy’s Law and the chloride-balance method: U.S. Geological Survey Water-Resources Investigations Report 96–4088, 74 p.

Maurer, D.K. and Berger, D.E., 1997, Subsurface flow and water yield from watersheds tributary to Eagle Valley hydrographic area, west-central Nevada: U.S. Geological Survey Water-Resources Investigations Report 97–4191, 56 p.

Maurer, D.K. and Thodal, C.E., 2000, Quantity and chemical quality of recharge, and updated water budgets, for the basin-fill aquifer in Eagle Valley, western Nevada: U.S. Geological Survey Water-Resources Investigations Report 99–4289, 46 p.

Maurer, D.K., Lopes, T.J., Medina, R.L. and Smith, J.L., 2004, Hydrogeology and hydrologic landscape regions of Nevada: U.S. Geological Survey Scientific Investigations Report 2004–5131, 41 p. Available at http://pubs.usgs.gov/sir/2004/5131/.

Maurer, D.K. and Berger, D.E., 2006, Water budgets and potential effects of land- and water-use changes for Carson Valley, Douglas County, Nevada, and Alpine County, California: U.S. Geological Survey Scientific Investigations Report 2006–5305, 64 p. Available at http://pubs.usgs.gov/sir/2006/5305/.

Maxey, G.B., and Eakin, T.E., 1949, Ground water in White River Valley, White Pine, Nye and Lincoln Counties, Nevada: Nevada State Engineer Water Resources Bulletin 8, 59 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239. Available at http://pubs.usgs.gov/sir/2008/5239/.

Moore, J.G., 1969, Geology and mineral deposits of Lyon, Douglas, and Ormsby Counties, Nevada: Nevada Bureau of Mines Bulletin 75, 45 p.

National Oceanic and Atmospheric Administration, 2002, Monthly station normals of temperature, precipitation, and heating and cooling degree days, 1971–2000, Climatology of the United States, No. 81, Nevada: National Climatic Data Center, Asheville, North Carolina, 26 p.

Nevada Division of Water Resources, 1999, Nevada State Water Plan, part 3, section 1, water supply and allocation, accessed February 28, 2010 at http://water.nv.gov/WaterPlanning/wat-plan/con-main.cfm.

Nevada State Demographer's Office, accessed September 11, 2006 at http://www.nsbdc.org/what/data_statistics/demographer/pubs/estimates/.

Oak Ridge National Laboratory, 2005, LandScan Global Population Database, accessed September 2006 at http://www.ornl.gov/landscan/.

Rosen, M.R., 2003, Trends in nitrate and dissolved-solids concentrations in ground water, Carson Valley, Douglas County, Nevada, 1985–2001: U.S. Geological Survey Water-Resources Investigation Report 03–4152, 6 p.

Rosen, M.R., Schaefer, D.H., Toccalino, P.L. and Delzer, G.C., 2006, Occurrence of anthropogenic compounds in ground water and finished water of community water systems in Eagle and Spanish Springs Valleys, Nevada, 2002–2004: U.S. Geological Survey Scientific Investigations Report 2006–5210, 31 p. Available at http://pubs.usgs.gov/sir/2006/5210/.

Schaefer, D.H., Green, J.M., and Rosen, M.R., 2007, Section 3. Hydrogeologic settings and ground-water flow simulations of the Eagle Valley and Spanish Springs regional study areas, Nevada, *in* Paschke, S.S., ed., Hydrogeologic settings and ground-water flow simulations for regional studies of the transport of anthropogenic and natural contaminants to public-supply wells—Studies begun in 2001: U.S. Geological Survey Professional Paper 1737–A, p. 3-1 to 3-33. Available at http://pubs.usgs.gov/pp/2007/1737a/.

Simpson, J.H., 1876, Report of Explorations Across the Great Basin in 1859: Government Printing Office, Washington D.C.

Stewart, J.H., 1980, Geology of Nevada: Nevada Bureau of Mines and Geology Special Publication 4, 135 p.

Thodal, C.E., 1996, Trends in chloride, dissolved-solids, and nitrate concentrations in ground water, Carson Valley and Topaz Lake areas, Douglas County, Nevada, 1959–88: U.S. Geological Survey Water-Resources Investigations Report 95–4254, p. 32.

Toll, D.W., 2008, Complete Nevada Traveler: Gold Hill Publishing Co., 256 p., accessed May 26, 2010 at http://www.nevadaweb.com/cnt/.

Trexler, D.T., 1977, Geologic map, Carson City quadrangle: Nevada Bureau of Mines and Geology Urban Map Series, Carson City folio, Map 1Ag, scale 1:24,000.

Trexler, D.T., Koenig, B.A., Flynn, T., and Bruce, J.L., 1980, Assessment of geothermal resources of Carson-Eagle Valleys and Big Smoky Valley, Nevada—First annual report, May 1, 1979–May 30, 1980: Nevada Bureau of Mines and Geology and University of Nevada, Reno, report no. DOE/NV/10039–2, 162 p.

Truner, R.M., and Bawic, W.J., 1996, Digital map of Nevada at scale of 1:500,000, digital dataset for Nevada geology.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed November 11, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed February 2007 at http://www.mrlc.gov/.

Welch, A.H., 1994, Ground-water quality and geochemistry in Carson and Eagle Valleys, western Nevada and eastern California: U.S. Geological Survey Open-File Report 93–33, 99 p.

Western Regional Climate Center, 2003, 1971–2000 Climate summary, accessed December 1, 2010 for Minden at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nv5191.

Worts, G.F. Jr. and Malmberg, G.T., 1966, Hydrologic appraisal of Eagle Valley, Ormsby County, Nevada, Water Resources—Reconnaissance Series, Report 39. Available at http://water.nv.gov/home/publications/recon.cfm.

This page intentionally left blank.

Section 5.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in Spanish Springs Valley, Nevada

By Jena M. Huntington

Basin Overview

Spanish Springs Valley is a relatively small basin about 5 mi northeast of Reno, Nevada within the Truckee River Basin (fig. 1) that is undergoing rapid population growth. The valley is bounded on the east by the Pah Rah Range, whose highest summit, Spanish Springs Peak, reaches an altitude of about 7,400 ft. To the west are Hungry Ridge and an unnamed extension that approaches an altitude of 6,000 ft. The northern border of the valley is a bedrock outcrop that creates a topographic divide less than 0.5 mi long between Hungry Ridge and the Pah Rah Range, while shallow bedrock marks the southern boundary (Berger and others, 1997). The drainage area for Spanish Springs Valley is about 77 mi^2, of which basin fill covers about 29 mi^2. The valley is about 11 mi long and 3 to 4 mi wide, and slopes from an altitude of about 4,600 ft in the north to about 4,400 ft in south.

Spanish Springs Valley has an arid to semiarid climate as a result of its location within the rain shadow of the Sierra Nevada Range. Summers are hot and dry, with daytime temperatures occasionally exceeding 100°F, and winters are cool, with temperatures sometimes falling below 0°F (Berger and others, 1997). Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 9 in. over the Spanish Springs Valley floor (McKinney and Anning, 2009). The surrounding mountains receive 9 to 11 in. of precipitation in an average year, and more than 13 in. may fall at the higher altitudes of the Pah Rah Range (Berger and others, 1997). There are no naturally perennial streams in the valley.

Rangeland covers much of Spanish Springs Valley, while only a small part is agricultural land (U.S. Geological Survey, 2003). The Orr Ditch, a diversion from the Truckee River used for irrigation, flows into the valley from the south and terminates near its center. Water from the diversion, combined with minor amounts of water from springs and wells, irrigated about 550 acres of agricultural land in 2001, primarily alfalfa and pasture. Irrigation return flow and some groundwater discharge is collected in the North Truckee Drain and returned to the Truckee River in the Truckee Meadows basin to the south.

In 2008, the population of Spanish Springs Valley was calculated as about 47,000 within the alluvial basin, of which about 18.5 mi^2, or about 23 percent, was residential land (Christian Kropf, Washoe County Department of Water Resources, written commun., 2009). Groundwater pumped from the basin-fill aquifer is an important source of drinking water in the valley, although plans are for future population growth to be supported by imported water from the Truckee River (Truckee Meadows Water Authority, 2004).

Infiltration from septic-tank systems has become a source of groundwater recharge in some residential areas in the valley as more than 2,000 systems were installed from the early 1970s to 1995 (Rosen and others, 2006a); in 2009, more than 2,300 such systems were in use (Christian Kropf, written commun., 2009). The Nevada Division of Environmental Protection has issued directives to ensure that existing homes currently on septic systems and new homes in the valley be connected to centralized sewage disposal systems because of increasing nitrate concentrations in groundwater (Rosen and others, 2006a). Elevated concentrations of nitrate in groundwater are an important water-quality concern for the valley.

Water Development History

Spanish Springs Valley was named after several springs on the south central part of the valley floor (fig. 1). A land survey in 1872 noted that the main spring area was about 66 ft long by 33 ft wide and was surrounded by smaller springs (Berger and others, 1997). Early agricultural activity in the valley used the water from these springs and from shallow flowing wells for irrigation. The amount of irrigated land increased in the southern part of the valley after construction of the Orr Ditch in 1878. Agricultural land use within the area serviced by the Orr Ditch has remained relatively unchanged based on comparisons of aerial photographs taken in 1956, 1977, and 1994 and of assessor parcel maps, although since 1994, agricultural acreage has decreased as new homes have been built in the southwestern part of the valley (Berger and others, 1997).

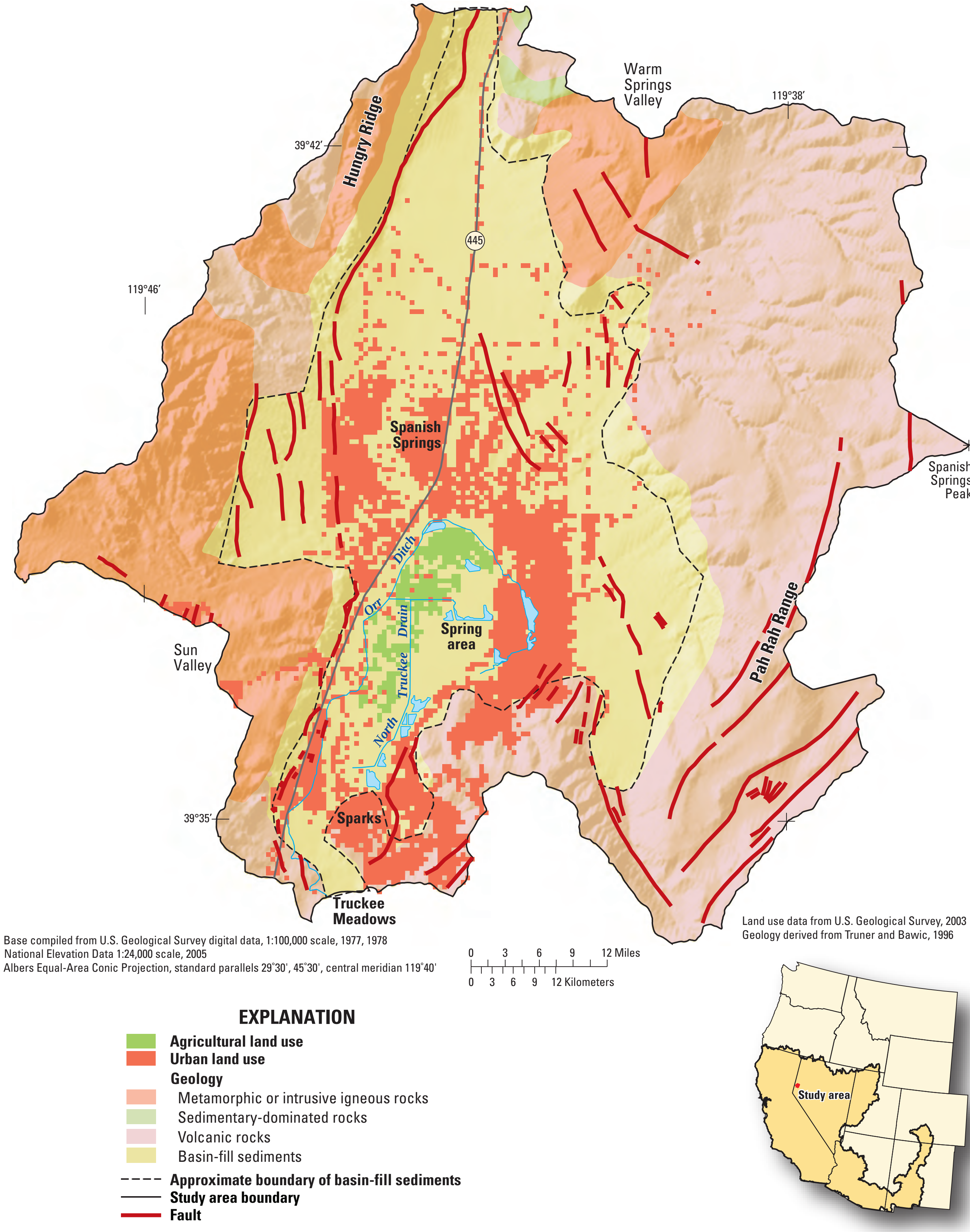


Figure 1. Physiography, land use, and generalized geology of Spanish Springs Valley, Nevada.

Groundwater was used primarily for agriculture until 1983, when it accounted for about half of the total amount pumped (Berger and others, 1997, table 10). Urban development has increased significantly since the late 1970s due to the proximity of Spanish Springs Valley to the expanding Reno-Sparks metropolitan area to the south. The addition of residential subdivisions and mobile home parks sharply increased the valley population of 790 in 1979 to 9,320 in 1994 (Berger and others, 1997, table 10), mostly in the central and southeastern parts of the valley. Homes also are now scattered to the north and near the mountain fronts with population estimates nearing 50,000. Because groundwater is the primary source for public and domestic supply in the valley, its use has increased with population growth. Depths of supply wells range from 200 ft to more than 800 ft, and depths to water range from 20 ft to nearly 200 ft below land surface (Christian Kropf, written commun., 2009).

Hydrogeology

Present-day topographic features, including the structural depressions that underlie Spanish Springs Valley, were formed by extensional faulting that began in the middle to late Tertiary period. The mountains surrounding the valley are composed of Mesozoic-age granitic and metamorphic rocks overlain by Tertiary-age volcanic rocks than contain lenses of sedimentary rocks (fig. 1). These consolidated rocks commonly have low porosity and permeability except were fractured and faulted. Although the volcanic rocks are mainly tuffs and volcanic flows and have little to no interstitial porosity, the interbedded sedimentary rocks are mostly fine-grained, partly consolidated lacustrine deposits with low permeability that may store moderate amounts of water. Connection between the basin-fill deposits and underlying consolidated rocks is suggested by an upward hydraulic gradient in the southeastern part of the valley (Berger and others, 1997).

Erosion from the surrounding mountains during Quaternary time was accompanied by the filling of the valley with interbedded, unconsolidated deposits of sand, gravel, clay and silt. The basin-fill deposits are thickest, at least 1,000 ft thick, on the western side of Spanish Springs Valley along a northeast trending trough-like feature, and become thinner to the east, based on geophysical data and drillers' logs (Berger and others, 1997; Makowski, 2006). The deposits are less than 50 ft thick along the topographic divide that forms the northern boundary of the valley. The basin-fill deposits in the southern part of Spanish Springs Valley are less than 100 ft thick and become less than 20 ft thick along the southern boundary with Truckee Meadows (Berger and others, 1997).

Conceptual Understanding of the Groundwater System

The basin-fill aquifer in Spanish Springs Valley is under mostly unconfined or water-table conditions. Although the basin is topographically closed and has a playa in the central part, the groundwater system is considered to be open, with subsurface outflow at both the northern and sourthern ends (fig. 2). The aquifer is recharged naturally by the infiltration of precipitation falling on the basin margins and on the surrounding mountains, and from human-related sources in the valley such as imported surface water, excess irrigation water, and effluent from septic-tank systems (fig. 2).

Basin-fill deposits originating from volcanic rocks are generally fine grained, and thus have lower hydraulic conductivity than coarser grained deposits derived from granitic rocks. In a groundwater flow model of the basin-fill aquifer, the top layer of the model, representing the upper 330 ft of saturated deposits, was assigned values of hydraulic conductivity ranging from less than 0.03 to 30 ft/d (Schaefer and others, 2007). In most of the central part of the valley, however, the top layer of the model was assigned a hydraulic conductivity of less than 3 ft/d.

Water Budget

Prior to any groundwater development in Spanish Springs Valley, the basin-fill aquifer was recharged by precipitation falling on the surrounding mountains that moved into the basin fill through subsurface fractures or by the infiltration of runoff at the mountain front (fig. 3*A*). Berger and others (1997) estimated this mountain-front recharge at about 830 acre-ft/yr using the Maxey-Eakin method (Maxey and Eakin, 1949 and Eakin and others, 1951), which applies a percentage of average annual precipitation within specified altitude zones to estimate recharge.

Rush and Glancy (1967, table 20) estimated that recharge to the valley under natural (predevelopment) conditions was about 1,000 acre-ft/yr, based on the assumption that the groundwater system was in equilibrium. They estimated groundwater discharge by evapotranspiration prior to construction of the Orr Ditch to be about 900 acre-ft/yr (table 1) (Rush and Glancy, 1967, table 14). Groundwater from Spanish Springs Valley may flow south to Truckee Meadows through a thin layer of basin-fill deposits or through the underlying fractured bedrock (Berger and others, 1997).

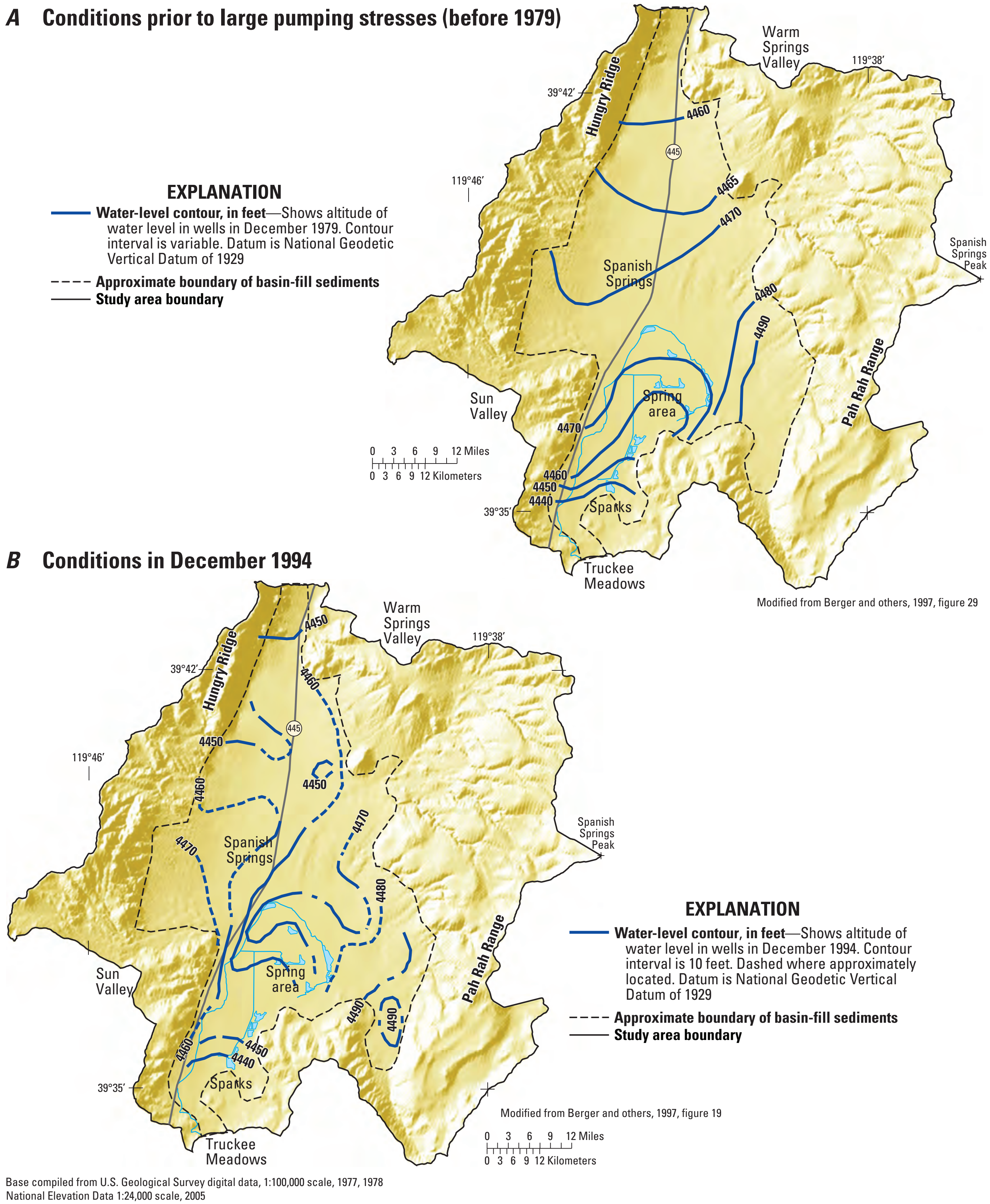


Figure 2. Groundwater levels assumed to represent conditions in Spanish Springs Valley, Nevada (*A*) prior to large pumping stresses (before 1979) and (*B*) in December 1994.

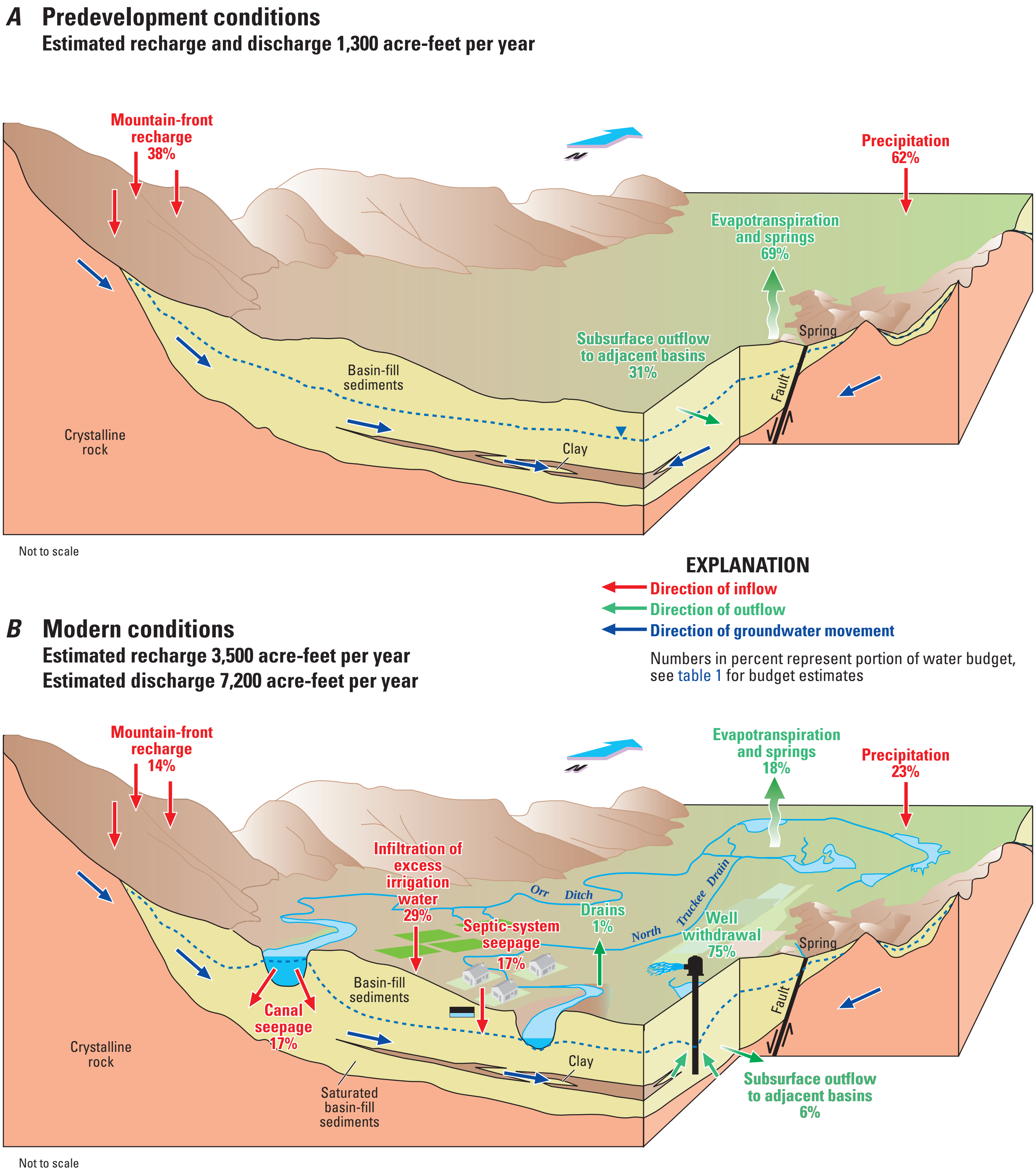


Figure 3. Generalized diagrams for Spanish Springs Valley, Nevada, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin-fill aquifer in Spanish Springs Valley, Nevada, under predevelopment and modern conditions.

[All values are in acre-feet per year and are rounded to the nearest hundred. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Mountain-front recharge	[1]500	[1]500	0
Infiltration of precipitation on alluvial basin	[2]800	[2]800	0
Canal seepage	0	[2]600	600
Infiltration of excess irrigation water	0	[2]1,000	1,000
Septic-system seepage	0	[6]600	600
Total recharge	**1,300**	**3,500**	**2,200**
Budget component		**Estimated discharge**	
Evapotranspiration and springs	[3] 900	[2]1,300	400
Well withdrawals	0	[5]5,400	5,400
Drains	0	[2]100	100
Subsurface outflow to south	[3]100	[3]100	0
Subsurface outflow to north	[4]300	[4]300	0
Total discharge	**1,300**	**7,200**	**5,900**
Change in storage (total recharge minus total discharge)	0	-3,700	-3,700

[1] Assumed to equal estimated residual between predevelopment recharge and discharge.

[2] Estimated for 1994 conditions by Berger and others (1997).

[3] Rush and Glancy (1967, table 14).

[4] Hadiaris (1988).

[5] Lopes and Evetts (2004, table 1).

[6] Rosen and others (2006a, p. 10)

Rush and Glancy (1967) used Darcy's Law to estimate about 100 acre-ft/yr of subsurface outflow to the south. A hydraulic gradient between Spanish Springs Valley and Warm Springs Valley may allow subsurface outflow to the north (Berger and others, 1997). About 280 acre-ft/yr was simulated in a steady-state flow model as subsurface outflow through the northern boundary (Hadiaris, 1988), and about 170 acre-ft was simulated under 1994 conditions (Berger and others, 1997, table 11). For this report, total recharge to the basin-fill aquifer in Spanish Springs Valley under predevelopment conditions is assumed to have been in equilibrium with natural discharge through evapotranspiration and estimated subsurface outflow to adjacent basins, and is estimated to have been about 1,300 acre-ft/yr (table 1).

The groundwater budget for Spanish Springs Valley changed with construction of the Orr Ditch in 1878 and the expansion of residential development since about 1979. Many of the estimates of recharge and discharge for the valley presented in this report are for conditions studied in 1994 by Berger and others (1997). In that year transmission losses from the 7-mile long, unlined Orr Ditch locally recharged an estimated 590 acre-ft of Truckee River water to shallow parts of the basin-fill aquifer. In basins similar to Spanish Springs Valley, about 40 percent of applied irrigation is assumed to infiltrate far enough to reach the groundwater system. Therefore, assuming that 40 percent of the water applied for irrigation infiltrates to the water table, about 860 acre-ft of water applied for irrigation from the Orr Ditch recharged shallow parts of the aquifer in 1994 (Berger and others, 1997, p. 47). Recharge from excess (unconsumed) groundwater applied for irrigation outside the area encompassed by the Orr Ditch was estimated to be about 170 acre-ft (Berger and others, 1997, p. 50). Precipitation in 1994 was below normal, resulting in less than the usual amount of water being diverted to the Orr Ditch. More recharge (than in 1994) to the shallow groundwater system from canal seepage and from excess irrigation water likely occurs during average and above average precipitation conditions.

Rapid population growth in Spanish Springs Valley resulted in a large increase in the use of individual septic-tank systems. Seepage from septic system leach fields was estimated to be 75 percent of the total amount of water delivered to homes during the winter months (Berger and others, 1997, p. 50). This equated to about 450 acre-ft of seepage in 1994. Rosen and others (2006a, p. 10) used an estimate of 227 gallons per day for septic tank discharge per household. With continued residential development in the valley, more than 2,300 homes now use septic systems (Rosen and others, 2006a, p. 3), and an estimated 585 acre-ft of seepage from septic-tank systems enters the basin-fill aquifer each year. Residential developments built since 2006 are connected to centralized sewage disposal systems (Joseph Stowell, Washoe County Department of Water Resources, written commun., 2009).

Water pumped from wells increased from about 500 acre-ft in 1979 to 2,600 acre-ft in 1994 (Berger and others, 1997, table 10). About 5,400 acre-ft was pumped in 2000; 56 percent from domestic wells, 43 percent from public-supply wells, and only one percent from irrigation wells (Lopes and Evetts, 2004, table 1). The 6 public-supply wells completed in basin-fill deposits and pumped in 2007 (Washoe County Department of Water Resources, 2008), are located within or near residential areas to the west and north of the Orr Ditch. Prior to residential development, these areas were rangeland, in which the only source of groundwater recharge was from the surrounding mountains. When wells are pumped, the natural directions of groundwater flow near these wells are likely affected, and some water recharged by losses from the Orr Ditch or from excess irrigation water may be intercepted by the wells nearest to the ditch.

Recharge to the shallow part of the basin-fill aquifer from canal seepage and infiltration of excess irrigation water has been accompanied by discharge to the North Truckee Drain and an increase in discharge through evapotranspiration compared to predevelopment conditions (table 1). The difference between estimated recharge and discharge under conditions in 1994, or the amount removed from aquifer storage, is about 3,700 acre-ft/yr. Most of the discharge from the basin-fill aquifer under modern conditions is from wells at least 200 ft deep, whereas most of the recharge is now to shallower parts of the aquifer system. The vertical connection between shallow and deep parts of the aquifer is dependent on the confining layers separating them and the hydraulic gradient between them, although there is little evidence to support laterally extensive confining layers in Spanish Springs Valley.

Groundwater Movement

Under predevelopment conditions, groundwater flowed predominantly from the west and east toward the center of the valley where it is was discharged by evapotranspiration and to springs (fig. 2*A*). North of the Orr Ditch, water also flowed north where it discharged from the valley into an adjacent basin. Following basin development, groundwater flows predominantly to the North Truckee Drain and municipal wells, although some water continues to flow north away from the influence of the Orr Ditch and pumping centers (fig. 2*B*).

Stable-isotope data indicate that Truckee River water moving into the aquifer by infiltration from the Orr Ditch is more enriched in the heavier isotopes of oxygen and hydrogen than is groundwater recharged near the mountain fronts (Berger and others, 1997, fig. 16). Water sampled from a 193-ft deep well about 2,000 ft north of the Orr Ditch had a tritium activity of 10 pCi/L, implying that a component of the water was recharged since about 1958 (Welch, 1994, p. 16; Berger and others, 1997, p. 41). Stable isotope analysis indicates that this water consists of about 35 percent natural recharge and 65 percent Truckee River water from the Orr Ditch (Berger and others, 1997, p. 41), which supports the assumption that some imported Truckee River water flows northward from the Orr Ditch.

Concentrations of stable isotopes in water collected from a well just south of the Spanish Springs Valley border, near the North Truckee Drain, indicate a mixture of natural recharge and Truckee River water (Berger and others, 1997, p. 41). Tritium activities measured in water sampled from wells deeper than 150 ft throughout the valley were less than 1 pCi/L, with the exception of the 2 wells mentioned previously that contain a component of Truckee River water and 3 other wells near the Orr Ditch. These data indicate that precipitation recharged after 1952 (the beginning of above-ground nuclear testing) has not yet reached deeper groundwater in the vicinity of these wells (Welch, 1994, p. 16). Water samples collected from shallow observation wells in the area irrigated with water from Orr Ditch had tritium activites greater than 20 pCi/L, indicative of modern water, or water recharged after 1952. Chloroflourocarbons (CFCs) were analyzed in samples collected in 1993–94 from 19 wells throughout Spanish Springs Valley (Berger and others, 1997). All 19 wells contained concentrations of CFCs such that each has a fraction of modern water.

Effects of Natural and Human Factors on Groundwater Quality

The general water-quality characteristics as well as the occurrence and concentrations of individual chemical constituents and organic compounds of groundwater in Spanish Springs Valley varies areally across the basin. Additional recharge to the basin-fill aquifer system has affected groundwater quality in parts of the basin. Surface water diverted from the Truckee River for irrigation and groundwater pumped for domestic and public supply are now sources of recharge in parts of the valley through water transported in the Orr Ditch, infiltration of excess irrigation water, and septic-tank effluent. Although the shallow part of the groundwater system has received most of this additional recharge water, with a consequent increase in concentrations of dissolved solids and nitrate, the concern is for the deeper aquifer that is the source of drinking-water supply in this growing residential area.

The chemical characteristics of groundwater in the valley were evaluated on the basis of data collected from 22 wells sampled as part of a study on the groundwater flow system in 1993–94 (Berger and others, 1997; data in Emett and others, 1994, p. 557 and Clary and others, 1995, p. 732) and data from 8 water-supply wells sampled as part of the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program in 2002–03 (data in Stockton and others, 2003) (fig. 4). Results of analyses of water from five of the NAWQA sampled wells were used to characterize the occurrence and concentrations of anthropogenic organic compounds in water used for public supply in the valley (Rosen and others, 2006b). Additional groundwater sampling was completed as part of studies to 1) identify sources of nitrate to the groundwater system (Seiler, 2005) and 2) determine the amount of nitrogen entering groundwater from septic-tank systems in the valley (Rosen and others, 2006a; data in Bonner and others, 2004).

General Water-Quality Characteristics and Natural Factors

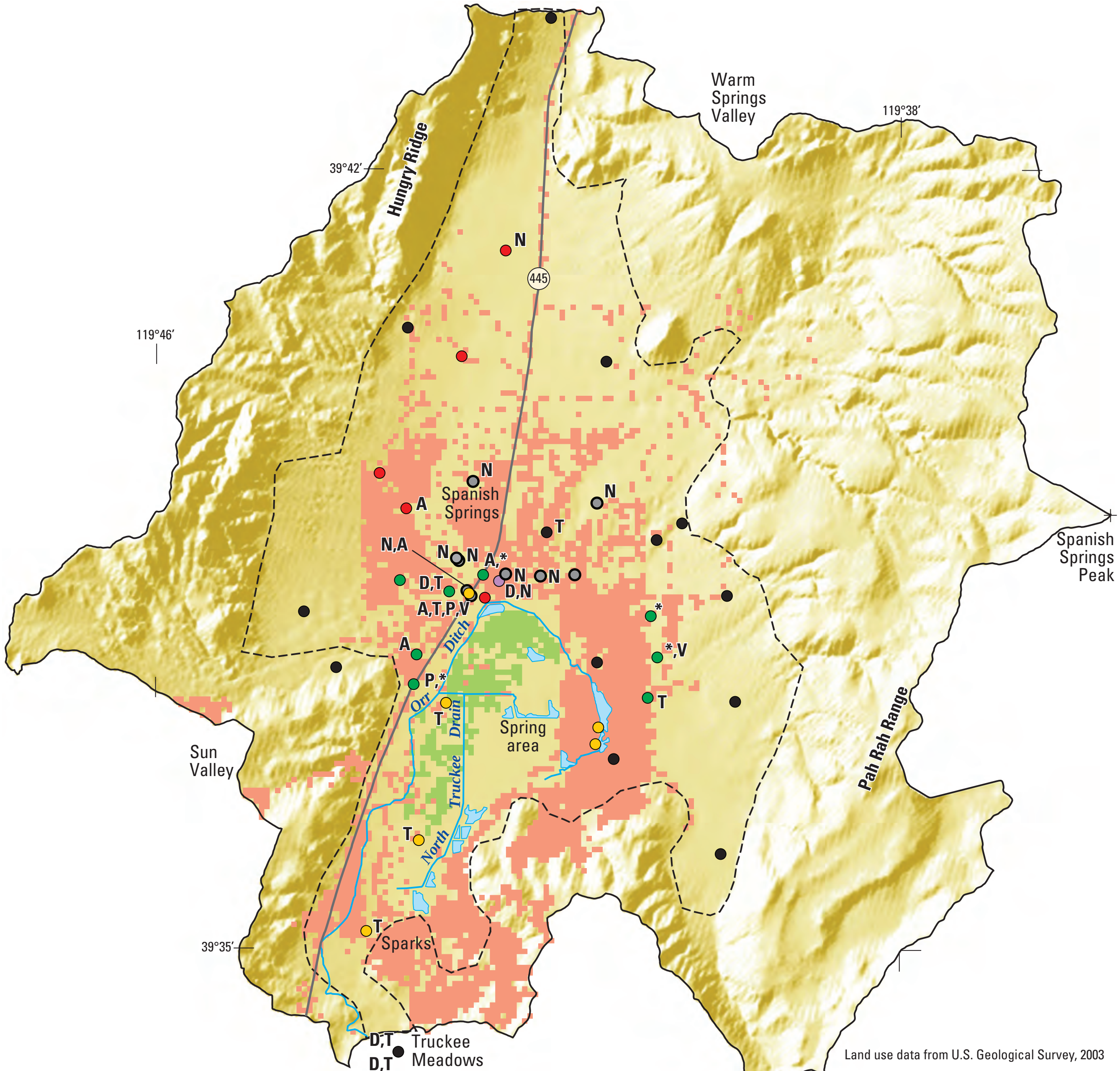
Water in the basin-fill aquifer system in Spanish Springs Valley is primarily a sodium-bicarbonate/calcium-bicarbonate type. Waters sampled from 2 wells completed in fractured bedrock in the southern part of the valley, however, are a sodium-sulfate type, which likely reflects the mineral composition of the volcanic rocks in the area. Analyses of samples of groundwater collected during several studies in the valley indicate the following general properties and chemical characteristics:

- The pH of the water in 22 wells sampled by Berger and others (1997) ranged from 7.0 to 9.0 with a median value of 7.8; dissolved-oxygen concentrations of these samples ranged from 0.1 to 8.6 mg/L with a median value 2.8 mg/L.
- Dissolved-solids concentrations in the 22 wells sampled by Berger and others (1997) ranged from 155 to 2,680 mg/L with a median value of 237 mg/L. Dissolved-solids concentrations were generally less than 200 mg/L in water from wells in the spring area and the area to the east, less than 260 mg/L in water from wells in the northern and western part of the valley, and greater than 350 mg/L in water from wells near the Orr Ditch.
- Oxidation-reduction (redox) conditions in the aquifer are primarily oxygen reducing (Schaefer and others, 2007, fig. 3.4*B*).
- Arsenic concentrations in water from the eight supply wells sampled in the NAWQA study ranged from 1.5 to 18.8 µg/L (Stockton and others, 2003). In samples from 3 of these wells, the maximum contaminant level (MCL) for arsenic in drinking water of 10 µg/L was exceeded (U.S. Environmental Protection Agency, 2008; each time "MCL" is mentioned in this section, it denotes this citation).

Potential Effects of Human Factors

Recharge associated with human activities accounts for more than half of Spanish Springs Valley's groundwater budget, while pumping from wells is now the primary path of discharge from the basin-fill aquifer (table 1). Pumping and the resulting changes in hydraulic gradients can cause changes in groundwater quality by enhancing the downward movement of shallow groundwater and the vertical or lateral movement of water from adjacent consolidated rock to parts of the basin-fill aquifer used for water supply.

Human activities in the rapidly expanding residential area in the valley have the potential to release chemicals that may ultimately reach supply wells (table 2). Analysis of water samples collected from five public-supply wells in the valley in 2002–03 showed the presence of anthropogenic organic compounds in three wells (Rosen and others, 2006b). The herbicide atrazine and its degradation product deethylatrazine were detected in one well at concentrations less than 0.05 µg/L, much less than the MCL of 3 µg/L for atrazine. The volatile organic compounds (VOCs) chloroform and bromoform (both disinfection byproducts of the water treatment process) were detected in water from one well each, also at very low concentrations, well below the MCL of 80 µg/L for both compounds (each detection was less than 0.05 µg/L; Rosen and others, 2006b, table 3). Although the concentrations of these compounds were not a health concern, their presence in the aquifer indicates the potential for their movement from the land surface and the possibility that higher concentrations may occur in the future.



Base compiled from U.S. Geological Survey digital data, 1:100,000 scale, 1977, 1978
National Elevation Data 1:24,000 scale, 2005
Albers Equal-Area Conic Projection, standard parallels 29°30′, 45°30′, central meridian 119°40′

EXPLANATION

- **Agricultural land use**
- **Urban land use**
- **Approximate boundary of basin-fill sediments**
- **Study area boundary**

Well—label represents water-quality constituent detected

- D,T ● 1993–94 sample, well depth greater than 100 feet
- T ● 1993–94 sample, well depth less than 100 feet or unknown
- *,P ● 1998 sample, well depth greater than 100 feet
- D,N ● 1998 sample, well depth less than 100 feet
- T ● 2002–03 sample, well depth greater than 100 feet
- N ● 2004–05 sample, well depth less than 100 feet

- **D** Dissolved-solids concentration greater than 500 milligrams per liter
- **V** One or more volatile organic compounds detected
- **P** One or more pesticides detected
- **T** Tritium activity greater than 1 picocurie per liter. Not analyzed for in wells labeled * or in 1998 samples.
- **A** Arsenic concentration greater than 10 micrograms per liter
- **N** Nitrate concentration greater than 10 milligrams per liter as nitrogen. Not analyzed for in the 1994 samples

Figure 4. Location and depths of wells, and chemical characteristics of groundwater sampled in various studies from 1993 to 2005, Spanish Springs Valley, Nevada.

Table 2. Summary of selected constituents in groundwater in Spanish Springs Valley, Nevada, and sources or processes that affect their presence or concentration.

[mg/L, milligrams per liter; n/a, not applicable]

Constituent	General location	Median value or detections	Possible sources or processes
		Shallow aquifers	
Dissolved solids	Central basin near irrigated land & North Truckee Drain	362 mg/L	Excess irrigation water infiltrating through canals and (or) fields
Sulfate	Central basin near irrigated land & North Truckee Drain	24 mg/L	Associated with altered consolidated rocks
Nitrate	Near urban areas	[1]12.6 mg/L	Wastewater infiltrating from septic systems
Volatile organic compounds	Central basin	[2]5	Chlorinated municipal-supply water infiltrating through irrigated yards/turf areas
Pesticides	n/a	n/a	n/a
		Principal aquifers	
Dissolved solids	Basin wide	216 mg/L	Evapotranspiration of shallow groundwater and dissolution
Sulfate	Basin wide	25 mg/L	Associated with altered consolidated rocks; excess irrigation water and canal infiltration
Nitrate	Basin wide	[3]4.1 mg/L	Wastewater infiltrating from septic systems
Volatile organic compounds	Central basin	[3]2	Chlorinated municipal-supply water infiltrating through irrigated yards/turf areas
Pesticides	Western near highway	[3]1	Excess irrigation water infiltrating through canals and (or) fields

[1] Bonner and others (2004).

[2] Michael Rosen, U.S. Geological Survey, written commun., July 22, 2009.

[3] Rosen and others (2006b).

Nitrate concentrations as nitrogen in some public-supply wells in Spanish Spring Valley are approaching the MCL of 10 mg/L. Nitrate concentrations in the five public-supply wells sampled range from 2.3 to 8.1 mg/L, although background concentrations in the aquifer are assumed to be less than about 2 mg/L (Rosen and others, 2006a, p. 8). Elevated concentrations of nitrate have been attributed to the increased use of septic-tank systems accompanying residential development in the valley rather than to the use of fertilizers (Seiler, 1999; Seiler, 2005).

Nitrate concentrations as nitrogen in water sampled from shallow wells (45 to 120 ft deep) in the Spanish Springs Valley ranged from 4.1 to 38.5 mg/L with a median value of 12.6 mg/L (data in Bonner and others, 2004). The median concentration of total dissolved nitrogen in more than 300 soil-water samples collected within the soil zone under four septic tank leach fields in residential areas north of the Orr Ditch was 44 mg/L (Rosen and others, 2006a). The concentration of total dissolved nitrogen in recharge water potentially reaching the water table ranged from 25 to 29 mg/L. Therefore, on the basis of mass-balance calculations, approximately 29 to 32 metric tons of nitrogen are contributed to the shallow groundwater from septic-tank systems and natural recharge each year; almost all of this nitrogen originates within the septic-tank systems.

Summary

The basin-fill aquifer underlying Spanish Springs Valley is a complex system that has undergone many changes during the basin's development. Generally under unconfined conditions, the aquifer is recharged naturally by the infiltration of precipitation falling on the basin margins and on the surrounding mountains. Estimated natural recharge to the aquifer (predevelopment) is assumed to equal estimated natural discharge from evapotranspiration and subsurface outflow to adjacent basins, about 1,300 acre-ft/yr.

The groundwater budget for Spanish Springs Valley changed with construction of the Orr Ditch in 1878 and the expansion of residential development since about 1979. Human-influenced sources of recharge to the groundwater system in the valley are imported surface water, excess irrigation water, and septic-tank system effluent. Continued residential development in the valley has resulted in more than 2,300 homes with septic systems with an estimated 585 acre-ft/yr of seepage from septic-tank systems to the basin-fill aquifer. New residential developments are connected to sewage systems.

Groundwater pumped from the basin-fill aquifer in Spanish Springs Valley is the primary source of drinking water, although plans are for future population growth to be supported by imported water from the Truckee River. Pumpage from wells increased from about 500 acre-ft in 1979 to 2,600 acre-ft in 1994. About 5,400 acre-ft was pumped in 2000, 56 percent from domestic wells, 43 percent from public-supply wells, and only one percent from irrigation wells. Much of the discharge from the basin-fill aquifer under modern conditions is from wells at least 200 ft deep, whereas much of the recharge is now to shallower parts of the aquifer system. The vertical connection between shallower and deeper parts of the aquifer is dependent on the potential confining layers separating them and the hydraulic gradient between them.

The additional anthropogenically derived recharge to the basin-fill aquifer system has affected groundwater quality in parts of Spanish Springs Valley. Pumping and the resulting changes in hydraulic gradients can cause changes in groundwater quality by enhancing the downward movement of shallow groundwater and the vertical or lateral movement of water from adjacent consolidated rock to parts of the basin-fill aquifer used for water supply. Concentrations of dissolved solids in the water samples ranged from 155 to 2,680 mg/L with a median value of 237 mg/L. Analysis of water samples collected from five public-supply wells in the valley in 2002–03 showed the presence of volatile organic compounds in two of the wells. Although the concentrations of these compounds were not a health concern, their presence in the aquifer indicates the potential for movement from the land surface and the possibility that higher concentrations may occur in the future.

Infiltration from septic-tank systems has become a source of groundwater recharge in some residential areas in the valley and elevated nitrate concentrations in groundwater from septic-tank effluent is an important water-quality concern. Nitrate concentrations as nitrogen in some public-supply wells are approaching the drinking-water standard of 10 mg/L. Increasing nitrate concentrations have been attributed to the increased use of septic-tank systems rather than to the use of fertilizers.

References Cited

Berger, D.L., Ross, W.C., Thodal, C.E., and Robledo, A.R., 1997, Hydrogeology and simulated effects of urban development on water resources of Spanish Springs Valley, Washoe County, west-central Nevada: U.S. Geological Survey Water-Resources Investigations Report 96–4297, 80 p.

Bonner, L.J., Evetts, D.M., Swartwood, J.R., and Wilson, J.W., 2004, Water resources data, Nevada, water year 2004: U.S. Geological Survey Water-Data Report NV–04–1. Available at http://pubs.usgs.gov/wdr/wdr-nv-04-1/.

Clary, S.L., McClary, D.R., Whitney, Rita, and Reeves, D.D., 1995, Water resources data, Nevada, water year 1994: U.S. Geological Survey Water-Data Report NV–94–1, 768 p.

Eakin, T.E., Maxey, G.B., Robinson, T.W., Fredericks, J.C., and Loeltz, O.J., 1951, Contributions to the hydrology of eastern Nevada: Nevada State Engineer, Water Resources Bulletin 12, 171 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Emett, D.C., Hutchinson, D.D., Jonson, N.A., and O'Hair, K.L., 1994, Water resources data, Nevada, water year 1993: U.S. Geological Survey Water-Data Report NV–93–1, 596 p.

Hadiaris, A.K., 1988, Quantitative analysis of groundwater flow in Spanish Springs Valley, Washoe County, Nevada: University of Nevada, Reno, unpublished M.S. thesis, 202 p.

Lopes, T.J., and Evetts, D.M., 2004, Ground-water pumpage and artificial recharge estimates for calendar year 2000 and average annual natural recharge and interbasin flow by hydrographic area, Nevada: U.S. Geological Survey Scientific Investigations Report 2004–5239, 88 p. Available at http://pubs.usgs.gov/sir/2004/5239/.

Makowski, Anna, 2006, Modeling nitrate transport in Spanish Springs Valley, Washoe County, Nevada: University of Nevada, Reno, M.S. thesis.

Maxey, G.B., and Eakin, T.E., 1949, Ground water in White River Valley, White Pine, Nye, and Lincoln Counties, Nevada: Nevada State Engineer, Water-Resources Bulletin 8, 59 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239. Available at http://pubs.usgs.gov/sir/2008/5239/.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM): Oregon State University, accessed on April 15, 2005 at http://www.ocs.oregonstate.edu/prism/.

Rosen, M.R., Kropf, Christian, and Thomas, K.A., 2006a, Quantification of the contribution of nitrogen from septic tanks to ground water in Spanish Springs Valley, Nevada: U.S. Geological Survey Scientific Investigations Report 2006–5206, 12 p. Available at http://pubs.usgs.gov/sir/2006/5206/.

Rosen, M.R., Schaefer, D.H., Toccalino, P.L., and Delzer, G.C., 2006b, Occurrence of anthropogenic organic compounds in ground water and finished water of community water systems in Eagle and Spanish Springs Valleys, Nevada, 2002–2004: U.S. Geological Survey Scientific Investigations Report 2006–5210, 31 p. Available at http://pubs.usgs.gov/sir/2006/5210/.

Rush, F.E., and Glancy, P.A., 1967, Water-resources appraisal of the Warm Springs-Lemmon Valley area, Washoe County, Nevada: Nevada Department of Conservation and Natural Resources, Water Resources Reconnaissance Report 43, 70 p. Available at http://water.nv.gov/home/publications/recon.cfm.

Schaefer, D.H., Green, J.M., and Rosen, M.R., 2007, Section 3. Hydrogeologic settings and ground-water flow simulations of the Eagle Valley and Spanish Springs regional study areas, Nevada, *in* Paschke, S.S., ed., Hydrogeologic settings and ground-water flow simulations for regional studies of the transport of anthropogenic and natural contaminants to public-supply wells—Studies begun in 2001: U.S. Geological Survey Professional Paper 1737-A, p. 3-1 to 3-33. Available at http://pubs.usgs.gov/pp/2007/1737a/.

Seiler, R.L., 1999, A chemical signature for ground water contaminated by domestic wastewater: Reno, University of Nevada, unpublished Ph.D dissertation.

Seiler, R.L., 2005, Combined use of ^{15}N and ^{18}O of nitrate and ^{11}B to evaluate nitrate contamination in groundwater: Applied Geochemistry, v. 20, p. 1626–1636.

Stockton, E.L., Jones, C.Z., Rowland, R.C., and Medina, R.L., 2003, Water resources data, Nevada, water year 2003: U.S. Geological Survey Water-Data Report NV–03–1, 679 p. Available at http://pubs.usgs.gov/wdr/wdr-nv-03-1/.

Truckee Meadows Water Authority, 2004, 2005–2025 Truckee Meadows Water Authority water facility plan.

Truner, R.M., and Bawic, W.J., 1996, Digital map of Nevada at scale of 1:500,000, digital dataset for Nevada geology.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed February 2007, at http://www.mrlc.gov/.

Washoe County Department of Water Resources, 2008, 2008 Consumer Confidence Report (2007 water quality data) Spanish Springs.

Welch, A.H., 1994, Ground-water quality and geochemistry in Carson and Eagle Valleys, western Nevada and eastern California: U.S. Geological Survey Open-File Report 93–33, 99 p.

Section 6.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in Las Vegas Valley, Nevada

By Jena M. Huntington

Basin Overview

Las Vegas Valley in southern Nevada is characteristic of basin and range topography. The valley is bounded by high mountain peaks surrounding a valley floor underlain by thick unconsolidated sediments that contain a basin-fill aquifer. The valley is about 30 mi wide and 50 mi long (approximately 1,640 mi^2) (fig. 1). The Spring Mountains to the west and northwest of the valley rise to an altitude of about 11,900 ft at Mount Charleston. The altitude of the valley floor sits at about 1,600 ft and drains southeastward through Las Vegas Wash into Lake Mead on the Colorado River, at about 1,200 ft. Other mountain ranges bordering Las Vegas Valley include the Sheep Range to the north, the Las Vegas Range to the northeast, the McCullough Range to the south, the River Mountains to the southeast, and Frenchman Mountain and Sunrise Mountain to the east.

Plume (1989, p. A2) divided Las Vegas Valley into three physiographic units: mountains, piedmont surfaces, and valley lowlands. The mountain blocks are separated from the valley lowlands by long, gently sloping, laterally continuous piedmont surfaces. These sloped surfaces were interpreted as coalescing alluvial fans in early investigations (Longwell and others, 1965, p. 6; Malmberg, 1965, p. 11), but have since been interpreted as pediment surfaces of older basin-fill deposits (Bell, 1981, p. 10).

The climate of Las Vegas Valley is considered arid, with about 4.5 in/yr of precipitation on the valley floor (period of record 1971–2000; National Oceanic and Atmospheric Administration, 2002, p. 12). Higher altitudes in the Spring Mountains can receive more than 24 in/yr. Mean annual temperature is 68°F on the valley floor (period of record 1971–2000; National Oceanic and Atmospheric Administration, 2002, p. 8), although typically daily high temperatures are 90°F or warmer on more than 125 days each year (Houghton and others, 1975, fig. 22).

In the past, most of the valley's population has resided in the lowlands, although recent expansion to the west, northwest, and southwest has been onto the sloping pediments that formerly were rangeland. The city of Henderson is on a piedmont surface. Further expansion of the urban areas is towards the pediment/mountain contact in the western, southern, and eastern parts of Las Vegas Valley. The population in the Las Vegas area increased from about 795,000 in 1990 to about 1,367,000 in 2000. By 2005 the population had increased by an additional 28 percent, to about 1,752,000 (fig. 2) (Nevada State Demographer's Office, 2009). Corresponding gross water use in the valley, almost all of which was for public supply, was about 325,100 acre-ft in 1990 (Coache, 1990, p. 5), about 529,800 acre-ft in 2000 (Coache, 2000, p. 5), and about 541,300 acre-ft in 2005 (Coache, 2005, p. 5). Surface water from Lake Mead contributed at least 80 percent of the water used in the valley during this same period (1990–2005) for each of these years.

Water Development History

The earliest known people to use water in Las Vegas Valley were the Anasazi, Mojave, and Paiute tribes (Wood, 2000, p. 2). Near the Old Spanish Trail (Mendenhall, 1909, p. 26), the area was named Las Vegas, which is Spanish for "the meadows", due to the lush grassy vegetation surrounding large springs near the center of the valley. In 1844, John C. Fremont described the area as having "…two narrow streams of clear water, 4 or 5 ft deep, with a quick current, from two singularly large springs" (Mendenhall, 1909, p. 92). The next few years saw failed lead mining and farming attempts, but by 1865 the first productive ranch was established. A railroad was built to the valley in 1905 because of its location between Los Angeles and Salt Lake City and its readily available water supply to operate the steam locomotives. Growth of the railroad increased the demand for water, and in 1905 the first well was drilled by the Las Vegas & Tonopah Railroad (Maxey and Jameson, 1948, p. 5; Wood, 2000, p. 8). By 1912, about 125 wells had been drilled in Las Vegas Valley, of which more than half were flowing-artesian wells (Pavelko and others, 1999, p. 52).

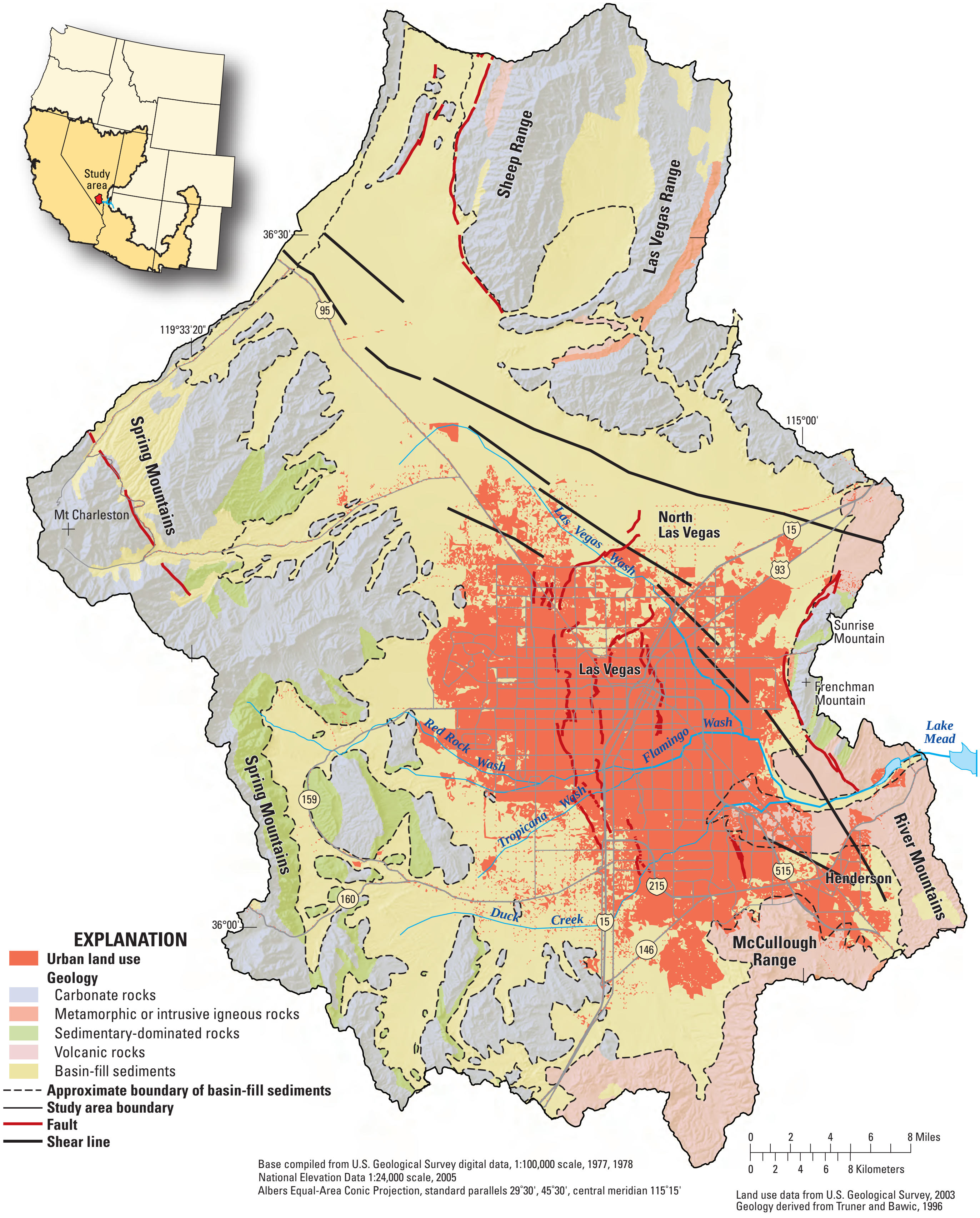


Figure 1. Physiography, land use, and generalized geology of Las Vegas Valley, Nevada.

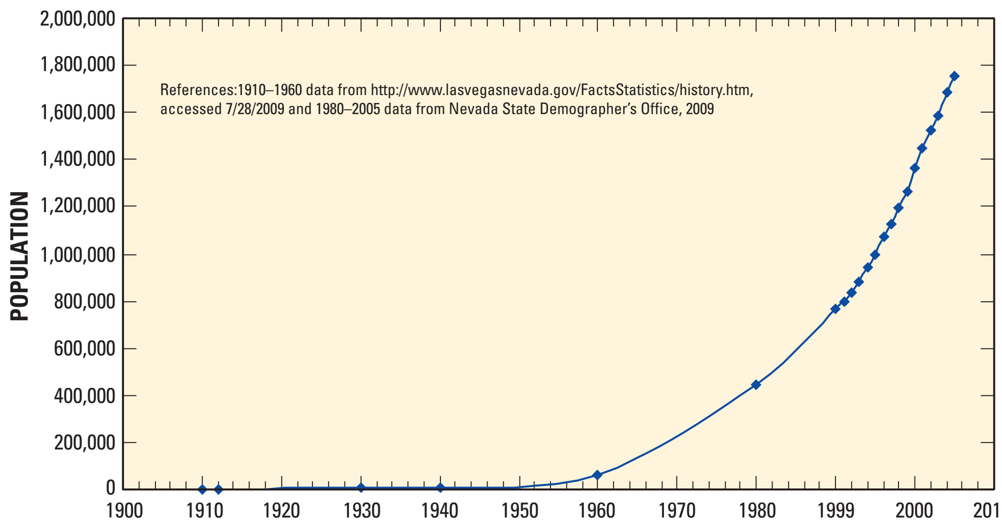


Figure 2. Historical population estimates for Las Vegas Valley, Nevada.

The construction of Boulder Dam (later named Hoover Dam) and Lake Mead began on the Colorado River in 1932. This large-scale project, which provides water and power to Las Vegas, brought in workers from all over the country, thereby accelerating growth in the valley. Industry, including military and gambling, were attracted to Las Vegas throughout the 1940s and 1950s by the availability of land, water, and electrical power. Groundwater development rapidly increased in the valley in the early 1940s and water levels declined in response. Rangeland and agricultural land was urbanized. In 1942, the City of Henderson constructed a pipeline to import Lake Mead water for industrial needs, and in 1955, the Las Vegas Valley Water District began using this pipeline to supplement it's public supply (Wood, 2000, p. 11).

In 1971, the Southern Nevada Water Project constructed a second, larger pipeline to import water from Lake Mead to meet additional water demands of the expanding population (Harrill, 1976, p. 21). Prior to construction of this pipeline, groundwater was the main source of supply for Las Vegas Valley. Projected population growth and federal limits on the importation of water from Lake Mead to the valley ensure a continued need for local groundwater resources (Pavelko and others, 1999, p. 63). Artificial recharge of Colorado River water through injection wells began in 1987, and nearly 16,000 acre-ft/yr was recharged in 2005 (Coache, 2005, p. 4).

Hydrogeology

Basin and Range extensional faulting during the Pliocene epoch broke up Precambrian- and Paleozoic-age carbonate rocks, Permian- through Jurassic-age clastic rocks, and early Tertiary-age igneous rocks into blocks that surround and underlie Las Vegas Valley. Carbonates, siltstone, and sandstone are the primary rock types to the west, north and east; while Tertiary-age volcanic rocks overlie Precambrian-age metamorphic and granitic rocks to the south and southeast (fig. 1). Sediment derived from these rocks fill the basin. Carbonate rocks may transmit groundwater through fractures and solution channels to the basin-fill deposits in the valley, but the other consolidated rocks in the area are likely to be barriers to groundwater flow.

Material eroded off the steep, uplifted mountain blocks has filled the basin with gravel, sand, silt and clay to thicknesses from 3,000 to 10,000 ft (Page and others, 2005, p. 47-48). The basin is interpreted to consist of a deeper depression (5,000 to 13,000 ft deep) beneath most of Las Vegas Valley (Page and others, 2005, fig. 6A) and, on the basis of geophysical data (Morgan and Dettinger, 1996, p. B22), a shallower consolidated-rock surface (less than 1,000 ft deep) on the western side of the valley.

Semiconsolidated material fills most of the valley, and the boundary between Quaternary- and Tertiary-age sediments is not known. The uppermost 1,000 ft of unconsolidated basin-fill deposits are the most productive part of the valley's groundwater system and generally consist of coarse-grained deposits associated with alluvial fans near the mountain fronts grading to predominantly fine-grained lacustrine or playa deposits interfingered with poorly sorted material and thin layers of sand and gravel in the lower parts of the valley (Plume, 1989, p. A10). The basin-fill deposits generally have higher hydraulic conductivities on the northern and western sides of the valley, where basin-fill sediments are derived mostly from carbonate rocks, than on the southern and eastern sides, where sediments are derived from mostly volcanic rocks (Kilroy and others, 1997, p. 9). Layers of sediment are laterally discontinuous because of the varying depositional environments. The precipitation of calcium carbonate from water in the alluvium has formed layers of cemented sediment (caliche) in the subsurface throughout the valley (Covay and others, 1996, p. 16).

Conceptual Understanding of the Groundwater System

The Las Vegas Valley is an open, sediment-filled basin with a complex aquifer system due to laterally and vertically discontinuous layers of clay, silt, sand, gravel, and caliche (fig. 3). Consolidated carbonate-rock aquifers are likely present beneath the sediments, but are not currently used as sources of water supply. The basin-fill deposits contain shallow and near-surface aquifers underlain by a more productive aquifer, called the developed-zone aquifer by Morgan and Dettinger (1996, p. B23) and the principal aquifer by Harrill (1976, p. 11). The most productive part of the basin-fill aquifer is within the uppermost 1,000 ft of sediments on the western side of the valley. The composite depth to water ranges from about 45 to 210 ft in the northern & northwestern parts of the valley, from about 20 to 510 ft in the west-central part, from 0 to about 75 ft in the central part, from about 15 to 110 ft in the east-central part, from flowing (above land surface) to about 30 ft in the southeastern part, and from about 30 to 380 ft in the southern part of the basin (U.S. Geological Survey, 2009a; Nevada Department of Water Resources, 2009; Las Vegas Valley Water District, 2009).

Shallow groundwater can occur within the upper 30 ft of laterally heterogeneous saturated sediments (Van Denburgh and others, 1982, p. 9) although these sediments generally have low hydraulic conductivity and the water is usually prevented from moving deeper than about 50 ft below land surface by impermeable clays or caliche deposits (Southern Nevada Water Authority, 2007). The shallow aquifer is recharged primarily by infiltration of excess irrigation water applied to urban landscapes; this recharge has greatly increased the extent of the shallow aquifer from that of under predevelopment conditions although it is also locally sustained by upward leakage from the deeper aquifer (Malmberg, 1965). Discharge from the shallow aquifer is by evapotranspiration (ET) and by seepage into Las Vegas Wash (Covay and others, 1996, p. 44). In some areas to the northwest of Las Vegas, the shallow aquifer is perched as a consequence of declining water levels in deeper aquifers resulting from groundwater withdrawals. Water in the shallow aquifer is not used as a drinking-water supply.

A near-surface aquifer is present locally within approximately the upper 200–300 ft of primarily fine-grained sediment in the central and eastern parts of Las Vegas Valley. Water occurs in lenses of sand and gravel interbedded with thicker layers of clay and silt that impede downward movement to the underlying principal aquifer. Under natural conditions, recharge was mostly by upward flow from the deeper confined aquifer, but with development, the near-surface aquifer is now also recharged by infiltration of excess irrigation water, leaking sewer lines, and industrial wastewater (Harrill, 1976, p. 11 and fig. 4).

The principal aquifer typically extends from depths of about 200–300 ft to about 1,000 ft below land surface in the central part of the valley and from the water table to about 1,000 ft below land surface along the sides of the valley. Layers and lenses of sand and gravel become separated by layers of clay and silt that create semiconfined to confined conditions toward the middle of the valley (Harrill, 1976, p. 11). The principal aquifer is more productive than the near-surface aquifer and is a source of public-supply water for Las Vegas Valley. Estimates of transmissivity for the principal aquifer range from 500 ft^2/d in the eastern part of the valley (Morgan and Dettinger, 1996, fig. 3.3.1-2) to greater than 14,000 ft^2/d in the western part (Plume, 1989, p. A10-A11). Transmissivity values based on aquifer test results from the northwestern part of the valley have been estimated to be as high as 30,000 ft^2/d (Joseph Leising, Southern Nevada Water Authority, written commun., 2009).

Water in basin-fill deposits deeper than about 1,000 ft probably constitutes a large percentage of the valley's storage capacity, but this deep aquifer is less permeable than the overlying material and yields little water to wells (Morgan and Dettinger, 1996, p. B23). Groundwater likely moves into and out of the deep aquifer from the surrounding and underlying consolidated rock and the overlying principal aquifer.

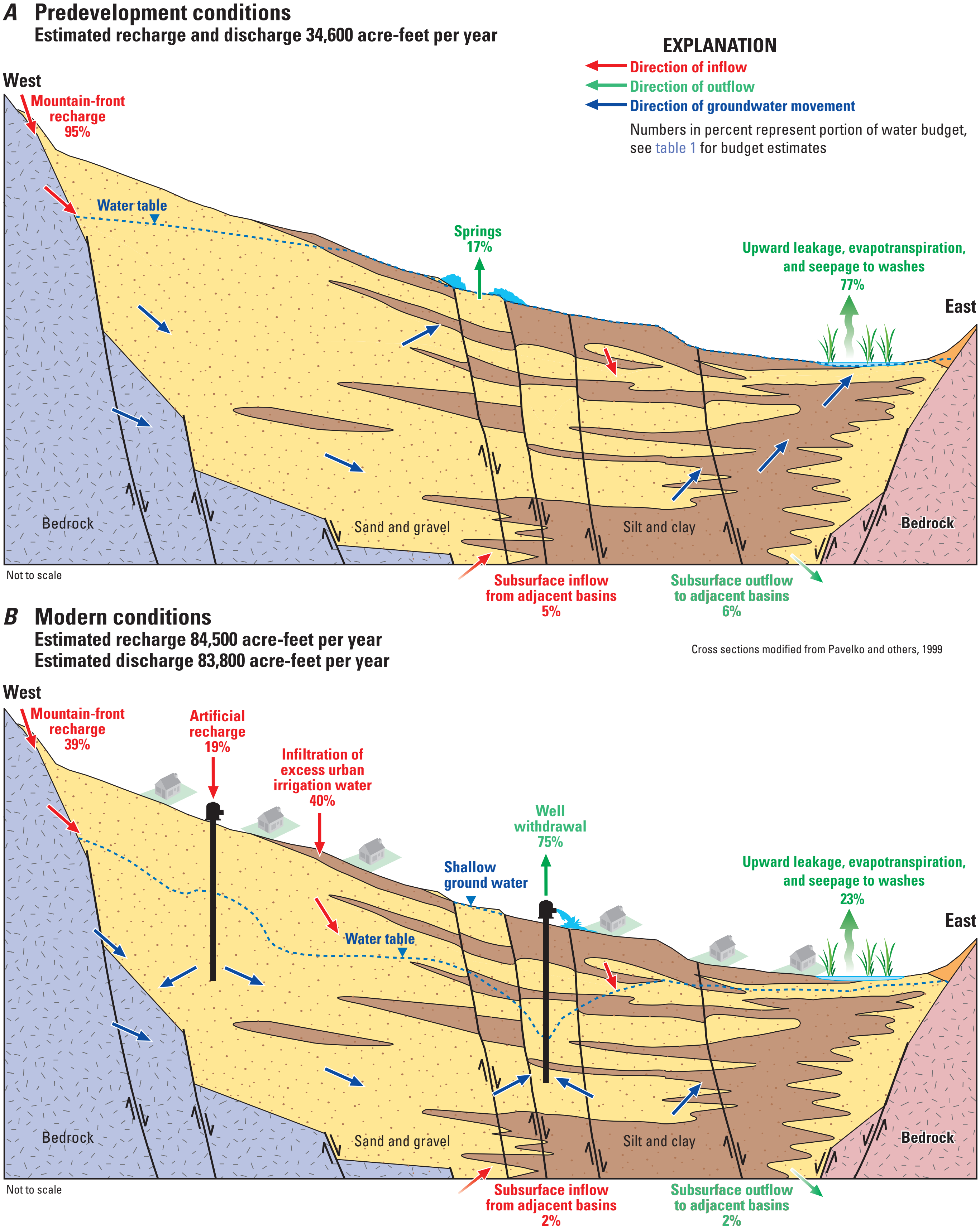


Figure 3. Generalized cross-sections for Las Vegas Valley, Nevada, showing the basin-fill deposits and components of the groundwater flow system under (*A*) predevelopment conditions and (*B*) modern conditions.

Water Budget and Groundwater Flow

Prior to urban development in Las Vegas Valley, recharge to the basin-fill deposits originated primarily as precipitation on the Spring Mountains and the Sheep Range (Bell, 1981, p. 22). This natural recharge entered the principal aquifer along the mountain front either as subsurface inflow from fractures in the consolidated rock, as runoff from the rock, or by a combination of these paths. Under predevelopment conditions, groundwater flowed from the northwest and west across the valley to the southeast and east (fig. 4) (Harrill, 1976, fig. 23). Discharge from the basin-fill aquifer system was by springflow, subsurface outflow to adjacent basins, and ET. Stream channels (washes) in Las Vegas Valley were generally dry, except during floods, with the exception of flow supported by discharge from the larger springs in the central part of the valley (Wood, 2000, p. 2 and fig. 3; Jones and Cahlan, 1975, p. 3-4 and 8).

Estimates of natural recharge along the mountain fronts to Las Vegas Valley made in previous studies and listed by Lopes and Evetts (2004, appendix 1) range from 25,000 to 35,000 acre-ft/yr. Donovan and Katzer (2000) calculated about 51,000 acre-ft of natural recharge to the valley using a modified Maxey-Eakin methodology (Maxey and Eakin, 1949; Eakin and others, 1951) that accounted for greater total precipitation at high elevation in the surrounding mountain blocks. Mountain-front recharge of 33,000 acre-ft/yr was simulated by a groundwater flow model for the valley constructed by Morgan and Dettinger (1996, p. B70), and that value is used in the predevelopment groundwater budget listed in table 1. About 1,600 acre-ft/yr is estimated to enter Las Vegas Valley as subsurface inflow from basins to the southwest (Glancy, 1968).

Discharge from the principal aquifer under predevelopment conditions was primarily by the upward leakage of water to the near-surface and shallow aquifers and then by ET. Evapotranspiration of about 27,000 acre-ft/yr was estimated by Malmberg (1965, table 17) and a value of 24,000 acre-ft/yr was simulated in the model of Morgan and Dettinger (1996, p. B75). Devitt and others (2002) modified the estimates of where ET occurred in the valley before development, as well as estimates of consumptive-use rates, to calculate a much higher discharge by ET of about 40,000 acre-ft/yr.

Major springs in Las Vegas Valley discharged along fault scarps in the basin-fill deposits (fig. 3*A*). Offset along the faults likely caused water that was moving laterally through permeable aquifer layers to be forced upward at contacts with less permeable material (Malmberg, 1965, p. 59). Spring flow in the valley before water development began, estimated to be 6,400 acre-ft/yr by Maxey and Jameson (1948, p. 95), was simulated at 6,000 acre-ft/yr by Morgan and Dettinger (1996, p. B75). Only a small amount of springflow and ephemeral streamflow is thought to have infiltrated into the subsurface to recharge the shallow and near-surface aquifers. Subsurface outflow to the southeast of Las Vegas Valley has been estimated to range from 400 acre-ft/yr (Rush, 1968, table 7) to 2,000 acre-ft/yr (Morgan and Dettinger, 1996, p. B70).

The Las Vegas Valley groundwater flow system has been greatly altered since the early 1900s, when water development began. Discharge from mostly flowing wells in the central part of the valley was almost 15,000 acre-ft in 1912 (Pavelko and others, 1999, p 52). Artesian pressures and the flow from springs declined as a result of the discharge from flowing wells and probably as a consequence of upward seepage from the lower sections of wells that were cased only at upper intervals (Carpenter, 1915, p. 41, p. 40-41). Groundwater pumping rates have exceeded the estimated range of natural recharge rates since the early 1950s (Pavelko and others, 1999, p. 61; Wood, 2000, figs. 2 and 5), and annual withdrawals from wells increased to a maximum of about 86,000 acre-ft in 1968 (Coache, 2005, table 7). Discharge from the largest artesian springs in the central part of the valley had virtually ceased by 1962 as a result of the pumping (Domenico and others, 1964, p. 25).

The northwestern part of Las Vegas Valley has been a major groundwater pumping area since the 1970s, and water-level declines of more than 300 ft were measured in the area by 1990 (fig. 5) (Burbey, 1995, figs. 8 and 9). Water-level declines ranging from 100 to 200 ft have been measured in the central and southeastern parts of the valley (Henderson). Pumping has created large cones of depression, both in the near-surface and principal aquifers, which have disrupted the natural direction of groundwater flow (Covay and others, 1996, p. 48). Instead of flowing generally to the southeast (fig. 4), some of the groundwater now moves toward major pumping centers. In some areas of the central part of the valley, the natural upward hydraulic gradient has been reversed such that there is little to no upward leakage from the principal aquifer. This reversal of gradient may allow leakage of poor-quality water from the land surface to the principal aquifer used for public supply (Dettinger, 1987, p. 18; Bell, 1981, p. 23, 25, and 32).

Land subsidence of more than 5 ft has resulted from groundwater withdrawals and the consequent lowering of hydraulic heads and compaction of fine-grained layers in the basin-fill deposits in areas of Las Vegas Valley. Synthetic aperture radar interferometry (InSAR) data indicate that land subsidence has occurred along a north-south trending zone punctuated by local “bowls” that are bounded by Quaternary-age faults in the central part of the valley (Bell and others, 2002, fig. 7). Although most of the withdrawals are from wells completed in the coarse-grained deposits west of the areas of maximum subsidence, it is hypothesized that these wells have intercepted groundwater that under natural conditions would have sustained the pore-water pressures in the down-gradient, fine-grained part of the aquifer system.

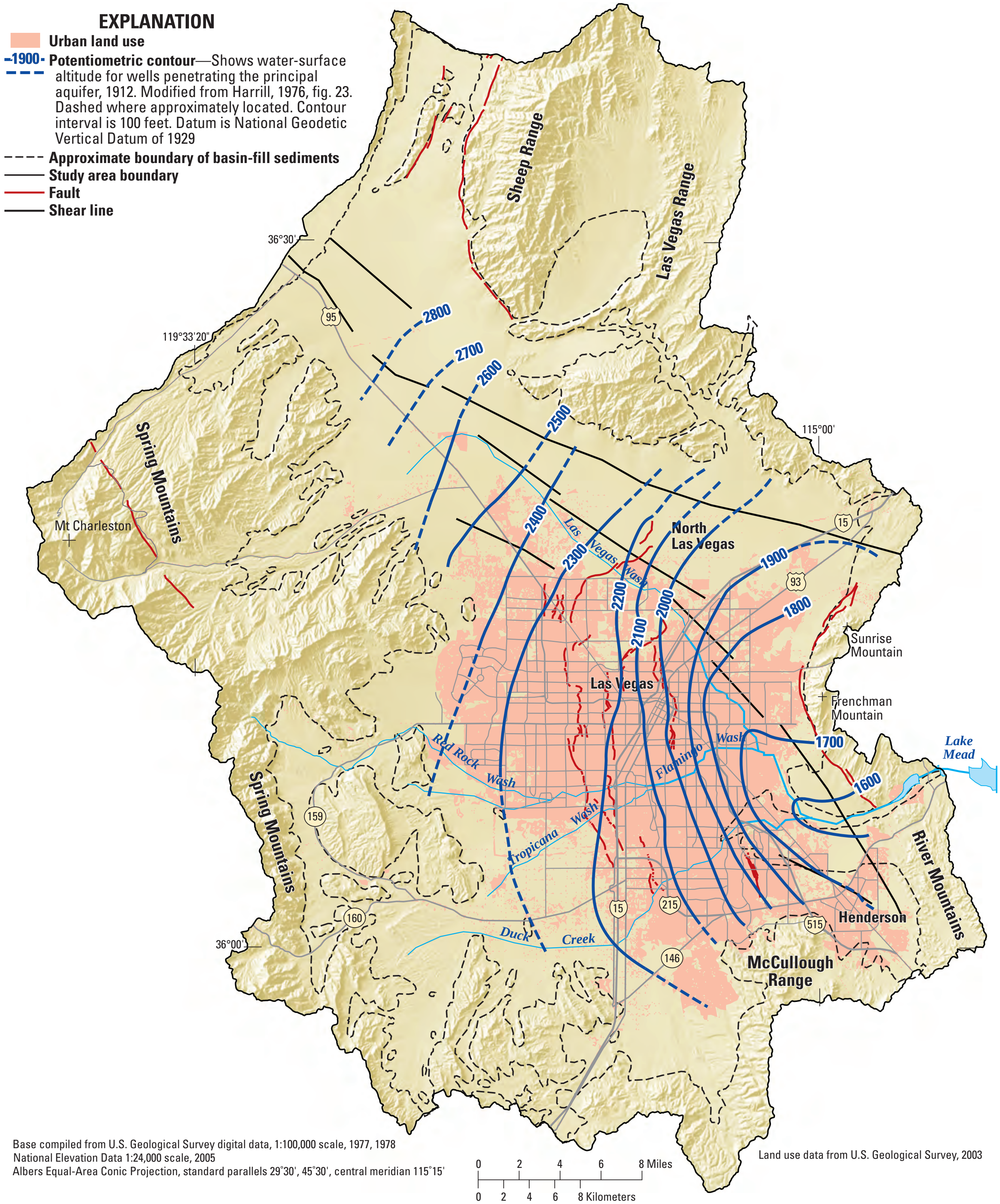


Figure 4. Approximate potentiometric surface in the principal basin-fill aquifer in Las Vegas Valley, Nevada, under predevelopment conditions.

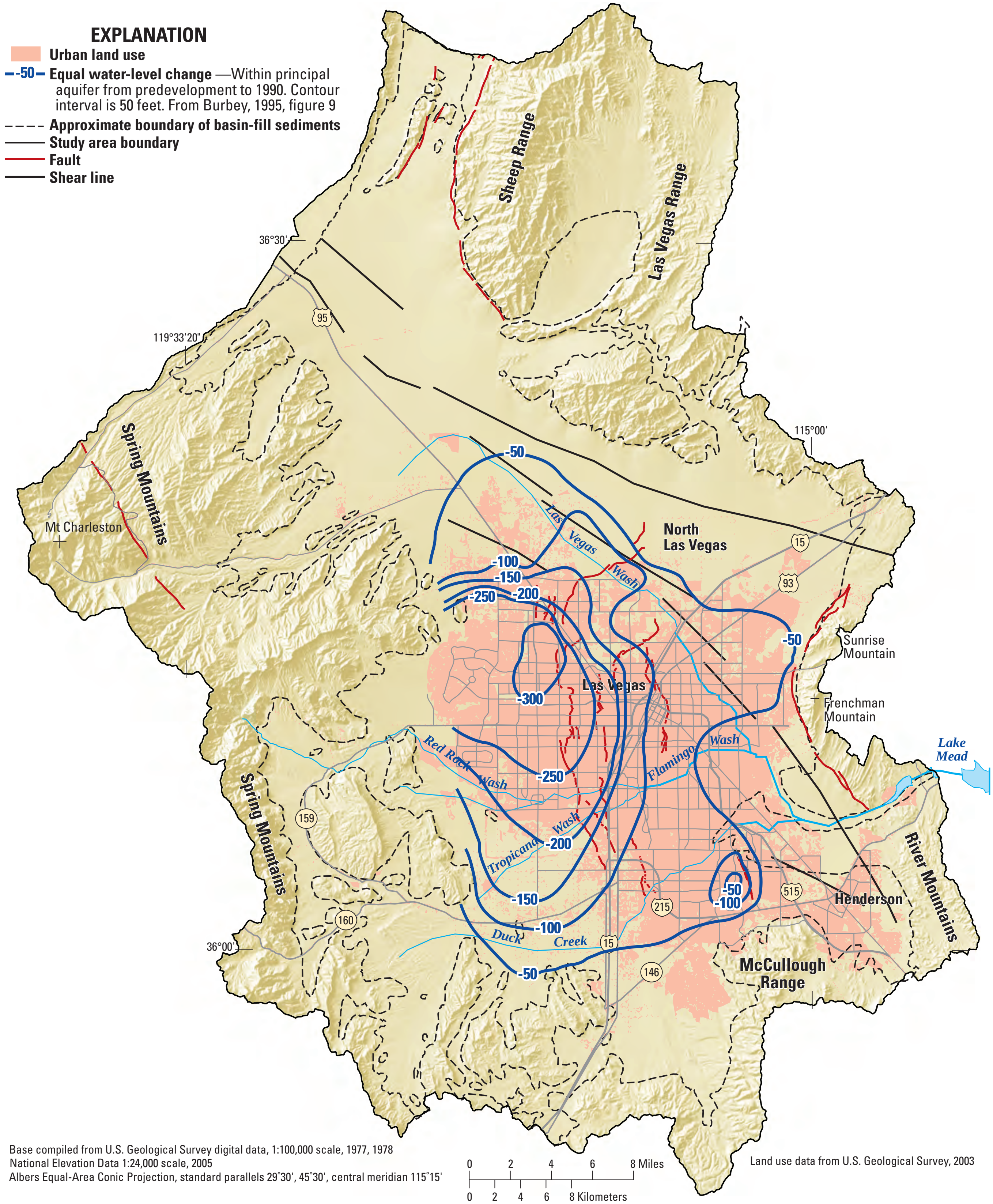


Figure 5. Groundwater-level declines from predevelopment conditions to 1990 in Las Vegas Valley, Nevada.

This results in less hydraulic pressure to support the fine-grained material in areas that have undergone land subsidence. Earth fissures, a type of subsurface ground failure resulting from sediment compaction and coincident pulling apart of the subsurface materials, are associated with groundwater withdrawals in Las Vegas Valley (Pavelko and others, 1999, p. 55-56). Earth fissures were documented as early as 1925 (Bell and Price, 1991, p. C-1) and the potential exists for these features to create pathways between water at the land surface and in the shallower aquifers to water in the deeper aquifers.

In 1971, additional water from Lake Mead was imported for public supply in Las Vegas Valley. Large-scale imports began in 1972, allowing groundwater withdrawals to subsequently decrease to 71,000 acre-ft during that year from the nearly 85,000 acre-ft withdrawn in 1971 (Coache, 2005, table 7). The injection of treated Colorado River water through wells into more transmissive parts of the principal aquifer in the northwestern and central parts of Las Vegas Valley began in 1987 (Coache, 2005, table 7; Wood, 2000, p. 10 and fig. 10). Generally, the water is injected during the winter months when the demand is least. About 32,400 acre-ft of treated Colorado River water was artificially recharged in 1999 and 15,900 acre-ft in 2005 (table 1) (Coache, 2005, table 7). Artificial recharge has allowed withdrawals from wells in the valley to be held to an average of about 71,000 acre-ft/yr for the period from 1988–2005 (Coache, 2005, table 7). Water levels have recovered almost 100 ft from 1990–2005 levels in some areas, and either subsidence has slowed or the land surface has rebounded (Bell and others, 2008, p. 2 and table 1). In areas where municipal pumpage takes place, water levels have in substantial measure been restored by artificial recharge and are maintained by adjusting pumping and recharge in conjunction with extensive monitoring (Joseph Leising, written commun., 2009).

Table 1. Estimated groundwater budget for the basin-fill aquifer in Las Vegas Valley, Nevada, under predevelopment and modern conditions

[All values are in acre-feet per year (acre-ft/yr) and are rounded to the nearest hundred. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Mountain-front recharge	[1]33,000	[1]33,000	0
Infiltration of excess urban irrigation water	0	[2]34,000	34,000
Subsurface inflow from adjacent basins	[3]1,600	[3]1,600	0
Artificial recharge of Colorado River water	0	[4]15,900	15,900
Total recharge	**34,600**	**84,500**	**49,900**
Budget component		**Estimated discharge**	
Upward leakage, evapotranspiration, and seepage to washes	[5]26,600	[2]19,000	-7,600
Well withdrawals	0	[4]62,800	62,800
Springs	[1]6,000	0	-6,000
Subsurface outflow	[1]2,000	[1]2,000	0
Total discharge	**34,600**	**83,800**	**49,200**
Change in storage (total recharge minus total discharge)	0	700	700

[1] Simulated by groundwater flow model of Morgan and Dettinger (1996).

[2] Average of 1972–81 amounts simulated by groundwater flow model of Morgan and Dettinger (1996).

[3] Listed in appendix 2 of Lopes and Evetts (2004).

[4] Water usage in 2005 (Coache, 2005, table 7).

[5] Residual amount between total estimated predevelopment recharge and estimates of other predevelopment discharge components.

Water use in Las Vegas Valley is affected by the large population in an arid climate. About 462,000 acre-ft of imported Colorado River water and 63,000 acre-ft of pumped groundwater was used in 2005 (Coache, 2005, tables 7 and 8), mostly to irrigate the urban landscape in the valley. A minor amount of the water is actually consumed by domestic, agricultural, municipal, and industrial uses. Most of the water used in the valley either evaporates, recharges the shallow groundwater system, or flows into Las Vegas Wash as urban runoff, shallow groundwater discharge, or as treated wastewater. Wastewater from homes and businesses in Las Vegas Valley is piped to water treatment plants for processing. About 16,200 acre-ft of treated wastewater effluent was reclaimed and used to irrigate greenspace such as parks and golf courses in 2005 (Coache, 2005, table 2). The remaining treated wastewater is discharged to the lower reaches of Las Vegas Wash and the wash is now perennial as it flows into Lake Mead at a mean annual flow of about 210,000 acre-ft for years 2003–2008 (U.S. Geological Survey, 2009b).

Recharge to the shallow groundwater system, mostly from excess landscape irrigation water, was simulated at about 30,000 acre-ft in 1972, 46,000 acre-ft in 1981 (Morgan and Dettinger, 1996, p. B94), and averaged 34,000 acre-ft/yr for the period 1972–81 (table 1). Recharge from the infiltration of excess urban irrigation water was estimated to be between 50,000 and 60,000 acre-ft in 1987 (Brothers and Katzer, 1988, p. 7), and likely has continued to increase with the expansion of urban areas in the valley, especially onto the coarse-grained piedmont surfaces. This shallow groundwater would have to move through the natural barriers of fine-grained sediment and caliche to recharge the principal aquifer. Morgan and Dettinger (1996, p. B94) simulated secondary recharge water reaching the near-surface aquifer and continuing downward to the principal aquifer in some areas. The water budget listed in table 1 combines components for the shallow and principal parts of the valley's groundwater system, resulting in little change in storage. Roughly 10,000 acre-ft is estimated to be removed from aquifer storage in 2005 assuming that the principal aquifer receives no recharge from the shallow groundwater.

Effects of Natural and Human Factors on Groundwater Quality

Shallow groundwater in the Las Vegas Valley has been affected by activities at the land surface. The potential exists for contaminants from the land surface to be transported through the shallow and near-surface aquifers to the principal aquifer, where the vertical gradient is downward and where the confining layers are discontinuous or have been breached by wells or by earth movement caused by subsidence. The potential for the transport of contaminants is most likely in areas where the pumping rate in the underlying principal aquifer is high (Hines and others, 1993, p. 41).

The U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program sampled 32 shallow monitoring wells in Las Vegas Valley in 1993 (Neal and Schuster, 1996) and 22 public-supply wells completed in the deeper principal aquifer during 1993-1995 (fig. 6). Data from these samples are used to assess whether water in the principal aquifer has been affected by the overlying shallow groundwater (Lico, 1998, p. 15).

General Water-Quality Characteristics and Natural Factors

Shallow groundwater in Las Vegas Valley sampled as part of a NAWQA study is a moderately saline, magnesium, calcium-sulfate type. Sulfate concentrations were high in these samples, with a median value of 2,000 mg/L. The sulfate is likely from the dissolution of gypsum in desert soils and the recharge of treated wastewater effluent in some areas. The uranium concentration in water sampled from 5 shallow monitoring wells ranged from 7 to 56 µg/L and exceeded the drinking-water standard of 30 µg/L in 2 of the samples (Lico, 1998, fig. 3F). The source of this uranium is not known.

Concentrations of dissolved solids in water samples collected from the shallow monitoring wells ranged from 351 to 5,700 mg/L, with a median of 3,240 mg/L, although the Southern Nevada Water Authority has collected groundwater samples in eastern Las Vegas in which the concentrations of dissolved solids exceeded 10,000 mg/L (Joseph Leising, written commun., 2009). Shallow groundwater in the valley can become mineralized by ET and the dissolution of evaporite deposits. Infiltrating excess landscape irrigation water and a rising water table can dissolve salts formerly precipitated in the unsaturated sediment that can then move into the groundwater system. Water imported to Las Vegas Valley from Lake Mead has a dissolved-solids concentration of approximately 625 mg/L (Anning and others, 2007, appendix 3). Concentrations of dissolved solids exceeding 15,000 mg/L were detected in groundwater samples collected near Henderson in an area that had been an industrial complex built during World War II (Carlsen and others, 1991, p. 39).

Water from 22 public-supply wells completed in the principal aquifer as part of a NAWQA study (fig. 6) was generally a dilute calcium-sulfate type (Lico, 1998, p. 15), with pH values ranging from about 6.2 to 8.3. Sulfate concentrations in water from these wells had a median concentration of 205 mg/L, and concentrations of dissolved arsenic ranged from 1 to 11 µg/L, with a median concentration of 2 µg/L. The elevated concentrations of sulfate are likely a consequence of recharge by sulfate-enriched water from Lake Mead (Joseph Leising, written commun., 2009).

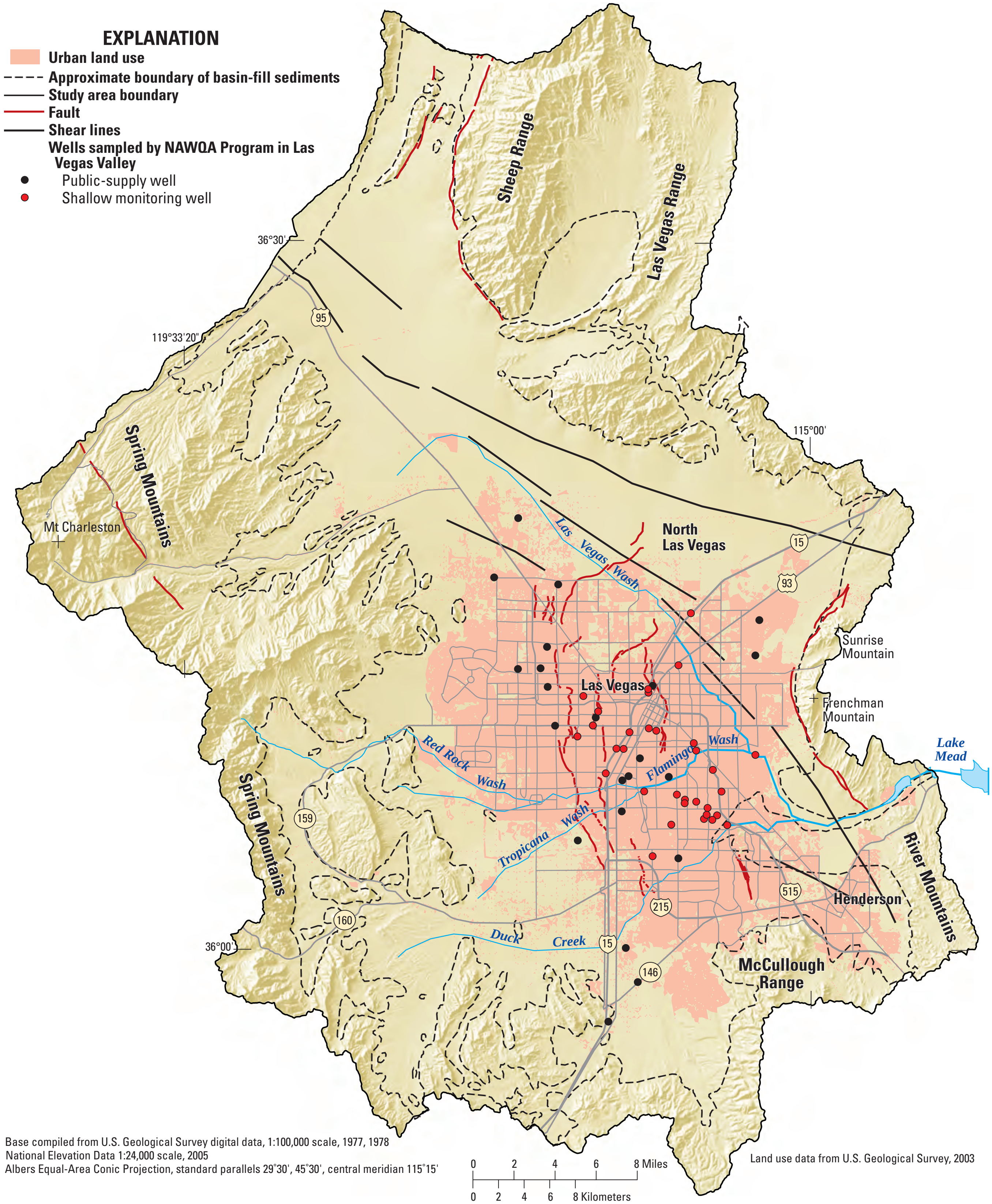


Figure 6. Location of wells sampled in Las Vegas Valley, Nevada, by the NAWQA Program.

Concentrations of dissolved solids in the principal aquifer in Las Vegas Valley typically increase from the northwestern and northern parts of the valley, where mountain-front recharge occurs, to the southeastern part of the valley, where Las Vegas Wash exits the valley (fig. 7). Dissolved-solids concentrations exceeded the secondary drinking-water standard of 500 mg/L (U.S. Environmental Protection Agency, 2008; each time a primary or secondary drinking-water standard is mentioned in this section, it denotes this citation) in samples from more than half of the 22 NAWQA sampled wells completed in the principal aquifer. The median dissolved-solids concentration in water sampled by NAWQA from the principal aquifer was 565 mg/L. The mixing of secondary recharge water and artificially recharged water imported from Lake Mead with native groundwater will likely result in an increase in dissolved-solids concentrations in parts of the principal-aquifer system in the valley.

Potential Effects of Human Factors

Factors that can affect the quality of water in the principal aquifer of Las Vegas Valley are the chemical composition of water recharged at land surface and injected into the aquifer; the reversal in hydraulic gradient caused by withdrawals from wells that can lead to downward leakage from shallow parts of the groundwater system; land subsidence resulting in the local release of poor-quality water owing to compaction of fine-grained sediments (Covay and others, 1996, p. 48); and fissures in confining layers, breaks in well casings, “leaky” well completions, and abandoned wells that allow “short circuiting” of water between aquifers (Lico, 1998, p. 21).

The effects of human activities on groundwater quality are most commonly detected in the shallow aquifer (table 2). The median nitrate concentration as nitrogen of water sampled from the 32 NAWQA monitoring wells was 4.6 mg/L, and the drinking-water standard of 10 mg/L was exceeded in 12 percent of the samples. A study of groundwater quality in the valley by Dinger (1977) showed that nitrate concentrations averaged 13 mg/L in 35 water samples from wells less than or equal to 100 ft deep and 3.2 mg/L in 250 samples from wells 100–300 ft deep. Likely sources of nitrate in shallow groundwater in Las Vegas Valley are fertilizers applied to lawns, irrigation using treated sewage effluent, and leakage from sewage disposal systems (Kaufmann, 1977, p. 85). Also, naturally occurring nitrate that has accumulated over thousands of years in desert soils can be flushed to the water table by excess irrigation water (Walvoord and others, 2003, p. 1021-1024). Hess and Patt (1977, p. 33) attributed nitrate concentrations greater than 10 mg/L in shallow groundwater in an area northwest of Las Vegas to natural sources. Nitrate concentrations as nitrogen in water from the 22 NAWQA-sampled public-supply wells were less than 2 mg/L, with a median concentration of 0.65 mg/L.

At least one volatile organic compound (VOC) was detected in 80 percent of the NAWQA samples from 31 shallow groundwater monitoring wells (1 well of 32 was not sampled) and 50 percent of the samples from 20 principal aquifer supply wells (2 principal aquifer wells were not sampled) in Las Vegas Valley. Chloroform was detected in 21 shallow groundwater samples at concentrations from 0.2 to 12 µg/L and in 10 principal aquifer samples at concentrations from 0.2 to 23 µg/L (Lico, 1998, table 2). A major source of chloroform is from the infiltration and injection of chlorinated water imported from Lake Mead and the infiltration of chlorinated groundwater applied at the land surface. Excess free chlorine in the treated water also can react with dissolved organic carbon present in the groundwater to produce chloroform. The solvent tetrachloroethylene (PCE) was detected in 8 shallow groundwater samples at concentrations from 0.2 to 89 µg/L and in 2 principal aquifer samples at concentrations of 0.4 and 21 µg/L (Lico, 1998, table 2). The drinking-water standard for PCE is 5 µg/L.

The herbicides atrazine and prometon were detected in water sampled from 3 and 5 shallow monitoring wells, respectively, and in water sampled from 2 and 1 deeper water-supply wells, respectively (Lico, 1998, table 1). These pesticides are commonly used in urban areas to control unwanted vegetation, and their presence, even at very low concentrations in a small percentage of the NAWQA-sampled wells, indicates the potential for human activities to affect the water quality of the basin-fill aquifers in Las Vegas Valley.

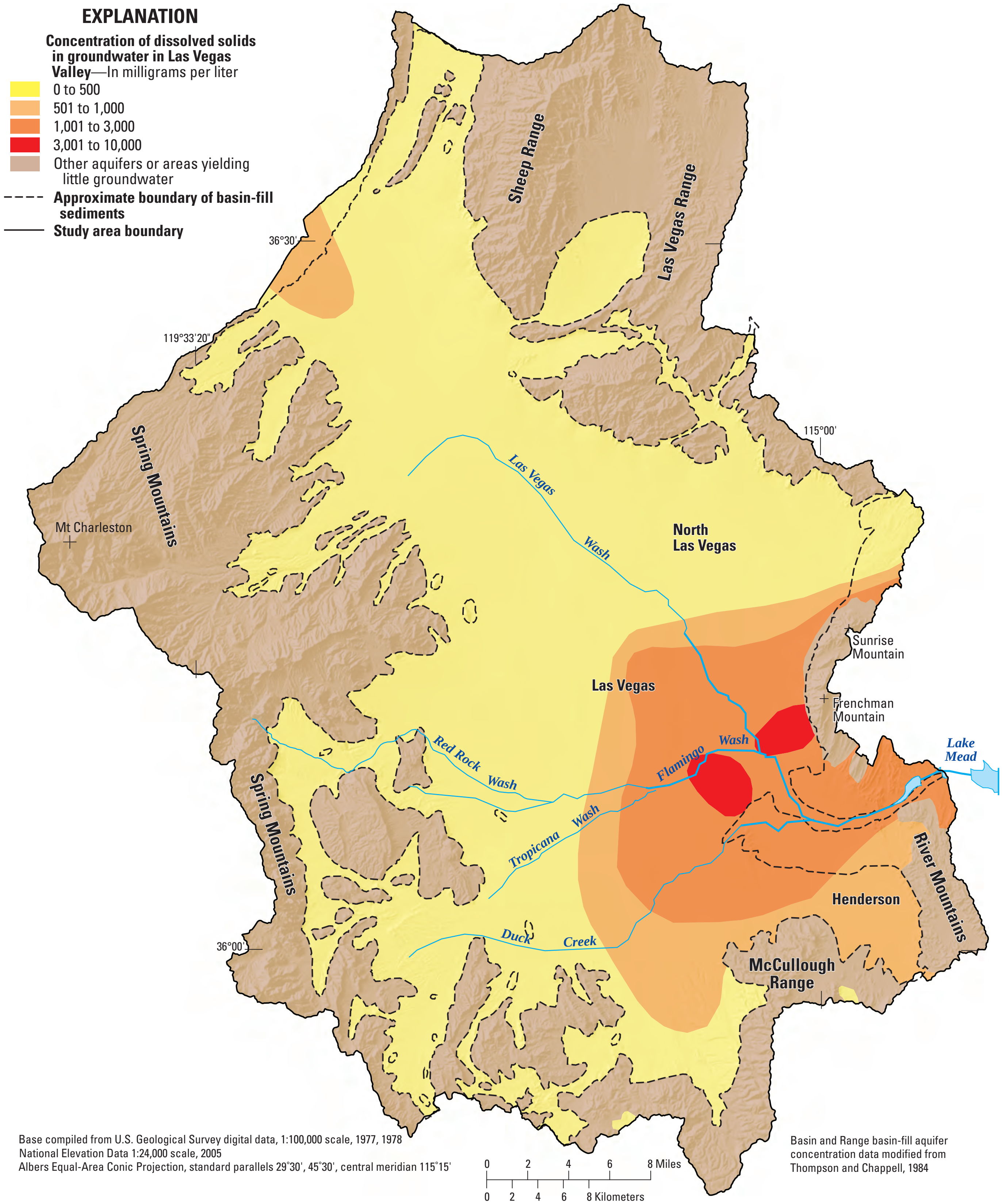


Figure 7. Concentrations of dissolved solids in groundwater in Las Vegas Valley, Nevada.

Table 2. Summary of selected constituents in groundwater in Las Vegas Valley, Nevada, and sources or processes that affect their presence or concentration.

[mg/L, milligrams per liter]

	General location	Median value or detections	Possible sources or processes
		Shallow aquifers	
Dissolved solids	Greatest in the southeast	3,240 mg/L	Evapotranspiration and dissolution
Sulfate	Basin wide	2,000 mg/L	Possibly gypsum dissolution due to irrigation with treated effluent
Nitrate	Urban areas	4.6 mg/L	Natural sources, fertilizers, treated wastewater, leaky sewer pipes, septic systems
Volatile organic compounds detections	Basin wide	71	Point sources including underground gasoline tanks and solvents from repair shops and dry cleaners
Pesticide detections	Urban areas	12	Lawn application
		Principal aquifers	
Dissolved solids	Greatest in the southeast	565 mg/L	Imported and artificially recharged Lake Mead water, dissolution
Sulfate	Central and southern	205 mg/L	Associated with altered consolidated rocks
Nitrate	Central basin/urban areas	0.65 mg/L	Natural sources, potential downward movement from shallow aquifers
Volatile organic compounds	Basin wide	40	Potential downward movement from shallow aquifers, artificially recharged Lake Mead water
Pesticide detections	Urban areas	6	Potential downward movement from shallow aquifers

Summary

Las Vegas Valley is a hydraulically open basin just east of the Spring Mountains in southern Nevada. Prior to urban development in the valley, recharge to the groundwater system originated primarily as precipitation in the headwater areas of the Spring Mountains and the Sheep Range. Groundwater flowed from the northwest across the valley to the southeast and discharged by springflow, subsurface outflow to adjacent basins, and evapotranspiration. The Las Vegas Valley aquifer system is complex due to the presence and effects of laterally and vertically discontinuous beds of clay, silt, sand, gravel, and caliche. Consolidated carbonate-rock aquifers are present at greater depth, but are not currently used as sources of water supply. The near-surface groundwater system is generally semiconfined while the deeper aquifers are confined in the central part of the valley.

Large population increases in an arid climate have resulted in a human-driven hydrologic cycle affected by groundwater pumping, artificial recharge, and secondary recharge. Pumping has created large cones of depression in parts of the valley. In areas where municipal pumping takes place, water levels have in substantial measure been restored by artificial recharge and are maintained by adjusting pumping and recharge in conjunction with extensive monitoring. Large declines in groundwater levels have caused compaction of fine-grained sediments within the principal aquifer, resulting in land subsidence of more than 5 ft and the development of earth fissures. Natural recharge to the principal aquifer is now supplemented by large volumes of secondary recharge from either pumped groundwater or imported Lake Mead water. Natural upward hydraulic gradients have also been reversed in the central part of the valley, leading to the cessation of springflow and the leakage of poorer-quality shallow groundwater into deeper aquifers.

Shallow groundwater in the Las Vegas Valley has been affected by activities at the land surface. Where the vertical gradient is downward and where the confining layers are discontinuous or have been breached by wells or by movement caused by subsidence, the potential exists for contaminants from the land surface to be transported through the shallow and near-surface aquifers to the principal aquifer. The median concentration of dissolved solids in shallow groundwater sampled by NAWQA was 3,240 mg/L. The shallow groundwater becomes mineralized as a consequence of evapotranspiration and the dissolution of evaporite deposits. Infiltrating landscape irrigation water and a rising water table can dissolve salts precipitated in the unsaturated sediment that can then move into the groundwater system. Dissolved-solids concentration in the principal aquifer typically increases

from the northwestern and northern parts of the valley where mountain-front recharge occurs to the southeastern part of the valley where Las Vegas Wash exits the valley. The median dissolved-solids concentration in water sampled by NAWQA from the principal aquifer was 565 mg/L.The continued addition of artificially recharged water and secondary recharge will likely result in an increase in dissolved-solids concentrations in many parts of the groundwater system in the valley.

Other factors that can affect the quality of water in the principal aquifer of Las Vegas Valley are the chemical composition of water recharged at land surface and injected into the aquifer; the reversal in hydraulic gradient caused by withdrawals from wells that can lead to leakage from shallow parts of the groundwater system; land subsidence resulting in the local release of poor-quality water owing to compaction of fine-grained sediments; and fissures in confining layers, breaks in well casings, improper “leaky” well completions, and abandoned wells that allow “short circuiting” of water between aquifers. The effects of human activities on groundwater quality is more commonly observed in the shallow aquifer with higher median nitrate concentrations and more frequent detections of volatile organic compounds and pesticides. Likely sources of nitrate to shallow groundwater in Las Vegas Valley are fertilizers applied to lawns, irrigation using treated sewage effluent, leakage from sewage disposal systems, and the flushing of naturally occurring nitrate from the unsaturated zone. The volatile organic compound chloroform was frequently detected in the NAWQA groundwater samples. Major sources of chloroform are the infiltration and injection of chlorinated water imported from Lake Mead and the infiltration of chlorinated groundwater used for irrigation.

References Cited

Anning, D.W., Bauch, N.J., Gerner, S.J., Flynn, M.E., Hamlin, S.N., Moore, S.J., Schaefer, D.H., Anderholm, S.K., and Spangler, L.E., 2007, Dissolved solids in basin-fill aquifers and streams in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2006–5315, 168 p. Available at http://pubs.usgs.gov/sir/2006/5315/.

Bell, J.W., 1981, Subsidence in Las Vegas Valley: Nevada Bureau of Mines and Geology Bulletin 95, 84 p.

Bell, J.W., Amelung, F., Ferritti, A., Bianchi, M., and Novali, F., 2008, Permanent scatterer InSAR reveals seasonal and long-term aquifer-system response to groundwater pumping and artificial recharge: Water Resources Research, v. 44, W02407, 18 p.

Bell, J.W., Amelung, Falk, Ramelli, A.R., and Blewitt, Geoff, 2002, Land subsidence in Las Vegas, Nevada, 1935–2000—New geodetic data show evolution, revised spatial patterns, and reduced rates: Environmental & Engineering Geoscience, v. 8, no. 3, August 2002, p. 155–174.

Bell, J.W., and Price, J.G., 1991, Subsidence in Las Vegas Valley, 1980–91—Final project report: Nevada Bureau of Mines and Geology, Open-File Report 93–4, 10 sect., 9 plates, scale 1:62,500.

Brothers, Kay, and Katzer, Terry, 1988, Ground-water chemistry changes resulting from stressed aquifer systems in Las Vegas Valley, Clark County, Nevada: Nevada Division of Environmental Protection, 66 p.

Burbey, T.J., 1995, Pumpage and water-level change in the principal aquifer of Las Vegas Valley, 1980–90: Nevada Division of Water Resources Information Report 34, 224 p.

Carpenter, Everett, 1915, Ground water in southeastern Nevada: U.S. Geological Survey Water-Supply Paper 365, 86 p.

Carlsen, C.L., Lunnis, R.C., and Prudic, D.E., 1991, Changes in water levels and water quality in shallow ground water, Pittman-Henderson area, Clark County, Nevada, resulting from diversion of industrial cooling water from ditch to pipeline in 1985: U.S. Geological Survey Water-Resources Investigations Report 89–4093, 69 p.

Coache, Robert, 1990, Las Vegas Valley water usage report, Clark County, Nevada, 1990: Nevada Division of Water Resources, 100 p.

Coache, Robert, 2000, Las Vegas Valley water usage report, Clark County, Nevada, 2000: Nevada Division of Water Resources, 109 p.

Coache, Robert, 2005, Las Vegas Valley water usage report, Clark County, Nevada, 2005: Nevada Division of Water Resources, 112 p.

Covay, K.J., Banks, J.M., Bevans, H.E., and Watkins, S.A., 1996, Environmental and hydrologic settings of the Las Vegas Valley area and the Carson and Truckee River basins, Nevada and California: U.S. Geological Survey Water-Resources Investigations Report 96–4087, 72 p.

Dettinger, M.D., 1987, Ground-water quality and geochemistry of Las Vegas Valley, Clark County, Nevada, 1981–83—Implementation of a monitoring network: U.S. Geological Survey Water-Resources Investigations Report 87–4007, 69 p.

Devitt, D.A., Donovan, D.J., Katzer, Terry, and Johnson, Michael, 2002, A reevaluation of the ground water budget for Las Vegas Valley, Nevada, with emphasis on ground water discharge: Journal of the American Water Resources Association, v. 38, no. 6, p. 1735–1751.

Dinger, J.S., 1977, Relation between surficial geology and near-surface ground-water quality, Las Vegas Valley, Nevada: University of Nevada, Reno, unpublished Ph.D. thesis, 215 p.

Domenico, P.A., Stephenson, D.A., and Maxey, G.B., 1964, Ground water in Las Vegas Valley: Nevada Department of Conservation and Natural Resources, Division of Water Resources Bulletin No. 29, 53 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Donovan, D.J., and Katzer, Terry, 2000, Hydrologic implications of greater ground-water recharge to Las Vegas Valley, Nevada: Journal of the American Water Resources Association, v. 36, no. 5, p. 1133–1148.

Eakin, T.E., Maxey, G.B., Robinson, T.W., Fredericks, J.C., and Loeltz, O.J., 1951, Contributions to the hydrology of eastern Nevada: Nevada State Engineer, Water Resources Bulletin 12, 171 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Glancy, P.A., 1968, Water-resouces appraisal of Mesquite-Ivanpah Valley area, Nevada and California: Nevada Division of Water Resources, Reconnaissance Report 46, 57 p. Available at http://water.nv.gov/home/publications/recon.cfm.

Harrill, J.R., 1976, Pumping and ground-water storage depletion in Las Vegas Valley, Nevada, 1955–74: Nevada Department of Conservation and Natural Resources, Division of Water Resources Bulletin No. 44, 70 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Hess, J.W., and Patt, R.O., 1977, Nitrogen contamination sources in shallow ground water, Las Vegas area, Nevada: Desert Research Institute Publication No. 32.

Hines, W.G., Cole, E.D., and Donovan, D.J., 1993, Ground-water quality in the Las Vegas Valley, Clark County, Nevada—An update and analysis of recent trends: Nevada Division of Environmental Protection, 43 p.

Houghton, J.G., Sakamoto, C.M., and Gifford, R.O., 1975, Nevada’s weather and climate: Nevada Bureau of Mines and Geology Special Publication 2, 78 p.

Jones, F.L., and Cahlan, J.F. 1975, Water, a history of Las Vegas—Volume I: Las Vegas Valley Water District, 171 p.

Kaufmann, R.F., 1977, Land and water use impacts on ground-water quality in Las Vegas Valley: Ground Water, v. 15, no. 1, p. 81–89.

Kilroy, K.C., Lawrence, S.J., Lico, M.S., Bevans, H.E., and Watkins, S.A., 1997, Water-quality assessment of the Las Vegas Valley area and the Carson and Truckee River Basins, Nevada and California—Nutrients, pesticides, and suspended sediment, October 1969–April 1990: U.S.Geological Survey Water-Resources Investigations Report 97–4106, 144 p.

Las Vegas Valley Water District, 2009, Artificial recharge annual report: Doc. No. LVVWD–ED–002.

Lico, M.S., 1998, Quality of ground water beneath urban and agricultural lands in Las Vegas Valley and the Carson and Truckee River Basins, Nevada—Implications for water supply: U.S. Geological Survey Water-Resources Investigations Report 97–4259, 24 p.

Longwell, C.R., Pampeyan, E.H., Bowyer, Ben, and Roberts, R.J., 1965, Geology and mineral deposits of Clark County, Nevada: Nevada Bureau of Mines Bulletin 62, 217 p.

Lopes, T.J., and Evetts, D.M., 2004, Ground-water pumpage and artificial recharge estimates for calendar year 2000 and average annual natural recharge and interbasin flow by hydrographic area, Nevada: U.S. Geological Survey Scientific Investigations Report 2004–5239, 81 p. Available at http://pubs.usgs.gov/sir/2004/5239/.

Malmberg, G.T., 1965, Available water supply of the Las Vegas ground-water basin, Nevada: U.S. Geological Survey Water-Supply Paper 1780, 116 p.

Maxey, G.B., and Eakin, T.E., 1949, Ground water in White River Valley, White Pine, Nye, and Lincoln Counties, Nevada: Nevada State Engineer, Water-Resources Bulletin 8, 59 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Maxey, G.B., and Jameson, C.H., 1948, Geology and water resources of Las Vegas, Pahrump, and Indian Springs Valleys, Clark and Nye Counties, Nevada: Nevada State Engineer, Water Resources Bulletin No. 5, 292 p. Available at http://water.nv.gov/home/publications/bulletins.cfm.

Mendenhall, W.C., 1909, Some desert watering places in southeastern California and southwestern Nevada: U.S. Geological Survey Water-Supply Paper 224, 98 p.

Morgan, D.S. and Dettinger, M.D., 1996, Ground-water conditions in Las Vegas Valley, Clark County, Nevada—Part 2, Hydrogeology and simulation of ground-water flow: U.S. Geological Survey Water Supply Paper 2320–B, 124 p.

National Oceanic and Atmospheric Administration, 2002, Monthly station normals of temperature, precipitation, and heating and cooling degree days, 1971–2000, Climatology of the United States, No. 81, Nevada: National Climatic Data Center, Asheville, North Carolina, 26 p.

Neal, E.G. and Schuster, P.F., 1996, Data on quality of shallow ground water, Las Vegas urban area, Nevada, 1993: U.S. Geological Survey Open-File Report 96–552, 1 plate.

Nevada Division of Water Resources, 2009, Water Level Database, accessed August 2009 at http://water.nv.gov/well%20net/download_data.cfm.

Nevada State Demographer's Office, 2009, Nevada County population estimates July 1, 1986 to July 1, 2008, includes cities and towns.

Page, W.R., Lundstrom, S.C., Harris, A.G., Langenheim, V.E., Workman, J.B., Mahan, S.A., Paces, J.B., Dixon, G.L., Rowley, P.D., Burchfiel, B.C., Bell, J.W., and Smith, E.I., 2005, Geologic and geophysical maps of the Las Vegas 30' x 60' quadrangle, Clark and Nye Counties, Nevada, and Inyo County, California: U.S. Geological Survey Scientific Investigations Map 2814, 58 p., 2 sheets.

Palvelko, P.T., Wood, D.B, and Laczniak, R.J., 1999, Las Vegas, Nevada—Gambling with water in the desert *in* Galloway, D., Jones, D.R., and Ingebritsen, S.E., eds., Land Subsidence in the United States: U.S. Geological Survey Circular 1182, 117 p. Available at http://pubs.usgs.gov/circ/circ1182/.

Plume R.W., 1989, Ground-water conditions in Las Vegas Valley, Clark County, Nevada—Part 1, hydrogeologic framework: U.S. Geological Survey Water-Supply Paper 2320–A, 15 p.

Rush, F.E., 1968, Water-resources appraisal of the Lower Moapa-Lake Mead area, Clark County, Nevada: Nevada Division of Water Resources, Reconnaissance Report 50, 66 p. Available at http://water.nv.gov/home/publications/recon.cfm.

Southern Nevada Water Authority, 2007, Las Vegas Valley groundwater management program, accessed October 22, 2008 at http://www.lasvegasgmp.com/html/lvgw_shallow_aquifer.html.

Thompson, T.H., and Chappell, Richard, 1984, Maps showing distribution of dissolved solids and dominant chemical type in ground water, Basin and Range province, Nevada: U.S. Geological Survey Water-Resources Investigations Report 83–4119–C, 6 p., 4 sheets, scale 1:500,000.

Truner, R.M., and Bawic, W.J., 1996, Digital Map of Nevada at scale of 1:500,000, Digital dataset for Nevada geology.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed February 2007 at http://www.mrlc.gov/.

U.S. Geological Survey, 2009a, Active Groundwater Level Network: data available at http://groundwaterwatch.usgs.gov/countymaps/NV_003.html.

U.S. Geological Survey, 2009b, National Water Information System (NWISWeb), accessed March 19, 2009 at http://waterdata.usgs.gov/nwis/annual/?format=sites_selection_links&search_site_no=09419700&referred_module=sw.

Van Denburgh, A.S., Seitz, H.R., Durbin, T.J., and Harrill, J.R., 1982, Proposed monitoring network for ground-water quality, Las Vegas Valley, Nevada: U.S. Geological Survey Open-File Report 80–1286, 25 p.

Walvoord, M.A., Phillips, F.M., Stonestrom, D.A., Evans, R.D., Hartsough, P.C., Newman, B.D., and Striegl, R.G., 2003, A reservoir of nitrate beneath desert soils: Science, v. 302, p. 1021–1024.

Wood, D.B., 2000, Water use and associated effects on ground-water levels, Las Vegas Valley and vicinity, Clark County, Nevada, 1980–95: Nevada Department of Conservation and Natural Resources, Division of Water Resources, Water-Resources Information Report 35, 101 p. Available at http://water.nv.gov/home/publications/infoseries.cfm.

This page intentionally left blank.

Section 7.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the West Salt River Valley, Arizona

By David W. Anning

Basin Overview

The West Salt River Valley (fig. 1) in central Arizona has an arid climate and significant water-resources development that support agricultural and urban activities. The basin-fill aquifer is an important resource as it provides about one-third to half of the water supply for the valley, the amount varying annually in part due to the availability of surface-water supplies, imported water, and treated wastewater effluent (Arizona Department of Water Resources, 1999). Groundwater development to support the population and their economic and cultural activities over the past century has caused substantial changes in the basin-fill aquifer, including about a 7-fold increase in recharge and an 8-fold increase in discharge. These and other changes to the aquifer have resulted in an increase in the intrinsic susceptibility of the aquifer to contamination. The effects of both natural and human-related factors on groundwater quality in the valley are discussed in this section.

The West Salt River Valley is a 1,438 mi^2 hydrogeologic area defined by McKinney and Anning (2009) that is approximately coincident with the West Salt River Valley groundwater basin defined by the Arizona Department of Water Resources, except that it encompasses an additional 108 mi^2 of land in areas near the Sierra Estrella, White Tank Mountains, and Hedgpeth Hills. The altitude of the valley floor ranges from about 2,500 ft in the northwestern part of the valley to about 800 ft along the Gila River west of Buckeye. The mountains comprise a smaller portion of the basin than the valley floor, and rise to about 4,500 ft in the Sierra Estrella. The valley is an open basin that is drained by the Gila River and its tributaries, which include the Salt River and Agua Fria River (fig. 1).

The climate of the West Salt River Valley is characterized by hot summers, mild winters, and large diurnal temperature cycles, and is among the warmest and most arid of the basins investigated in this study. Average precipitation for the basin for 1971–2000 was about 9 in/yr, making it the second driest basin of those in the study (McKinney and Anning, 2009). For the period 1961–90, the mean monthly maximum temperature at Buckeye was 68.1°F in January and 109.2°F in July (Owenby and Ezell, 1992).

A large part of the West Salt River Valley has been developed for agricultural, residential, commercial, and industrial uses. Population of the valley for 2005 is estimated to be about 1.97 million people (McKinney and Anning, 2009), most of whom live in the Phoenix metropolitan area. The remainder of the population lives in surrounding farming communities and new communities that have replaced farmland. Land use within the alluvial basin, excluding the surrounding mountainous areas, consists of about 22 percent agricultural and 34 percent urban use (fig. 1; McKinney and Anning, 2009). Most of the present-day urban land was previously agricultural land. Important agricultural crops in the valley include cotton, alfalfa, wheat, and vegetables.

Water demands for municipal and agricultural purposes in the West Salt River Valley are met using a variety of water sources. These include groundwater from the basin-fill aquifer; surface water from the Agua Fria, Gila, Salt, and Verde Rivers, most of which is stored in reservoirs outside the valley; water from the Colorado River imported through the Central Arizona Project (Hayden Rhodes Aqueduct in fig. 1); and recycled water from municipal wastewater-treatment plants. Development of these sources, which has significantly altered the hydrologic system of the valley, is discussed further in the following parts of this section.

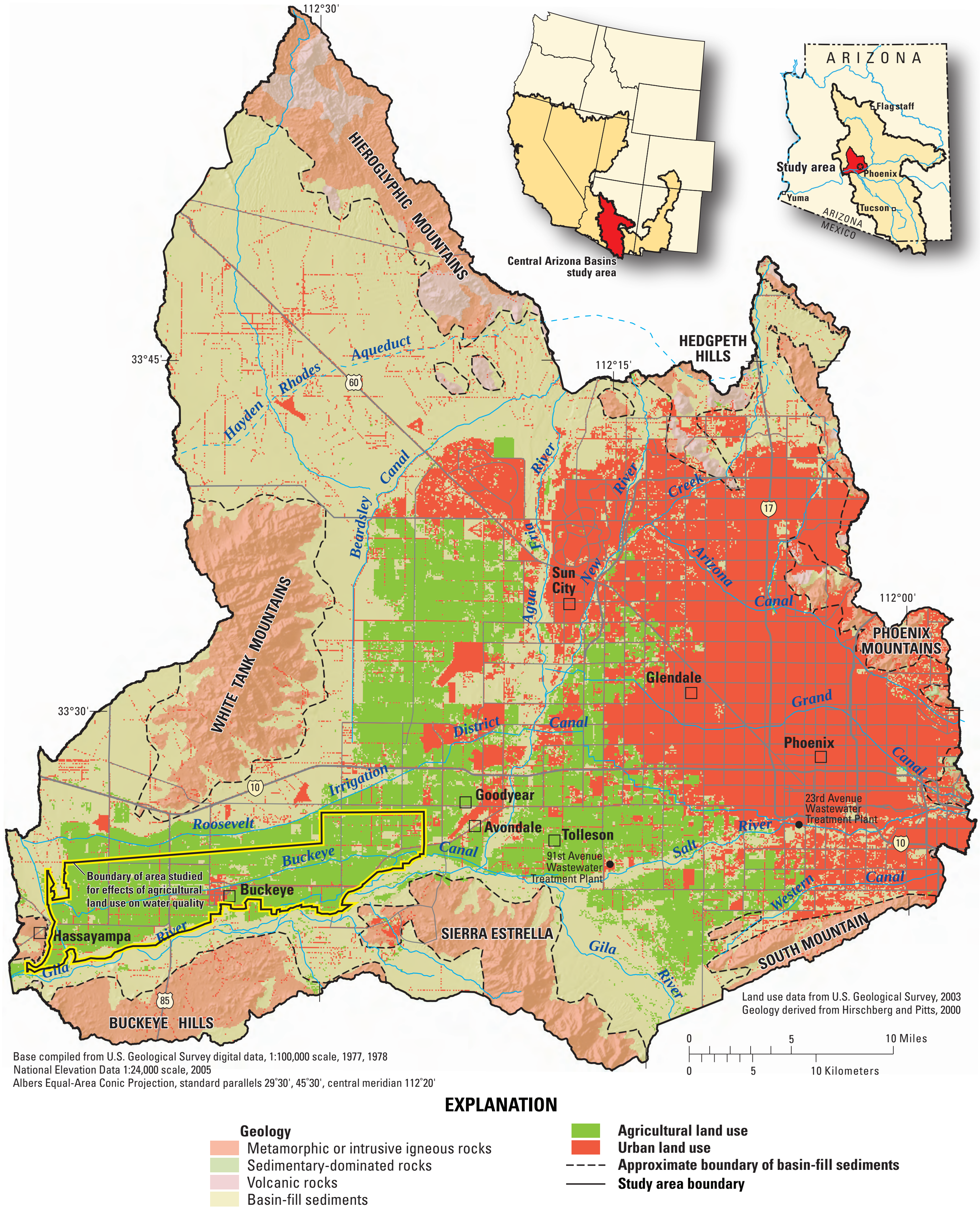


Figure 1. Physiography, land use, and generalized geology of the West Salt River Valley, Arizona.

Water Development History

The Salt River Valley, which contains both the East Salt River Valley and the West Salt River Valley, was originally inhabited by Hohokam Indians from about 300 to 1450 AD. The Hohokam are believed to have been peaceful farmers who built a canal system that traversed about 500 mi and may have supported about 50,000 people (Salt River Project, 2006). The canals laid dormant over the next about 400 years until the 1860s, when pioneers settled in the area and established farming communities. These farmers developed their canals generally using the same routes laid out by the Hohokam canals, and for the most part these canals are still used today (Turney, 1929; Salt River Project, 2006). Fruits and vegetables were grown to support mining communities and the U.S. Cavalry elsewhere in central Arizona. By about 1885, approximately 60,000 acres were irrigated under the Arizona Canal System (Davis, 1897). The population in Phoenix and surrounding rural communities grew to about 16,000 by 1900 (Sargent, 1988).

Water-resources development in the early part of the 20th century allowed for expansion of agricultural lands. Construction of large reservoirs on the Agua Fria, Salt, and Verde Rivers outside the Salt River Valley between 1903 and 1946 provided the capacity to store up to 2.18 million acre-ft of surface water (table 2 in Cordy and others, 1998). Water stored from winter storm runoff in the mountains was later released and diverted into canals (Arizona, Grand, and Western Canals, fig. 1) during the spring and summer months for delivery to agricultural lands in the Salt River Valley.

The early part of the 20th century was also an important period of growth in groundwater development in the Salt River Valley. Groundwater was used when and where surface water was unavailable. Prior to 1920, groundwater withdrawals were less than about 60,000 acre-ft/yr (Anning and Duet, 1994). Significant groundwater withdrawals, however, began with the onset of widespread use of high-capacity turbine pumps in the 1920s (Edmonds and Gellenbeck, 2002). Withdrawals for the entire Salt River Valley (both the East and West parts) steadily increased from about 95,000 acre-ft/yr in 1920 to about 2.3 million acre-ft/yr in the 1950s, after which withdrawal rates began to decline. Withdrawals in 1990 were about 1.1 million acre-ft for the entire valley (Anning and Duet, 1994).

In the later part of the 20th century, the consequences of pumping groundwater at such large rates as those in the 1950s were recognized, and solutions were pursued. For most of the West Salt River Valley, the withdrawals caused water-level declines of more than 50 ft, and in some areas declines were more than 300 ft (Anderson and others, 1992). The lower water levels have resulted in increased pumping costs, and in addition, water depletion has led to aquifer compaction and land subsidence. In an area east of the White Tank Mountains and north of the Interstate-10 freeway, the land surface had subsided as much as 18 ft by 1995 and resulted in a flow reversal in part of the Dysart Drain, a flood drainage canal (Schumann, 1995).

Several water management actions were taken to reduce pumping and its associated problems such as groundwater storage depletion, land subsidence, and increased pumping costs. These include importing surface water, artificial recharge, and use of treated wastewater effluent for irrigation. In 1980, the Groundwater Management Code was passed by the Arizona Legislature to eliminate severe groundwater overdraft and to provide a means for allocating Arizona's limited groundwater resources. This legislation established the Arizona Department of Water Resources and the Phoenix Active Management Area, and contained regulations that encouraged use of Central Arizona Project (CAP) water to reduce groundwater overdraft. The CAP is a series of aqueducts that provide a means to import Colorado River water to central Arizona. CAP deliveries to the West Salt River Valley began in 1985, and by 2005 deliveries were about 222,000 acre-ft/yr (Central Arizona Project, 2006a).

In 1993, the legislature created the Central Arizona Groundwater Replenishment District, the purpose of which is to provide a legal and physical framework for replacing groundwater mined in the Active Management areas, including the West Salt River Valley. In 2005, deliveries to the Agua Fria and Hieroglyphic Mountains recharge facilities were about 52,000 acre-ft (Central Arizona Project, 2006b). Stormwater is also deliberately recharged through thousands of dry wells that are installed in urban areas to enhance infiltration of runoff collected in detention basins.

As the population in the Phoenix metropolitan area grew, so did its "production" of wastewater effluent, which became a valued resource. Most of the effluent from Phoenix and surrounding communities is treated at the 23rd Avenue wastewater treatment plant (WWTP) and at the 91st Avenue WWTP. The Roosevelt Irrigation District Canal (fig. 1) receives effluent from the 23rd Avenue WWTP and this water is applied to crops. Treated municipal wastewater from the 91st Avenue WWTP is released to the Salt River, which flows into the Gila River and is then diverted into the Buckeye Canal (fig. 1) for irrigation of crops. Flow at the head of the Buckeye Canal was 137,500 acre-ft in water year 2000 (Tadayon and others, 2001). Water in the Buckeye Canal, despite its treated-wastewater origin, is often less saline than groundwater from wells in the western part of the valley.

Hydrogeology

The West Salt River Valley is one of several structural basins formed by high-angle faulting of the Basin and Range disturbance (5 to 15 million years ago; Menges and Pearthree, 1989) superimposed on the effects of crustal extension and low-angle detachment faulting of the mid-Tertiary disturbance (15 to 37 million years ago; Dickinson, 1989). Subsidence of the structural basins formed closed drainages that slowly filled with locally derived sediments and evaporite deposits. After subsidence slowed and the basins filled with sediment, streams began to flow through the lowest divides into adjacent basins, and ultimately this process resulted in the integrated drainage system of the Gila River and its tributaries (Damon and others, 1984).

Mountains surrounding the valley are composed primarily of granitic and metamorphic rocks, and secondarily of sedimentary and volcanic rocks (fig. 1; Hirschberg and Pitts, 2000). The mountains generally form barriers to groundwater flow because of the low hydraulic conductivity values of these rocks. A major linear subsurface structure, probably a fault in the crystalline rocks, is aligned with Highway 60 and divides the valley into northeastern and southwestern areas (Brown and Pool, 1989). The northeastern area is characterized by a series of structural blocks tilted to the northeast and trending northwest that are overlain by basin-fill deposits, which are generally less than 2,000 ft thick (fig. 2). Within the southwestern area, the deposits are generally less than 2,000 ft thick in the western part, but increase in thickness to the east to greater than 10,000 ft.

The basin-fill deposits are divided into upper, middle, and lower units (Brown and Pool, 1989). The lower unit was deposited when the basin was closed and subsiding and it consists of playa, alluvial-fan, fluvial, and evaporite deposits. The sediments in the lower unit are generally fine grained, with coarse-grained facies at the basin margins and at depth, and the unit is further divided into upper and lower parts. The thickness of the lower part exceeds 10,000 ft in the center of the basin, whereas the thickness of the upper part is generally less than 1,000 ft. The lower part of the lower unit tends to be more consolidated and the clast type and stratigraphy is more homogeneous than the upper part. Estimated hydraulic conductivity values range from 6 to 14 ft/d for the lower part, and from 3 to 25 ft/d for the upper part (Brown and Pool, 1989). Evaporites were deposited near the center of the southwestern part of the basin in the lower part of the basin fill (Brown and Pool, 1989). Evaporites in the lower part of the lower unit are generally massive and consist of anhydrite, gypsum, and halite, whereas evaporite units in the upper part consist only of gypsum that is interbedded or finely disseminated within the clastic sediments. The Luke Salt Body, the major evaporite deposit in the basin, has a pronounced local effect on the salinity of the groundwater and an indirect effect on the transmissivity of the basin fill (Eaton and others, 1972). Both the upper and lower parts of the lower unit are fully saturated in most of the basin.

The middle unit was deposited when the basin was open and drained by the Agua Fria, Salt, and Gila Rivers (Brown and Pool, 1989). This unit is as much as 800 ft thick near the center of the basin, and includes playa, alluvial-fan, and fluvial deposits of silt, clay, siltstone, and silty sand and gravel. The areal extent of fine-grained sediments increases with depth, and their occurrence is less common in the middle unit than the lower unit. Some lenses or zones in the middle unit with more than 80 percent silt and clay, however, are present in areas near Goodyear and Glendale (Brown and Pool, 1989) and form a confining bed that retards vertical movement of groundwater (Edmonds and Gellenbeck, 2002). Hydraulic conductivity values in the middle unit range from 4 to 60 ft/d. Overdraft of groundwater has significantly dewatered the middle unit in much of the valley, and has completely dewatered it in a large area east of the White Tank Mountains (Brown and Pool, 1989).

The upper unit was deposited primarily by the Agua Fria, Salt, and Gila Rivers, as well as by local tributaries. The unit comprises channel, floodplain, and alluvial-fan deposits consisting largely of gravel, sand, and silt. The thickness of the upper unit ranges from 200 ft or less near the basin margins to about 400 ft near the confluence of the Salt and Gila Rivers. Hydraulic conductivity values in the upper unit are much higher than those of the lower and middle unit, and range from 180 to 1,700 ft/d (Brown and Pool, 1989). Overdraft of groundwater has dewatered most of the upper unit over large parts of the valley (Brown and Pool, 1989).

Conceptual Understanding of the Groundwater Flow System

The groundwater system was under steady-state conditions prior to the beginning of water development by settlers in the 1860s (Corkhill and others, 1993). Long term recharge and discharge of the basin-fill aquifer were in balance with each other and equal to about 68,000 acre-ft/yr (fig. 3; table 1). Most of the recharge was from streamflow infiltration and from subsurface inflow through basin-fill aquifers of adjacent basins (table 1). Most of the discharge took place through evapotranspiration of shallow groundwater and by subsurface outflow northwest of the Buckeye Hills (table 1). Groundwater movement prior to water development is presumed to have been primarily horizontal, and on the basis of early water-level maps, the flow was toward and along the Salt and Gila Rivers (fig. 4*A*; Corkhill and others, 1993; Lee, 1905).

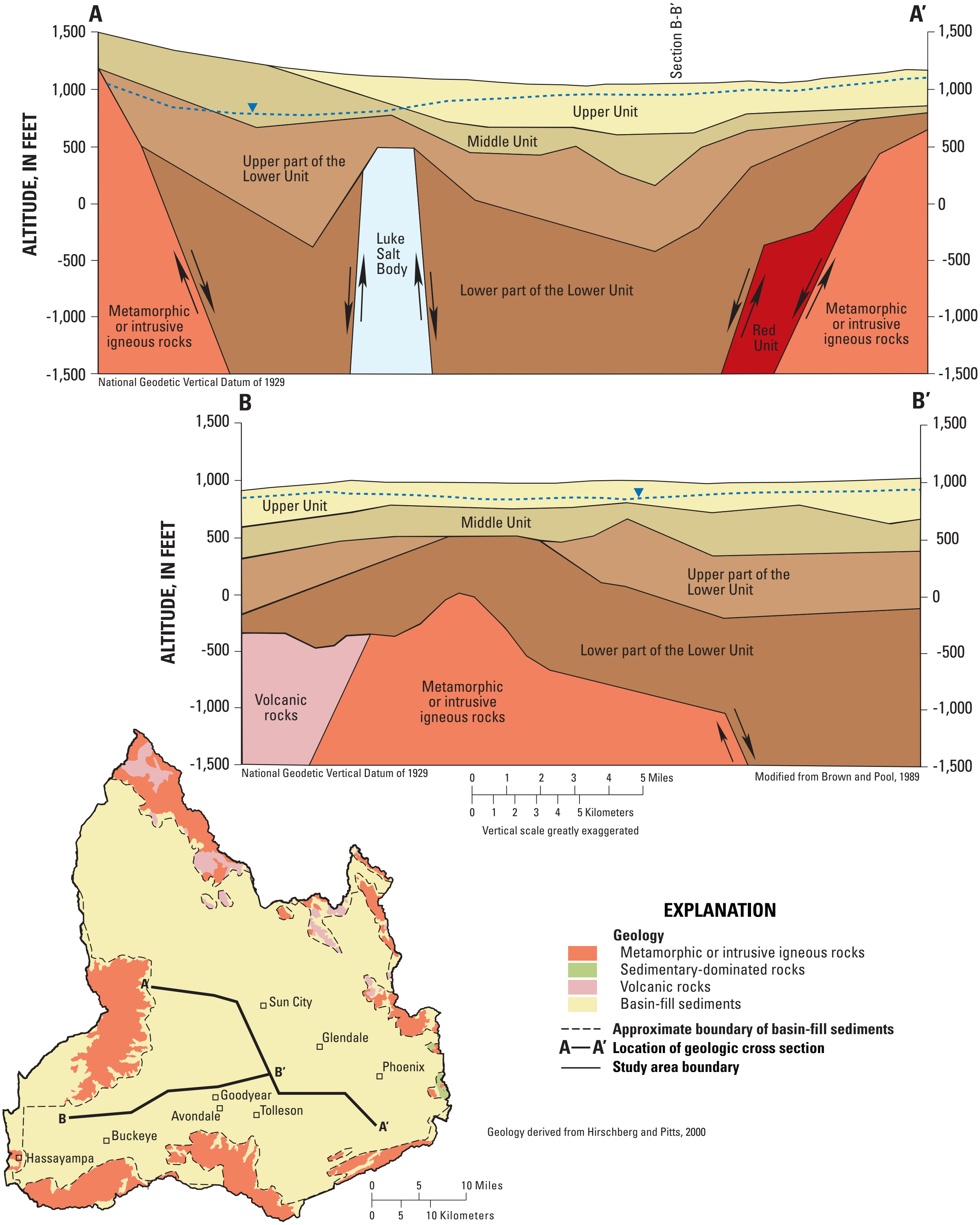


Figure 2. Generalized geologic cross sections of the West Salt River Valley, Arizona.

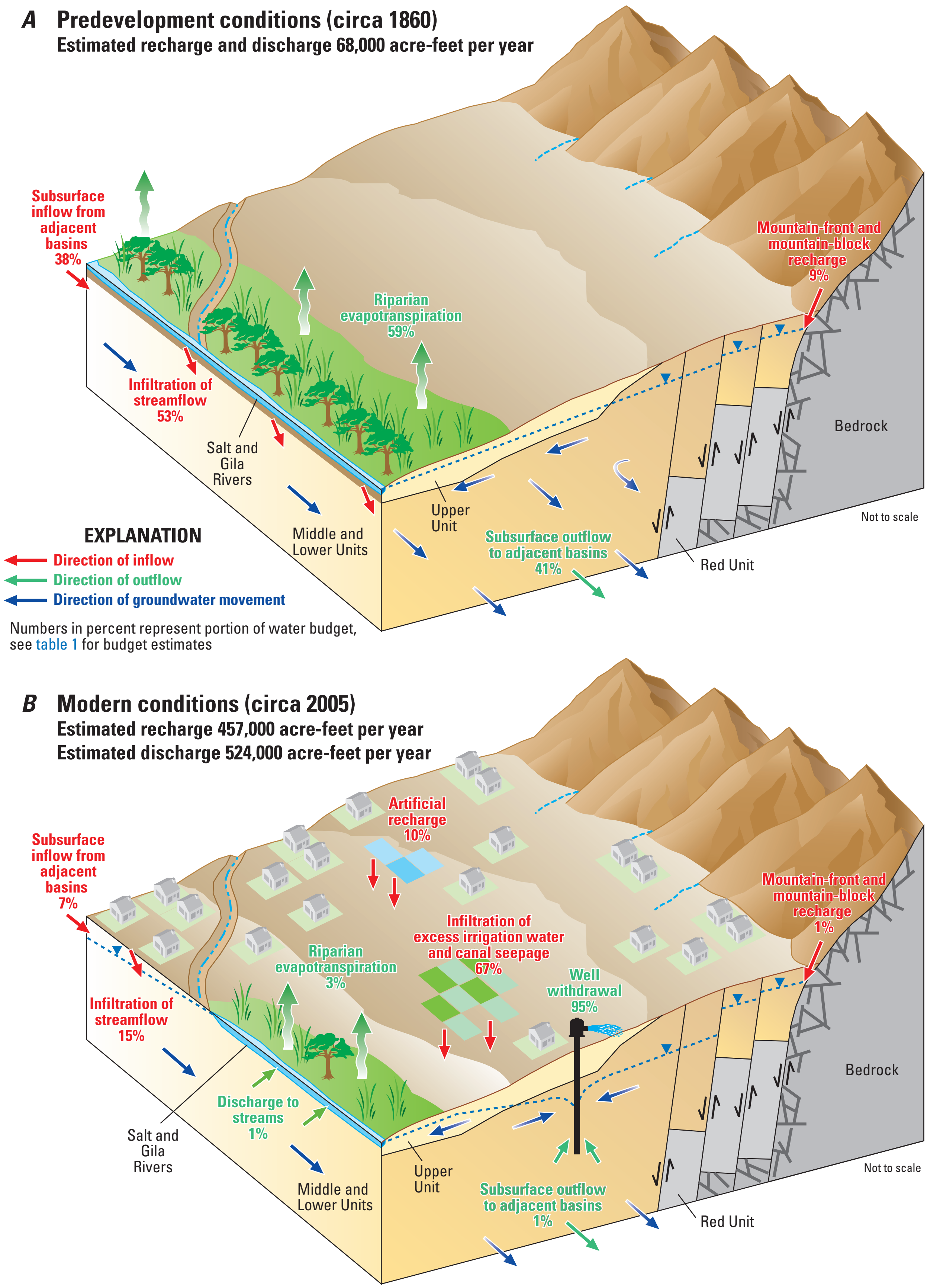


Figure 3. Generalized diagrams for the West Salt River Valley, Arizona, showing components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin-fill aquifer in the West Salt River Valley, Arizona, under predevelopment and modern conditions.

[All values are in acre-feet per year and are rounded to the nearest thousand. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3]

Budget component	Predevelopment conditions, before 1860	Modern conditions, 2005	Change from predevelopment to modern conditions
Estimated recharge			
Subsurface inflow from adjacent basins[1]	26,000	30,000	4,000
Mountain-block and mountain-front recharge[2]	6,000	6,000	0
Infiltration of precipitation on basin[1]	0	0	0
Infiltration of streamflow[1]	36,000	68,000	32,000
Infiltration of excess irrigation water and canal seepage[1]	0	308,000	308,000
Artificial recharge[1]	0	45,000	45,000
Total recharge	68,000	457,000	389,000
Estimated discharge			
Subsurface outflow to adjacent basins[1]	28,000	7,000	-21,000
Evapotranspiration[1]	40,000	15,000	-25,000
Discharge to streams[1]	0	5,000	5,000
Discharge to springs and drains[1]	0	0	0
Well withdrawals[1]	0	497,000	497,000
Total discharge	68,000	524,000	456,000
Estimated change in storage (recharge - discharge)	0	-67,000	-67,000

[1]Predevelopment conditions from Freethey and Anderson (1986), and modern conditions from Corkhill and others (2004).

[2]Predevelopment and modern conditions from Freethey and Anderson (1986).

The groundwater flow system has been substantially altered by water development related stresses such as withdrawals from regional pumping centers and recharge supplied by canal seepage and irrigation losses on croplands and turf (table 1; fig. 4*B*; Corkhill and others, 1993; Corkhill and others 2004). Estimated recharge for the modern (2005) water budget is 457,000 acre-ft/yr, which is nearly seven times the estimated recharge under predevelopment conditions. Most of this increase is due to recharge from excess irrigation water (table 1). Estimated discharge for the modern water budget is 524,000 acre-ft/yr, which is nearly eight times the discharge under predevelopment conditions. Nearly all of this increase is due to withdrawals through wells (table 1). Annual change in storage is assumed to be zero under predevelopment conditions but an estimated 67,000 acre-ft/yr are lost under modern conditions. Use of surface water and imported water from the Central Arizona Project for municipal and agricultural purposes reduces the need for groundwater withdrawals (aquifer discharge) and provides a significant portion of the irrigation losses that recharge the aquifer, both of which help mitigate storage losses.

Groundwater withdrawals in the valley through 1988 depleted storage by about 23 million acre-ft and have resulted in large water-level declines in most areas (fig. 4*C*; Corkhill and others, 1993). Whereas under predevelopment conditions groundwater flow was predominantly horizontal, pumping has created cones of depression within which the flow is vertically downward. In some areas, the direction of groundwater flow has changed and is now toward large depressions in the water table caused by regional pumping centers, such as the one north of the Arizona Canal and the one west of Sun City (fig. 4*B*; Corkhill and others, 1993). Depth to groundwater under modern conditions varies from less than 100 ft in the southern part of the basin, along the Salt and Gila Rivers, to greater than 400 ft west of Sun City (fig. 4*D*).

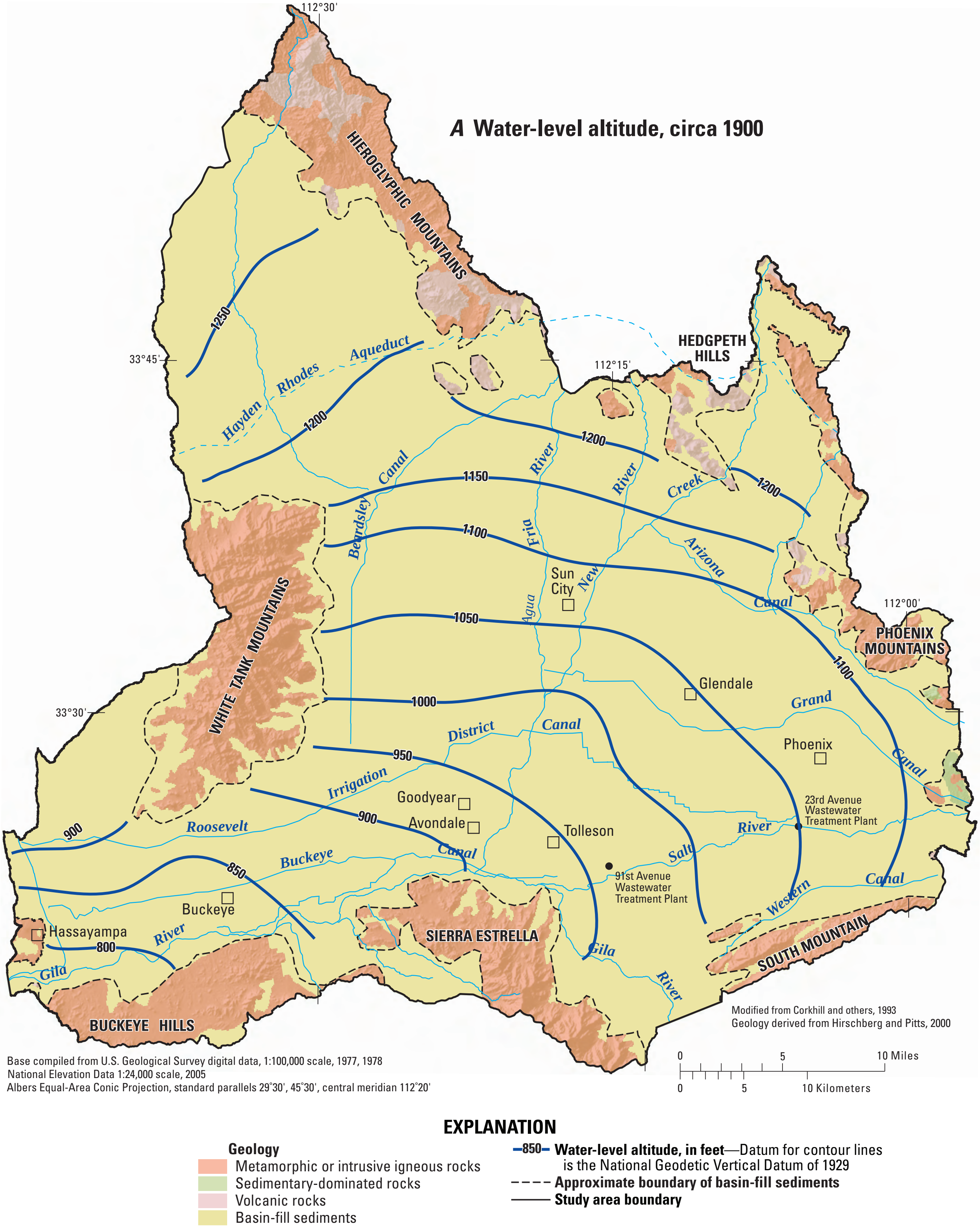


Figure 4. Water levels in the basin-fill aquifer of the West Salt River Valley, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1985. (*C*) Water-level change for 1900–1980. (*D*) Depth to water, 1983.

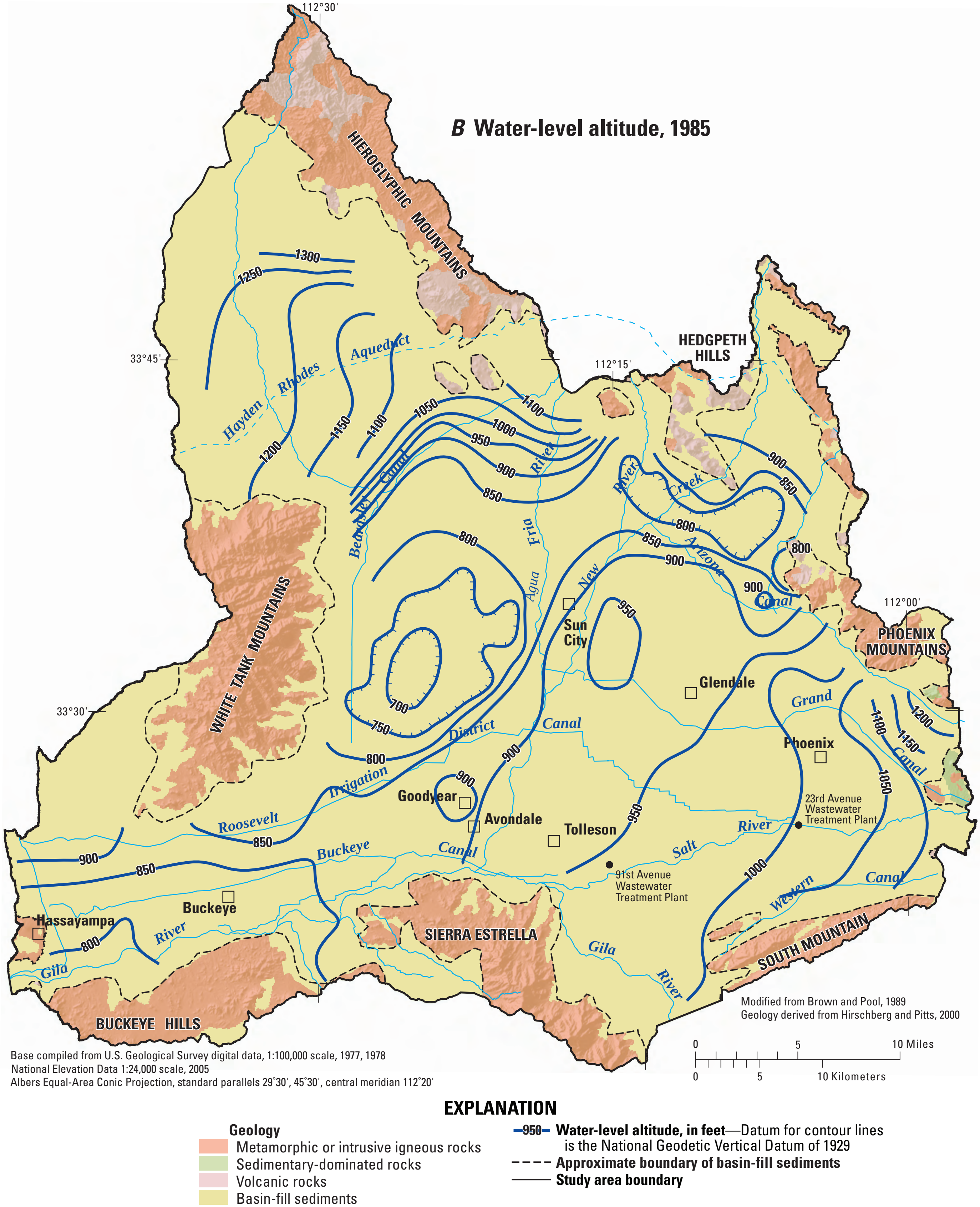
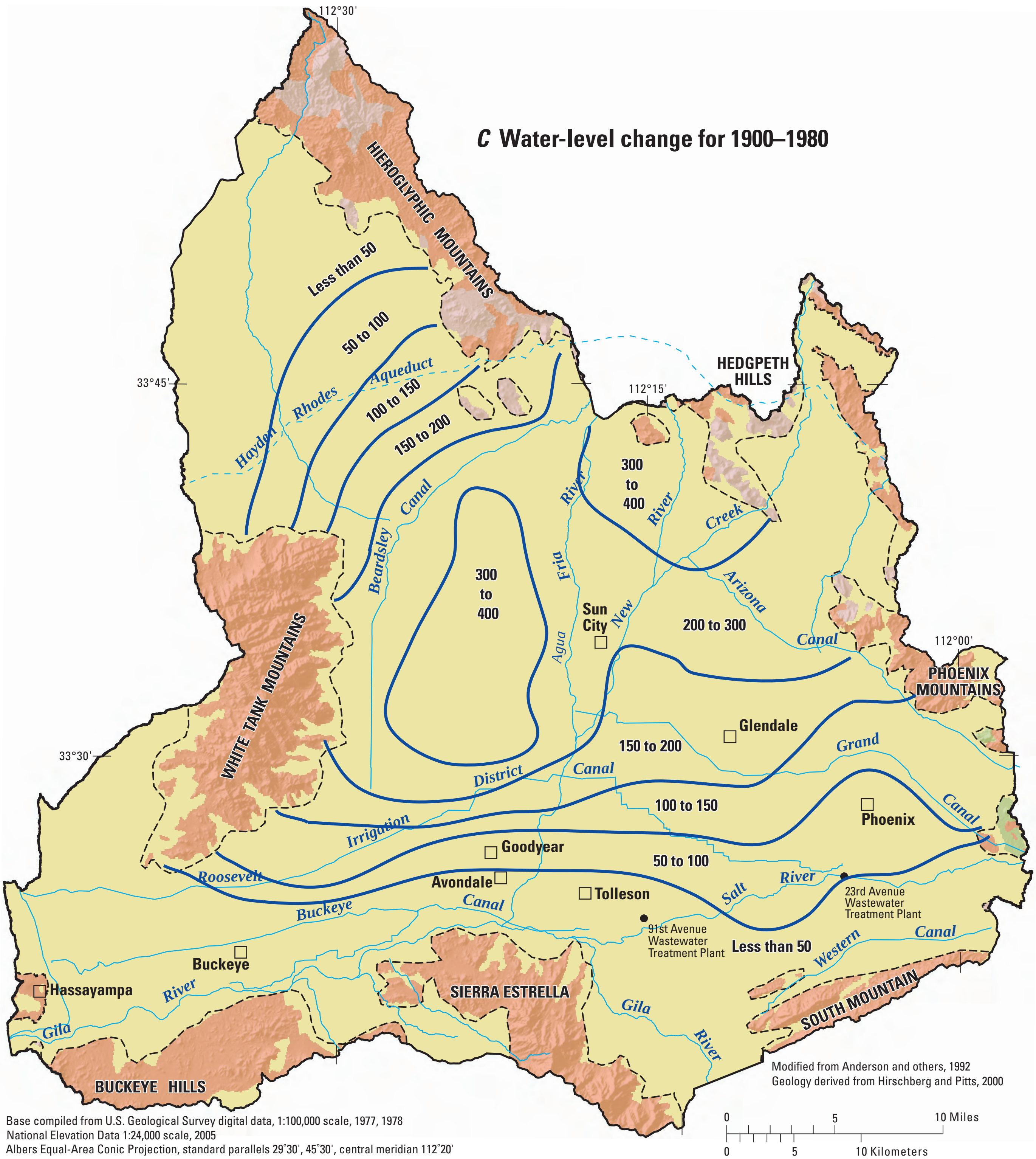


Figure 4. Water levels in the basin-fill aquifer of the West Salt River Valley, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1985. (*C*) Water-level change for 1900–1980. (*D*) Depth to water, 1983—Continued.



EXPLANATION

Geology
- Metamorphic or intrusive igneous rocks
- Sedimentary-dominated rocks
- Volcanic rocks
- Basin-fill sediments

Approximate boundary for listed water-level decline, in feet
Approximate boundary of basin-fill sediments
Study area boundary

Figure 4. Water levels in the basin-fill aquifer of the West Salt River Valley, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1985. (*C*) Water-level change for 1900–1980. (*D*) Depth to water, 1983—Continued.

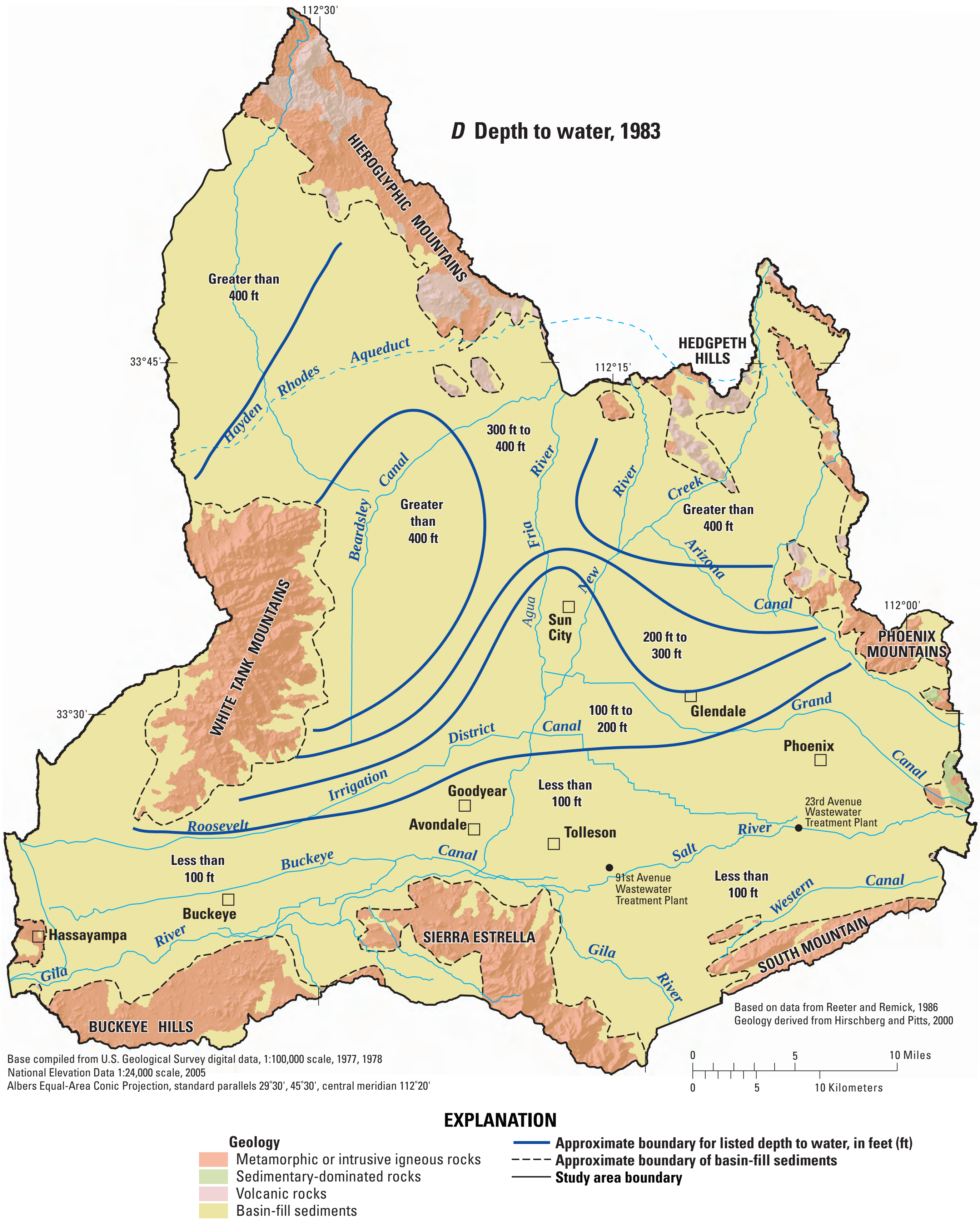


Figure 4. Water levels in the basin-fill aquifer of the West Salt River Valley, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1985. (*C*) Water-level change for 1900–1980. (*D*) Depth to water, 1983—Continued.

Effects of Natural and Human Factors on Groundwater Quality

Groundwater quality in the West Salt River Valley is affected by the hydrogeology of the basin-fill aquifer, as well as land and water use on the ground surface (Edmonds and Gellenbeck, 2002; and Gellenbeck and Anning, 2002). These findings are based on analyses of data collected from 1996–98 as part of the following studies by the U.S. Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) Program: (1) a basin-wide assessment of groundwater-quality conditions in the basin-fill aquifer, and (2) an assessment of groundwater-quality conditions specific to an area of predominantly agricultural land use (Edmonds and Gellenbeck, 2002; and Gellenbeck and Anning, 2002). Results of these studies are described in the remainder of this section and are integrated with findings from other water-quality investigations.

Groundwater-Quality Conditions Across the Valley

The basin-wide assessment of groundwater quality consisted of an analysis and interpretation of the chemical characteristics of samples from 35 wells completed in the basin-fill aquifer. The wells were selected, through a stratified-random sampling design, to represent the developed part of the aquifer. Most of the wells were used for domestic or commercial purposes and were greater than 100 ft deep. The occurrence, concentrations, and distribution of dissolved solids, nutrients, trace elements, pesticides, and volatile organic compounds (VOCs) in the water in the basin-fill aquifer across the valley are described below.

Concentrations of dissolved solids vary across the basin, and were higher in water from wells south of the Interstate-10 freeway (median = 790 mg/L) than in water from wells north of the freeway (median = 316 mg/L). Dissolved-solids concentrations in water from wells completed in the shallowest parts of the aquifer (less than 350 ft below land surface) were higher (median = 745 mg/L) than water from wells completed in deeper parts of the aquifer (median = 348 mg/L). Other investigations also found that dissolved-solids concentrations are lower in the northern part of the basin than in the southern part, near Buckeye (Thompson and others, 1984; fig. 5*A*), which may be due, in part, to groundwater in the southern part being affected by recharge of excess irrigation water (Brown and Pool, 1989). Brown and Pool found that concentrations of dissolved solids in the upper unit of the basin-fill aquifer generally were higher than those in the middle and lower units, and that concentrations in the middle and lower units are generally similar, except where the presence of evaporites increased concentrations in the lower unit. Eaton and others (1972) found dissolved-solids concentrations in groundwater were affected by the Luke Salt Body, and Brown and Pool (1989) noted that concentrations near the body range from 1,000 to 100,000 mg/L.

Median concentrations of dissolved nitrate (as nitrogen) and dissolved oxygen were 2.7 mg/L and 4.1 mg/L, respectively. Sources of nitrogen include dairies and feedlots, wastewater treatment plants, agricultural activities (manure from livestock, and application of fertilizers), and natural sources—decomposed vegetation or nitrogen fixed by bacteria associated with desert legumes (Gellenbeck, 1994). Elevated concentrations of nitrate and dissolved oxygen in the basin-fill aquifer of West Salt River Valley are possibly due to a lack of organic matter and associated biological processes in the aquifer matrix that typically consume oxygen and nitrate. On the basis of positive correlations with oxygen isotope data, Edmonds and Gellenbeck (2002) found that elevated concentrations of dissolved solids and nitrate resulted from the application of nitrogen fertilizers to crops and evaporation during irrigation of crops and landscaping. High nitrate concentrations detected in the samples corroborate the findings by Long and others (1997), who found that nitrate concentrations were correlated with dissolved-solids concentrations and who estimated that during the period 1986–90, nitrate exceeded the U.S. Environmental Protection Agency's (USEPA) primary drinking-water standard for nitrate of 10 mg/L (U.S. Environmental Protection Agency, 2009) in groundwater beneath a 190-mi^2 area near Phoenix and Glendale, and an 85-mi^2 area near Buckeye (fig. 5*B*).

Arsenic and uranium also are present in the water of the basin-fill aquifer. Arsenic was detected in samples from each of the 35 wells, and the median arsenic concentration was 6 µg/L. Concentrations of arsenic in samples from 11 wells (31 percent) exceeded the USEPA primary drinking-water standard for arsenic of 10 µg/L (fig. 6). The source of the arsenic is presumed to be minerals in the basin-fill deposits that originated from hydrothermal sulfide and arsenide deposits in the surrounding mountains (Robertson, 1991). The median uranium concentration was 3 µg/L, and concentrations for 4 wells (11 percent) exceeded the USEPA primary drinking-water standard for uranium of 30 µg/L (fig. 6).

Edmonds and Gellenbeck (2002) found that the water in 12 of the wells (34 percent) they sampled contained tritium (fig. 6), which indicated that part of the aquifer in which the wells were completed contained a component of groundwater that was recharged after 1953 (See Section 1 of this report for a discussion of groundwater age and environmental tracers). Organic compounds generally related to human activities on the land surface, such as pesticides and VOCs, were detected in samples from 11 of these 12 wells. This high detection rate of organic compounds for recently recharged groundwater emphasizes the susceptibility of the basin-fill aquifer to contamination.

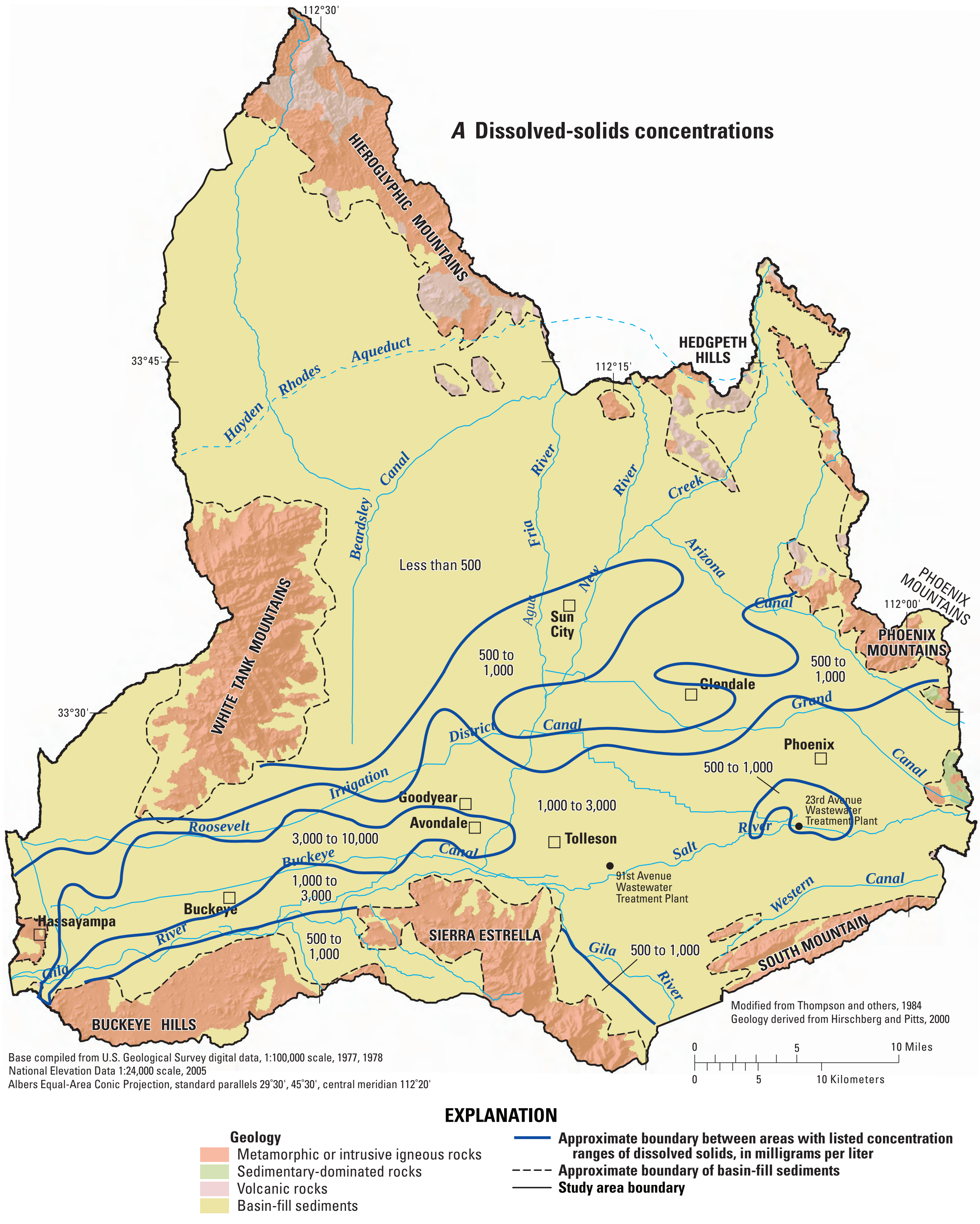
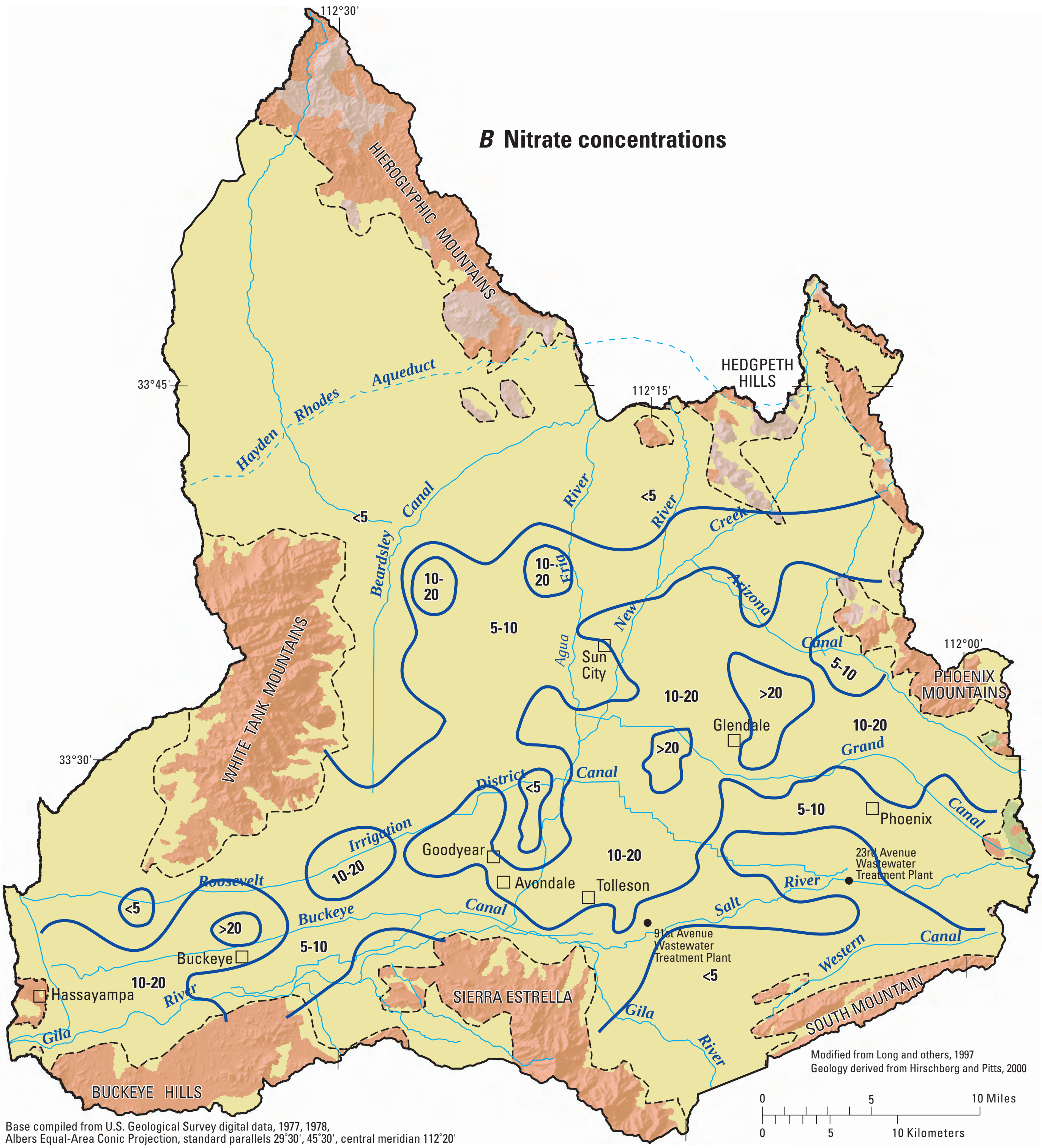


Figure 5. Concentrations of (*A*) dissolved solids and (*B*) nitrate in the basin-fill aquifer in the West Salt River Valley, Arizona.



EXPLANATION

Geology
- Metamorphic or intrusive igneous rocks
- Sedimentary-dominated rocks
- Volcanic rocks
- Basin-fill sediments

Approximate boundary between areas with listed concentration ranges of nitrate (as nitogen), in milligrams per liter— <, less than; >, greater than

Approximate boundary of basin-fill sediments

Study area boundary

Figure 5. Concentrations of (*A*) dissolved solids and (*B*) nitrate in the basin-fill aquifer in the West Salt River Valley, Arizona—Continued.

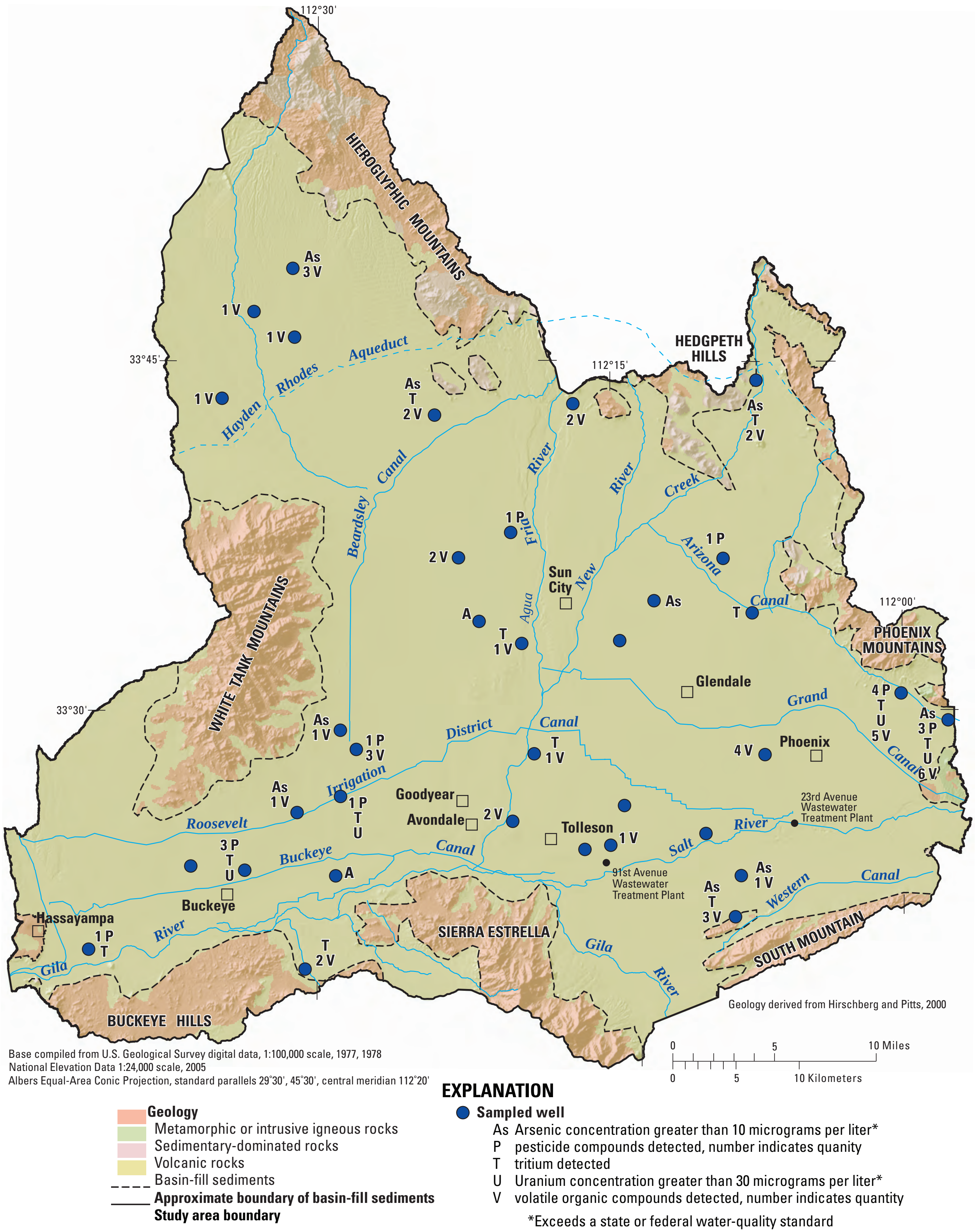


Figure 6. Elevated concentrations and detections of selected compounds in samples from the basin-wide water-quality assessment of groundwater in the West Salt River Valley, Arizona, 1996–98.

Of the 35 wells sampled in the study, one or more pesticides were detected in samples from 8 wells (fig. 6). Detected compounds include one insecticide degradation product: DDE, and seven herbicides and herbicide degradation products: atrazine, simazine, deethylatrazine, prometon, acetochlor, S-ethyl dipropylthiocarbamate, and triallate. None of the compounds were present at concentrations greater than any USEPA drinking-water standard. The spatial distribution of the detections indicates that pesticides applied at the land surface reached the groundwater in both agricultural and nonagricultural land-use settings. Also, pesticide detections did not directly correlate with the pesticide application rates in the valley (Gellenbeck and Anning, 2002).

Thirty-three detections of 18 different VOCs were identified in samples from 21 (70 percent) of the 30 wells that had VOC data (fig. 6). Detected compounds include:

1,2,4-trimethylbenzene (8 samples)	1,1-dichloroethane (2 samples)
Chloromethane (4 samples)	Methyl *tert*-butyl ether (MTBE; 2 samples)
Carbon disulfide (4 samples)	Benzene (1 sample)
Iodomethane (4 samples)	Trichlorofluoromethane (CFC-11; 1 sample)
Trichloromethane (chloroform; 3 samples)	1,1,1-trichloroethane (TCA; 1 sample)
Trichloroethene (TCE; 3 samples)	1,1,2-trichloro-1,2,2 trifluoroethane (1 sample)
1,4-dichlorobenzene (3 samples)	1-chloro-2-methyl benzene (1 sample)
Bromodichloromethane (2 samples)	1,1- dichloroethene (1 sample)
Tetrachloroethylene (PCE; 2 samples)	Acetone (1 sample)

The detected VOCs all have potential anthropogenic sources; however, a few of the detections may not necessarily indicate contamination of the aquifer due to human activities. Chloromethane and carbon disulfide have human sources, but could also have been produced by fungi and enter groundwater from that natural source (Gellenbeck and Anning, 2002). Also, the presence of trichloromethane and bromodichloromethane in groundwater samples can result from chlorination of a well as a treatment for bacteria and odors, and may not represent aquifer contamination. That said, the large variety of VOCs and the large area where samples contained VOCs, indicate that groundwater in the West Salt River Valley is affected by human activities (Gellenbeck and Anning, 2002).

Groundwater-Quality in an Agricultural Land Use Setting

In addition to assessing groundwater-quality conditions in the West Salt River Valley on a basin-wide scale, NAWQA investigators also sampled wells in an agricultural land use setting to characterize the effects of that land use on water quality. That assessment consisted of an analysis of samples from 9 monitoring wells in an agricultural area near Buckeye in the southwestern part of the valley (fig. 1). The monitoring wells were completed within the top 10 ft of the water table in the basin-fill aquifer, where the most recent recharge from excess irrigation water is expected to accumulate. Tritium activity levels in samples from the 9 wells were 15 pCi/L or greater, which confirmed the representation of recent recharge by the samples.

For several constituents, concentrations were higher and detections more frequent in samples from wells in the agricultural area than in samples from wells included in the basin-wide assessment. For example, median concentrations of dissolved solids, nitrate, fluoride, arsenic, barium, chromium, and strontium for wells in the agricultural area were greater than median concentrations for wells in the basin-wide assessment (Edmonds and Gellenbeck, 2002). Concentrations of these inorganic constituents were higher, in part, owing to evapotranspiration of the irrigation water before it percolated to the shallow groundwater body.

Pesticides were detected in samples from all nine monitoring wells, clearly indicating that pesticides are reaching the shallow groundwater and that the agricultural land use is affecting water quality (Gellenbeck and Anning, 2002). Samples were collected from each well both during and after the irrigation season. Ten different pesticides were detected during the irrigation season, whereas only seven different pesticides were detected afterward, which may reflect the degradation of pesticide compounds following the irrigation season.

The most commonly detected pesticides in samples from wells in the agricultural area were atrazine, which was detected in water from all nine wells, and deethylatrazine, a degradation product of atrazine, which was detected in water from 8 wells. Other detected pesticides include simazine, DDE, diuron, dieldrin, chlorpyrifos, acetochlor, prometon, metribuzin, and trifluralin. The compound DDE (1,1-dichloro-2,2-bis(chlorophenyl)ethylene), is a degradation product of DDT (1,1,1-trichloro-2,2-bis(chlorophenyl)ethane), an insecticide used in agricultural areas from 1944 until its use was banned in Arizona in 1965. This compound and its degradation products are highly persistent in the soil, have a low solubility in water, and, over long periods of time, may leach into the groundwater.

At least one VOC was detected in each of the nine wells sampled in the agricultural area, and a total of 20 VOCs were detected amongst samples from all the wells. Trichloromethane, a byproduct of the chlorination of drinking water also known as chloroform, was the most commonly detected VOC with occurrences in every sample during both sampling periods. Land in the agricultural area is irrigated with treated effluent from the Phoenix wastewater treatment plants that process chlorinated city water. Trichloromethane also can enter the groundwater in recharge of lawn irrigation water, leaking water-supply mains, and sewers. The presence of trichloromethane in the shallow groundwater in this area indicates that the water is affected by human activities.

The second most commonly detected VOC in samples from the agricultural area wells was tetrachloroethylene (PCE), which was detected in water from 5 of the 9 wells. PCE was detected in samples collected from three wells during both sampling periods, and trichloroethene (TCE) also was detected at one of these same wells. All but one of the detections were at concentrations below the minimum reporting level for non-estimated values in the laboratory analysis. Four of the five wells in which samples contained PCE or TCE are downgradient from a Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) site near Goodyear, where the groundwater is known to be contaminated with TCE and PCE (Gellenbeck and Anning, 2002). Another possible source of the PCE and TCE detections in these 5 wells is the local use of these compounds as solvents.

Relation of Groundwater Quality to Hydrogeology, Water Use, and Land Use

Edmonds and Gellenbeck (2002) developed a relation of groundwater quality to hydrogeology, water use, and land use by comparing dissolved-solids concentrations, nitrate concentrations, and pesticide detections for 5 different classes of wells sampled in the basin-wide assessment and the agricultural land use study, and for 15 other wells sampled as part of the NAWQA Program (table 2; fig 7). Water-quality conditions in wells in class A, which are in areas with minimal agricultural or urban development, served as indicators of reference, or background, conditions. The median dissolved-solids concentration (257 mg/L) and median nitrate concentration (1.7 mg/L) for these wells were the lowest amongst the different classes. In addition, no pesticides were detected in any samples from these wells.

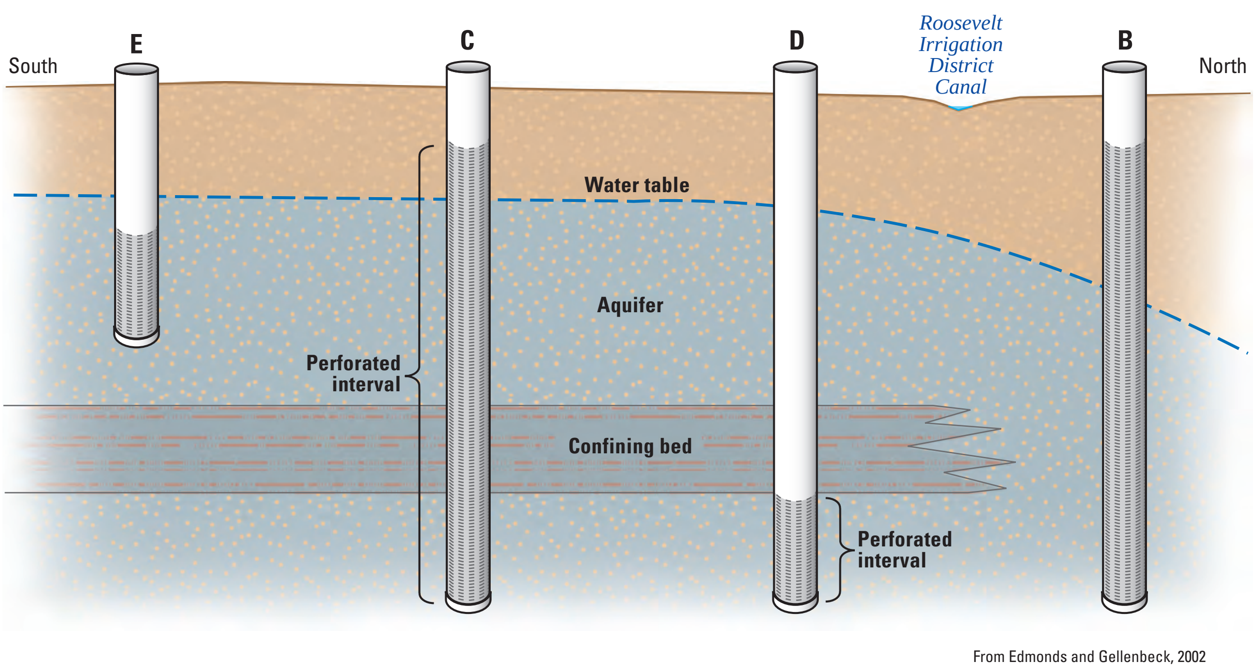


Figure 7. Construction typical of classified wells, West Salt River Valley, Arizona. (Class A wells not shown.)

Table 2. Summary of physical and water-quality characteristics for five classes of wells in the West Salt River Valley, Arizona.

[See figure 7 for typical well construction of classes B, C, D, and E. Data from Edmonds and Gellenbeck, 2002]

Well class and number of wells	A. 8 wells	B. 13 wells	C. 11 wells	D. 18 wells	E. 9 wells
Physical characteristics					
General location	Throughout the valley, but generally along margins	North of the Roosevelt Irrigation District Canal or east of the Agua Fria River	Southwestern part of valley	Southeastern and southwestern part of valley	Southwestern part of valley
Land use and irrigation supply	Undeveloped; no irrigation	Agricultural areas irrigated with groundwater or Agua Fria River water	Agricultural areas served by the Buckeye and Roosevelt Irrigation District Canals, which contain pumped groundwater, surface water and treated municipal effluent	Agricultural and urban areas; unspecified water supplies	Agricultural areas irrigated with groundwater and water from the Buckeye and Roosevelt Canals, which contain treated municipal effluent
Hydrogeology, confinement, and well perforations	Unspecified	No appreciable amounts of fine grained sediments penetrated by well, or perforations above any fine-grained confining beds	Perforations are above fine-grained confining beds of middle unit	Perforations are completely below fine-grained sediment beds of middle unit with unperforated casing extending from land surface through these beds	Monitoring wells were constructed to sample the top of the aquifer, with perforations above fine-grained confining beds of middle unit
Water-quality characteristics					
Median dissolved-solids concentration, milligrams per liter	257	668	3,050	747	3,350
Median nitrate concentration, milligrams per liter	1.7	11.4	19.0	2.0	16.9
Pesticide detections	0	11	35	0	78
Number of wells where pesticides were detected	0	6	10	0	9
Number of pesticide compounds detected	0	7	11	0	10

Although wells in class D are in both agricultural and urban areas, analyses of samples did not reflect any effects of recent recharge by excess irrigation water, probably because the wells are perforated below poorly permeable, fined-grained confining beds in the middle unit of the basin-fill aquifer (table 2, fig. 7). The lack of tritium detections in samples from these wells confirms this lack of recent recharge. Statistical analyses of the data indicated that median concentrations of dissolved solids (747 mg/L) and nitrate (2.0 mg/L) were comparable to those for samples from class A wells, but significantly less than those for class C and E wells which receive recharge from excess irrigation water. Brown and Pool (1989) also found that dissolved-solids concentrations were lower in the water in deeper aquifer units than in water in the uppermost saturated unit. No pesticides were detected in any samples from class D wells, further indicating the lack of effects of recharge from excess irrigation water.

Wells in classes C and E are in agricultural areas served by the Buckeye and Roosevelt Irrigation District canals, which convey pumped groundwater, surface water, and treated municipal effluent to agricultural fields. Fine-grained sediments of the middle unit of the basin-fill aquifer form confining beds in this area; however, in contrast to wells in class D, wells in classes C and E are perforated above these confining beds. Wells in class E are in agricultural areas and were designed to sample the top 10 ft of the basin-fill aquifer; results of analyses of samples from these wells were discussed above. As a consequence of being perforated above the confining beds, class E wells yield water in which median concentrations of dissolved solids (greater than 3,000 mg/L) and nitrate (greater than 16 mg/L) were higher than in water from well classes A, B, and D (table 2). In addition, pesticide detections, the number of wells in which pesticides were detected, and the number of compounds detected in samples from wells in classes C and E, were higher than those for wells in classes A, B, and D (table 2).

Wells in class B were in agricultural areas outside the Buckeye and Roosevelt Irrigation Districts that lack confining conditions created by the presence of fine-grained sediments. Median concentrations of dissolved solids and nitrate, and pesticide detections for wells in class B generally are in between the concentrations and detections frequency for well classes A and D, which were not affected by recharge of excess irrigation water, and wells in classes C and E, which were affected by recharge of excess irrigation water. The water table generally is deeper in wells in class B than in wells in classes C and E. The deeper water table in wells in class B may be the reason for the lesser effects of recharge of excess irrigation water on groundwater quality in these wells as compared to the effects detected in samples from wells in classes C and E.

Summary

The West Salt River Valley in central Arizona is an arid basin with significant water-resources development that supports agricultural and urban activities. The mountains surrounding the valley are composed primarily of granitic and metamorphic rocks, and the valley is a structural basin filled with consolidated to unconsolidated sediments. Where saturated, these sediments form the basin-fill aquifer. Water demands for municipal and agricultural needs are met using a variety of water sources, including groundwater from the basin-fill aquifer; surface water from the Agua Fria, Gila, Salt, and Verde Rivers, most of which is stored in reservoirs outside the valley; imported water from the Central Arizona Project; and recycled water from municipal wastewater-treatment plants.

The groundwater system is considered to have been under steady-state conditions prior to the beginning of water development by settlers in the 1860s. Groundwater fluxes were estimated to have been about 68,000 acre-ft/yr, and groundwater movement was primarily horizontal and towards and along the Salt and Gila Rivers. The natural groundwater flow system has been substantially altered by water-development related stresses such as withdrawals from regional pumping centers and recharge supplied by canal seepage and infiltration of excess irrigation water applied to croplands and turf. Estimated recharge under modern-day development conditions is estimated at about 457,000 acre-ft/yr, which is nearly seven times that before development began. Most of this increase is due to recharge from excess irrigation water. Estimated discharge under modern-day development is 524,000 acre-ft/yr, which is nearly eight times the rate for predevelopment conditions. Most of this increase is due to groundwater pumping, and in some areas, groundwater now flows towards large depressions in the water table caused by withdrawals at regional pumping centers.

Water-quality issues for the basin-fill aquifer include elevated concentrations of dissolved solids, nitrate, and arsenic; and the presence of pesticides and volatile organic compounds (VOC) in groundwater in parts of the valley. The occurrence and concentrations of these water-quality constituents result from natural and human-related factors such as hydrogeology, water use, and land use. Examples of natural factors that affect groundwater quality include occurrence of evaporites in the basin-fill deposits that elevate concentrations of dissolved solids, natural nitrogen fixation that elevate concentrations of nitrate, and geological sources and geochemical reactions that elevate arsenic concentrations (table 3). Examples of human-related factors that affect groundwater quality include irrigation of cropland and urban landscaped areas, which through multiple mechanisms, can elevate concentrations of dissolved solids, nitrate, and selected trace elements and result in pesticide and VOC detections (table 3).

Table 3. Summary of documented effects of natural and human-related factors on groundwater quality in the West Salt River Valley, Arizona.

[VOC, volatile organic compound]

Groundwater-quality effect	Cause	General location(s)	Reference(s)
Primarily natural factors			
Elevated concentrations of dissolved solids	Dissolution of evaporites in the lower part of the lower unit of the basin fill	Areas adjacent to and downgradient of the Luke Salt Body	Eaton and others (1972), Brown and Pool (1989)
Elevated concentrations of nitrate	Transport of nitrogen from decomposed vegetation or nitrogen fixed by bacteria associated with desert legumes	Basin wide	Gellenbeck (1994)
Elevated concentrations of arsenic	Geochemical reactions between the groundwater and compounds in the basin fill that are presumed to come from hydrothermal sulfide and arsenide deposits in the surrounding mountains	Basin wide	Robertson (1991)
Primarily human-related factors			
Elevated concentrations of nitrate and dissolved solids	Application of nitrogen fertilizers and evaporation during irrigation of crops and urban landscaped areas	Agricultural and urban areas, especially in the upper part of the aquifer above confining beds	Edmonds and Gellenbeck (2002), Brown and Pool (1989)
Elevated concentrations of nitrate	Transport of nitrogen from dairies and feedlots, wastewater treatment plants, and cultivated lands	Basin wide	Gellenbeck (1994)
Elevated concentrations of nitrate, dissolved solids, fluoride, arsenic, barium, chromium, and strontium	Evaporation of irrigation water before seeping to shallow groundwater	The shallow part of the aquifer in the western part of basin where wells in the agricultural area were sampled	Edmonds and Gellenbeck (2002)
Occurrence of pesticides	Application of pesticide compounds to croplands and urban landscaped areas	Agricultural and urban areas, especially in the upper part of the aquifer above confining beds	Edmonds and Gellenbeck (2002)
Occurrence of volatile organic compounds	Use of municipal wastewater containing VOCs for irrigation of crops and urban landscaped areas, and urban and industrial activities on the land surface and subsequent transport of compounds to aquifer	Agricultural and urban areas, especially in the upper part of the aquifer above confining beds	Edmonds and Gellenbeck (2002)

References Cited

Anderson, T.W., Freethey, G.W., and Tucci, P., 1992, Geohydrology and water resources of alluvial basins in south-central Arizona and parts of adjacent states: U.S. Geological Survey Professional Paper 1406–B, 67 p., 3 plates, scale 1:250,000.

Anning, D.W., and Duet, N.R., 1994, Summary of groundwater conditions in Arizona, 1987–90: U.S. Geological Survey Open File Report 94–476, 1 plate.

Brown, J.G., and Pool, D.R., 1989, Hydrogeology of the western part of the Salt River Valley, Maricopa County, Arizona: U.S. Geological Survey Water-Resources Investigations Report 88–4202, 5 sheets.

Central Arizona Project, 2006a, Central Arizona Project: Operations and water deliveries, monthly data calendar year 2005, accessed on September 18, 2006 at http://www.cap-az.com/operations/index.cfm?action=deliveries&subSection=15.

Central Arizona Project, 2006b, Recharge program operations: project deliveries, delivery data table, accessed on October 12, 2006 at http://www.cap-az.com/recharge/index.cfm?action=deliveries&subSection=70.

Cordy, G.E., Rees, J.A.H., Edmonds, R.J., Gebler, J.B., Wirt, Laurie, Gellenbeck, D.J., and Anning, D.W., 1998, Water-quality assessment of the Central Arizona Basins, Arizona and Northern Mexico—Environmental setting and overview of water quality: U.S. Geological Survey Water-Resources Investigations Report 98–4097, 71 p.

Corkhill, E.F., Corell, Steve, Hill, B.M., Carr, D.A., 1993, A regional groundwater flow model of the Salt River Valley—Phase I: Phoenix Active Management Area hydrogeologic framework and basic data report, Arizona Department of Water Resources Modeling Report No. 6, 120 p., 2 plates.

Corkhill, F., Bota, L., Jahnke, P., Mason, D., and Hipke, W., 2004, Revised ground-water modeling results for the East Valley Water Forum Regional Water Planning Study: Arizona Department of Water Resources, accessed on October 13, 2006 at http://www.water.az.gov/dwr/content/Publications/files/EVWF_revised_run1_12_2_04.pdf.

Damon, P.E., Lynch, D.J., and Shafiqullah, Mohammed, 1984, Cenozoic landscape development in the Basin and Range province of Arizona, in Smiley, T.L., Nations, J.D., Péwé, T.L., and Schafer, J.P., eds., Landscapes of Arizona: Lanham, Maryland, University Press of America, p. 175–206.

Davis, A.P., 1897, Irrigation near Phoenix, Arizona: U.S. Geological Survey Water-Supply Paper No. 2, 98 p.

Dickinson, W.R., 1989, Tectonic setting of Arizona through geologic time, in Jenny, J.P., and Reynolds, S.J., eds., Geologic Evolution of Arizona: Arizona Geological Society Digest No. 17, p. 1–16.

Eaton, G.P., Peterson, D.L., and Schumann, H.H., 1972, Geophysical, geohydrological, and geochemical reconnaissance of the Luke salt body, central Arizona: U.S. Geological Survey Professional Paper 753, 28 p.

Edmonds, R.J., and Gellenbeck, D.J., 2002, Ground-water quality in the West Salt River Valley, Arizona, 1996–98—Relations to hydrogeology, water use, and land use: U.S. Geological Survey Water Resources Investigations Report 01–4126, 60 p. Available at http://az.water.usgs.gov/pubs/WRIR01-4126intro.html.

Freethey, G.W., and Anderson, T.W., 1986, Predevelopment hydrologic conditions in the alluvial basins of Arizona and adjacent parts of California and New Mexico: U.S. Geological Survey Hydrologic Investigations Atlas 664, 3 plates.

Gellenbeck, D.J., 1994, Isotopic compositions and sources of nitrate in ground water from Western Salt River Valley, Arizona: U.S. Geological Survey Water-Resources Investigations Report 94–4063, 53 p.

Gellenbeck, D.J., and Anning, D.W., 2002, Occurrence and distribution of pesticides and volatile organic compounds in ground water and surface water in Central Arizona Basins, 1996–98, and their relation to land use: U.S. Geological Survey Water Resources Investigations Report 01–4144, 107 p. Available at http://az.water.usgs.gov/pubs/WRIR01-4144intro.html.

Hirschberg, D.M., and Pitts, S.G., 2000, Digital geologic map of Arizona: a digital database from the 1983 printing of the Wilson, Moore, and Cooper 1:500,000 scale map of Arizona: U.S. Geological Survey Open-File Report 2000–409. Available at http://geopubs.wr.usgs.gov/open-file/of00-409/.

Lee, W.T., 1905, The underground waters of Salt River Valley, Arizona: U.S. Geological Survey Water Supply and Irrigation Paper 136, 196 p. 2 plates.

Long, A.E., Brown, J.G., and Gellenbeck, D.J., 1997, Statistical analysis of nitrate in ground water, West Salt River Valley, Arizona: U.S. Geological Survey Water-Resources Investigations Report 97–4185, 38 p.

Menges, C.M., and Pearthree, P.A., 1989, Late Cenozoic tectonism in Arizona and its impact on regional landscape evolution: *in* Jenny, J.P., and Reynolds, S.J., eds., Geologic Evolution of Arizona, Arizona Geological Society Digest No. 17, p. 649–680.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the Southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

Owenby, J.R., and Ezell, D.S., 1992, Climatography of the United States—Monthly station normals of temperature, precipitation, and heating and cooling degree days, 1961–1990 for Arizona: Ashville, North Carolina, U.S. Department of Commerce, National Oceanic and Atmospheric Agency, National Climatic Data Center Report No. 81.

Reeter, R.W., and Remick, W.H., 1986, Maps showing groundwater conditions in the West Salt River, East Salt River, Lake Pleasant, Carefree, and Fountain Hills sub-basins of the Phoenix Active Management Area, Maricopa, Pinal, and Yavapai Counties, Arizona—1983: Arizona Department of Water Resources, Hydrologic Map Series Report Number 12, 3 plates, scale 1:125,000

Robertson, F.N., 1991, Geochemistry of ground water in alluvial basins of Arizona and adjacent parts of Nevada, New Mexico, and California: U.S. Geological Survey Professional Paper 1406–C, 90 p.

Salt River Project, 2006, Canal origins, Salt River Project, accessed on October 4, 2007 at http://www.srpnet.com/water/canals/origins.aspx.

Sargent, C.S. ed., 1988, Metro Arizona: Scottsdale, Arizona, Biffington Books, Annual Meeting of Association of American Geographers, April 6–10, 1988, Phoenix, Arizona, 195 p.

Schumann, H.H., 1995, Land subsidence and earth-fissure hazards near Luke Air Force Base, Arizona, *in* Prince, K.R., Galloway, D.L., and Leake, S.A., eds. U.S. Geological Survey Subsidence Interest Group Conference, Edwards Air Force Base, Antelope Valley, California, November 18–19, 1992—Abstracts and Summary: U.S. Geological Survey Open-File Report 94–532, p. 18–21.

Tadayon, S., Duet, N.R., Fisk, G.G., McCormack, H.F., Partin, C.K., Pope, G.L., and Rigas, P.D., 2001, Water resources data, Arizona, water year 2000: U.S. Geological Survey Water-Data Report AZ–00–1, 390 p.

Thompson, T.H., Nuter, Janet, and Anderson, T.W., 1984, Maps showing distribution of dissolved solids and dominant chemical type in ground water, Basin and Range province, Arizona: U.S. Geological Survey Water Resources Investigations Report 83–4114–C, 4 p., 4 sheets, scales 1:500,000.

Turney, O.A., 1929, Prehistoric irrigation canals, fifth ed., Phoenix, Arizona, 1 plate.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed March 3, 2009 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003b, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

Section 8.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the Upper Santa Cruz Basin, Arizona

By David W. Anning and James M. Leenhouts

Basin Overview

The Upper Santa Cruz Basin hosts a growing population in the Tucson metropolitan area and several other communities in south central Arizona (fig. 1). Groundwater development to support the population and their economic and cultural activities over the past century has caused substantial changes in the basin-fill aquifer, including water-level declines, a 51 percent increase in groundwater recharge, and a 171 percent increase in groundwater discharge. These and other changes to the aquifer have resulted in an increase in the intrinsic susceptibility of the aquifer to contamination, and the effects of both natural and human-related factors on groundwater quality are discussed in this section of the report.

As a Basin and Range feature, the 2,530-mi^2 Upper Santa Cruz Basin consists of an elongated sediment-filled valley bounded by a several mountain ranges, including the Pajarito, Atascosa, Tumacacori, Cerro Colorado, Sierrita, Tucson, and Tortolita mountains to the west and the Patagonia, San Cayetano, Santa Rita, Rincon, and Santa Catalina mountains to the east (fig. 1). Altitudes of the valley floor range from about 2,100 to 3,900 ft, and the tops of the surrounding mountains reach as high as 9,460 ft. The basin is topographically open and drained by the Santa Cruz River. The basin does not include all tributaries to the Santa Cruz River, but rather it receives surface-water inflow from the Cienega Creek drainage, Sonoita Creek drainage, and also the uppermost part of the Santa Cruz River drainage that is upstream from Nogales, Arizona, and extends into Mexico.

The Upper Santa Cruz Basin has an arid to semiarid climate, but wide variations in altitude cause large differences in precipitation and temperature. Precipitation is greater in the mountains than on the valley floor due to orographic effects, and the higher elevation mountains on the east side of the basin receive more precipitation than the lower elevation mountains on the west side of the basin. Mean annual precipitation from 1971 to 2000 for the entire basin, including the mountains, is about 17 in., whereas the valley floor on average receives about 15 in. (McKinney and Anning, 2009). Rainfall from July through September generally results from the North American Monsoon weather pattern (Adams and Comrie, 1997) and occurs as intense, local convective storms. Precipitation records for Tucson for 1949–06 indicate that about 45 percent of the annual rainfall falls during the months of July through September. Precipitation during the remainder of the year typically results from Pacific frontal storms and dissipating tropical cyclones.

Temperatures are highly variable spatially owing to the variations in elevation. Large diurnal temperature cycles are common and occur due to low atmospheric moisture and frequent cloudless days. For the period 1912–94, mean January temperature in Tucson was about 50°F while mean July temperature was 86°F (Western Regional Climate Center, 2007).

Frequent high temperatures combined with low relative humidity results in potential evaporation that is several times higher than the annual precipitation in many areas of the Upper Santa Cruz Basin. Potential evaporation calculated for a site near Nogales in 1995–96 was about 57 in. (Unland and others, 1998). As a result of the evaporation excess, groundwater recharge is generally thought not to occur by infiltration through the open desert floor, but instead is concentrated at the mountain fronts or at other locations where water collects at least temporarily, such as ephemeral-stream channels (Scanlon and others, 1999).

Land cover for the alluvial basin, excluding the surrounding mountainous areas, is estimated to be about 16 percent urban and about 1 percent agriculture (McKinney and Anning, 2009). Major crops include hay, cotton, grains, nuts, and vegetables. Most of the present-day urban land was previously desert rangeland, although some of the urbanized areas along the Santa Cruz and Rillito Rivers had previously been agricultural lands. The Tucson metropolitan area accounts for much of the urban land, and like many urban areas in the southwestern United States, the population there has undergone steady but rapid growth in the last few decades. The total population of the Upper Santa Cruz Basin in 2005 was estimated to be about 914,000 people (McKinney and Anning, 2009), most of whom are in the Tucson metropolitan area.

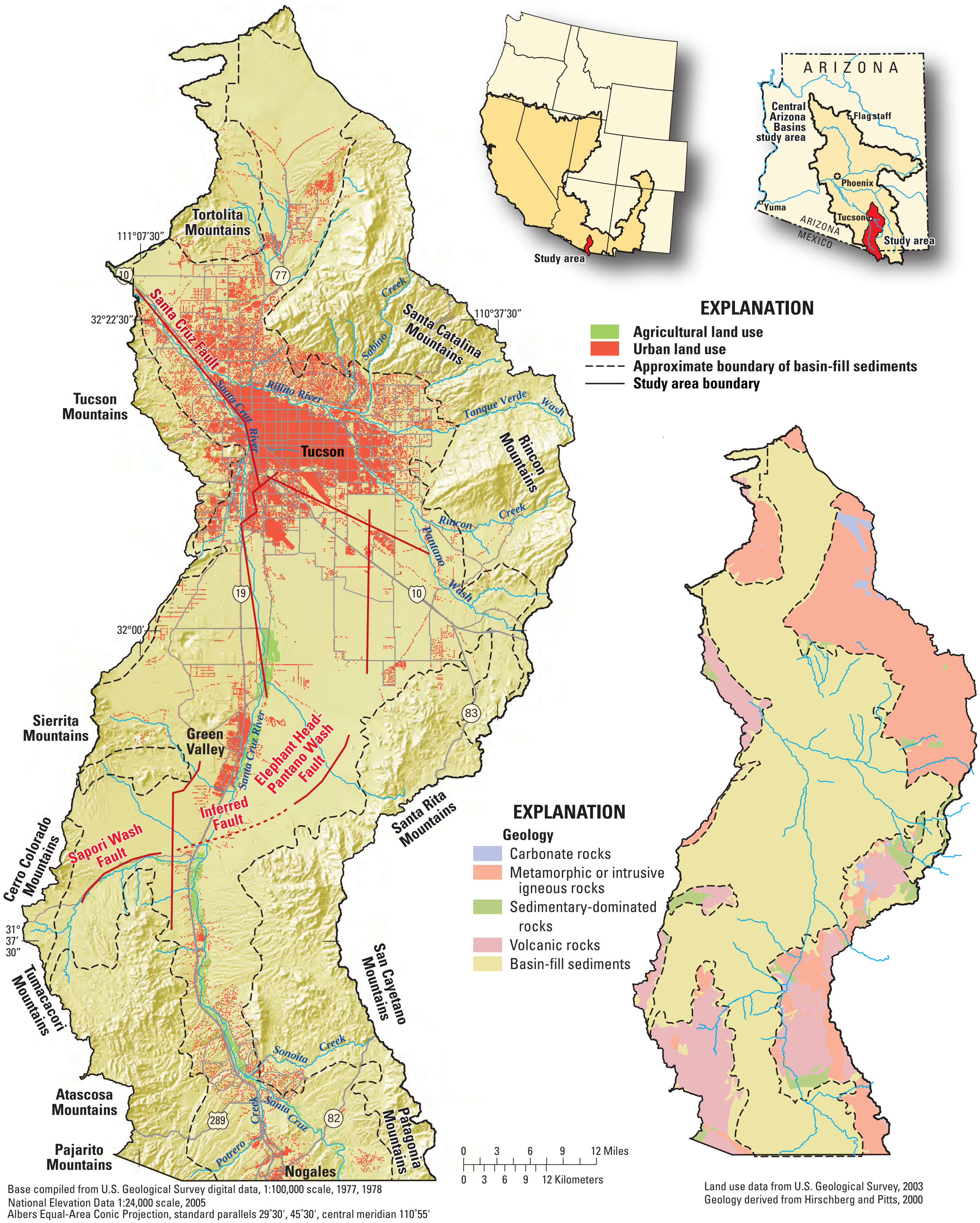


Figure 1. Physiography, land use, and generalized geology of the Upper Santa Cruz Basin, Arizona.

Water Development History

The Upper Santa Cruz Basin has a rich water development history from the Paleoindian period through modern times, and the resulting water supply infrastructure includes several well networks for the City of Tucson and other municipalities in the basin, as well as aqueducts for delivery of Colorado River water through the Central Arizona Project. Archeological evidence indicates that at least transient human occupation of the Upper Santa Cruz Basin began as early as the Paleoindian period between 11,500 and 7,500 BC (Thiel and Diehl, 2006). The groups existing at that time were largely hunter-gathers who did not likely take up permanent residence. The original nomadic residents of the Upper Santa Cruz Basin likely utilized available water supplies only for drinking and cleaning purposes.

Evidence shows that by 400 BC, groups were living in agricultural settlements in the floodplain of the Santa Cruz River (Thiel and Diehl, 2006). The early farming peoples utilized water from then-perennial sections of the Santa Cruz River and from springs to irrigate crops. Native residents continued to exclusively farm and hunt in the area until the arrival of Father Eusebio Franscisco Kino in 1694 (Thiel and Diehl, 2006). The first historical accounts of Tucson, written by Father Kino during the 1690s, suggest that virtually the entire flow of the Santa Cruz River was diverted into irrigation canals (Klimas and others, 2006). In the following 150 years, the Upper Santa Cruz Basin saw a continued increase in numbers of Spanish and Mexican explorers and settlers who brought European agriculture and technology to the area. In 1856, the U.S. Army opened its first outpost in Tucson, thus encouraging further settlement of the area by residents of European descent (Thiel and Diehl, 2006). The residents and temporary occupants of the area relied on surface-water diversions and springs until the development of pump technology allowed significant exploitation of groundwater resources.

Among the communities in the Upper Santa Cruz Basin, Tucson became the population center of the area, and by 1900, Tucson's population had grown to about 7,500. In 1914, a City of Tucson bond issue financed a new storage reservoir and the installation of 6 new wells that utilized new technology that could extract 1 million gallons of water per day per well from the underlying aquifer (Gelt and others, 2006). Groundwater withdrawals in the Upper Santa Cruz Basin were about 7,000 acre-ft in 1915 (Anning and Duet, 1994) and were relatively uniform from 1920 to 1939, at an average of about 34,000 acre-ft/yr (Hanson and Benedict, 1994). As population increased, withdrawals for urban and agricultural uses increased (fig. 2), and groundwater levels declined in response. By the 1940s, groundwater levels had declined sufficiently that surface flow in the perennial reaches of the Santa Cruz River in the Tucson area was captured and the channel became ephemeral (Hanson and Benedict, 1994).

The period after World War II and continuing to the current time was characterized by rapid growth, and the City of Tucson responded by drilling many additional wells. In the 1960s, Tucson determined that the then-established well fields were inadequate to meet growing demands for water and began purchasing land and drilling wells in Avra Valley, which is adjacent and to the west of the Upper Santa Cruz Basin. Groundwater withdrawals increased to a peak of about 281,000 acre-ft in 1976 and generally declined after that to about 210,000 acre-ft in 1990, mostly as a result of decreased agricultural water demand (fig. 2; Anning and Duet, 1994). For the period 1940–86, about 52 percent of withdrawals were for agricultural use, 33 percent for public-supply use, and 15 percent for industrial use (Hanson and Benedict, 1994). In the late 1990s, estimated withdrawals were about 221,000 acre-ft/yr, and by use categories were about 19 percent for agriculture, 55 percent for public supply, and 26 percent for industry (based on data from Arizona Department of Water Resources, 1999b; and Mason and Bota, 2006).

Municipal wastewater effluent has long been treated as a source of water in the Upper Santa Cruz Basin. From about 1900 to 1950, effluent from Tucson was used to irrigate crops (Gelt and others, 2006), and release of treated effluent to the Santa Cruz River and subsequent recharge began in 1951 (Hanson and Benedict, 1994). Starting in 1983, a tertiary treatment plant was constructed to deliver the treated effluent to golf courses and public turf areas for irrigation. Currently, treated effluent is delivered to more than 200 water users including 13 golf courses, 25 parks, and 30 schools. In 1998, about 13,000 acre-ft of treated effluent was delivered for reuse in the Tucson area, and about 54,000 acre-ft was recharged as infiltration through the Santa Cruz River, some of which occurred downstream from the Upper Santa Cruz Basin (Gelt and others, 2006). In the same year, about 14,600 acre-ft of effluent was released to the Santa Cruz River from the Nogales International Wastewater Treatment Plant, which treats sewage from both the United States and Mexico (Nelson and Erwin, 2001).

An additional source of water had been considered for Arizona even before statehood was achieved in 1912: the Colorado River. Arizona's politicians were unified behind the concept by 1960, and in 1968 President Lyndon B. Johnson signed a bill approving the construction of the Central Arizona Project (CAP); construction of the project was started in 1973 (Central Arizona Project, 2007). In the 1970s, however, President Jimmy Carter expressed doubt that project building would solve western water problems and demanded changes in Arizona water laws to promote conservation (Gelt and others, 2006). The response by the Arizona Legislature was the creation of the Groundwater Management Act of 1980. This act established specific Active Management Areas that are subject to a set of specific requirements dealing with water use and development. The Upper Santa Cruz Basin eventually included two Active Management Areas within its bounds.

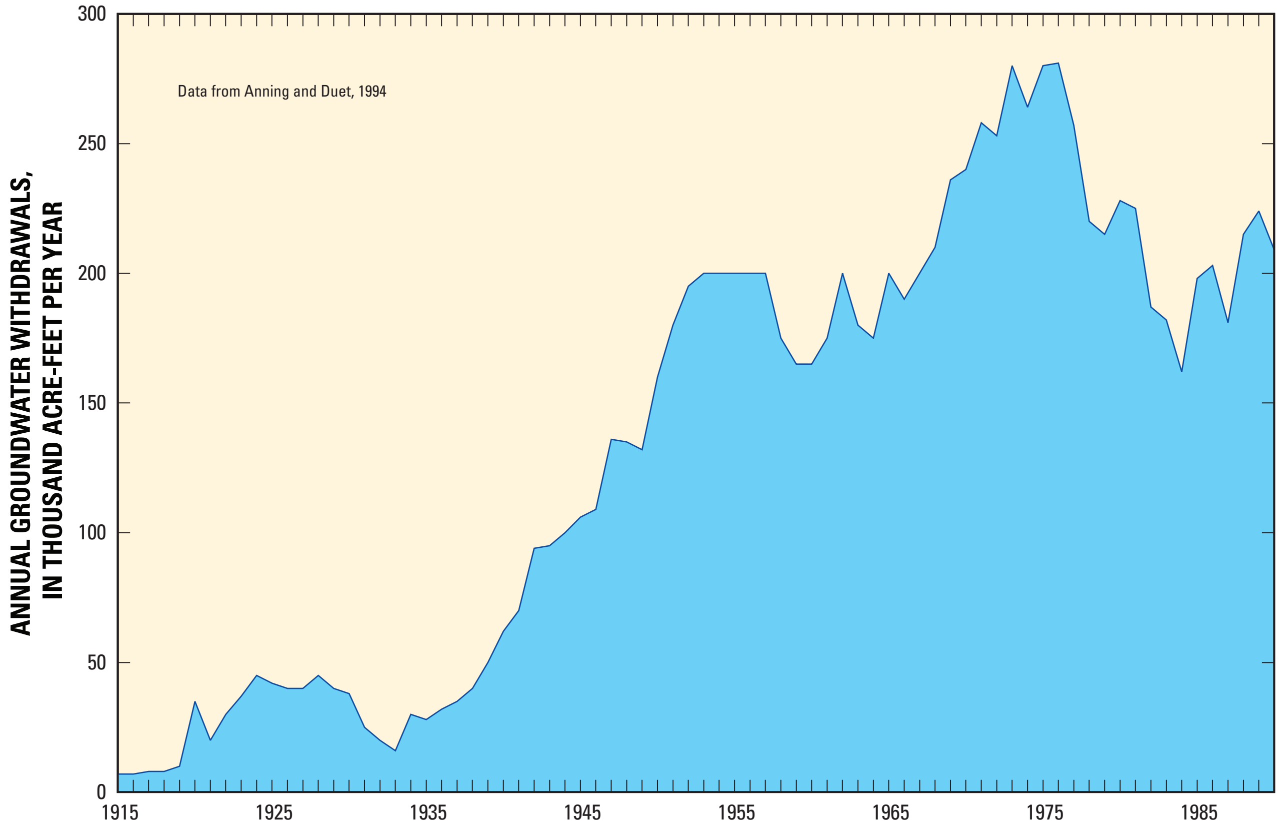


Figure 2. Annual groundwater withdrawals in the Upper Santa Cruz Basin, Arizona, 1915–1990.

The CAP infrastructure to Tucson was completed by 1990. Originally the CAP water was destined for use in agriculture and mining activities, two of the dominant water uses in the state. Eventually, however, few mines or farms utilized the new water owing to concerns about its quality and cost. The City of Tucson began delivering CAP water to customers in 1992, but differences in quality relative to the native groundwater led to problems such as delivery of water with pipe corrosion to consumers' homes (Gelt and others, 2006). As a result, in 1996, the City of Tucson began recharging its CAP allotment in Avra Valley and then pumped the mixture of recharged CAP water and native groundwater for delivery within the Upper Santa Cruz Basin. The delivery of blended CAP water to areas of metropolitan Tucson, which in 2005 was in excess of 150,000 acre-ft, allowed the reduction or cessation of pumping from some wells and resulted in recovery of water levels in certain areas. The chemical characteristics of CAP water differs from that of native groundwater, most notably in dissolved-solids content, so its delivery to customers and eventual appearance as wastewater have implications for groundwater quality in the Upper Santa Cruz Basin.

Hydrogeology

The Upper Santa Cruz Basin is characteristic of the Basin and Range Physiographic Province (Fenneman, 1931) and consists of block-faulted mountains separated by a north-south trending sediment-filled valley. The mountains block virtually all subsurface flow between adjacent valleys and thus serve as hydrologic boundaries both for the groundwater and surface-water systems. The rocks of the surrounding mountains range in age from Precambrian to Tertiary; they consist primarily of granite, andesite, rhyolite, basalt, monzonite, granodiorite, gneiss, and secondarily of limestone, quartzite, conglomerate, sandstone, and shale. The crystalline and metamorphic rocks of the mountains are capable of storing and transmitting small amounts of water through connected fracture systems; the sedimentary rock units of the mountains are generally of low porosity and permeability but locally can have significant capacity to store and transmit water (Davidson, 1973; fig. 1).

Most of the capacity to store and transmit groundwater in the Upper Santa Cruz Basin resides in the sequence of Tertiary to Quaternary alluvial sediments that fill the basin, and where saturated, these sediments form the basin-fill aquifer. The thickness of these sediments varies from a thin veneer along the basin margins where bedrock emerges to more than 11,200 ft in the center of the basin (Hanson and Benedict, 1994). The sediments are of significantly different character and thickness (figs. 3*A*, 3*B*) to either side of a northeast-southwest trending fault that is inferred to connect the Sopori Wash and Elephant Head-Pantano Wash faults (Halpenny and Halpenny, 1988; fig. 1). The sediments north of the inferred fault are thicker than those to the south, and consist of the Pantano Formation, Tinaja beds, and Fort Lowell Formations (Davidson, 1973; fig. 3*A*). A veneer of alluvial fan, sheetflow, and stream alluvial deposits comprise the surface sediments. South of the inferred fault, the sediments that form the basin-fill aquifer consist of the Nogales Formation, thought to be correlative with the lower Tinaja beds (Anderson, 1987), and groupings of older and younger alluvium (fig. 3*B*; Halpenny and Halpenny, 1988).

The Pantano Formation is a consolidated to semiconsolidated sedimentary unit of Tertiary age that overlies bedrock north of the inferred fault. It ranges in thickness from hundreds to thousands of feet and consists of reddish-brown silty sandstone to gravel that is weakly to strongly cemented (Davidson, 1973). Wells in shallow portions of the Pantano Formation tend to yield only small volumes of water, but the unit yields greater volumes at depth. The Pantano Formation crops out along the southern slopes of the Santa Catalina Mountains, the western slopes of the Rincon Mountains, and the northeastern slopes of the Sierrita Mountains. The hydraulic conductivity values of the formation range from about 1 to 10 ft/d (Davidson, 1973).

Unconformably overlying the Pantano Formation are three divisions (lower, middle, and upper) of the Tertiary age Tinaja beds. The Tinaja beds range in thickness from 0 to more than 2,000 ft and crop out only along the margins of the basin where exposed by erosion or where they were never covered by younger sediments. The Tinaja beds consist largely of sandy gravel near the basin periphery, but transitions to gypsiferous clayey silt and mudstone toward the center of the basin. The Tinaja beds constitute a significant portion of the aquifer in the Upper Santa Cruz Basin, with hydraulic conductivity values ranging from about 1 to 50 ft/d (Davidson, 1973).

The Fort Lowell Formation of Quaternary age is a locally derived sedimentary unit that unconformably overlies the Tinaja beds and underlies most of the surface of the Upper Santa Cruz Basin. Its thickness ranges from 0 near the edge of the basin up to about 400 ft in the center of the basin (Davidson, 1973). The Fort Lowell Formation grades from silty gravel near the basin margins to silty sand and clayey silt near the basin center (Coes and others, 2000) and is the most productive part of the aquifer in the Tucson area. The Fort Lowell formation is typically loosely packed to weakly cemented, and hydraulic conductivity values range from about 20 to as much as 100 ft/d (Davidson, 1973).

A veneer of about 5 to 100 ft of alluvium unconformably overlies older sediments in most of the Upper Santa Cruz Basin. These sediments consist of alluvial fan and sheetflow deposits over the basin floor and stream-channel deposits along the Santa Cruz River and many tributary channels (Davidson, 1973). These surficial sediments are unsaturated in most of the Upper Santa Cruz Basin, except along stream channels, where in places they yield useable quantities of groundwater.

South of the inferred fault, the basin fill consists of the Nogales Formation, older alluvium, and younger alluvium. The Nogales Formation is a consolidated conglomerate of Tertiary age that overlies the bedrock in the basin (fig. 3*B*; Halpenny, 1963). The Nogales Formation consists of sandstone, claystone, and conglomerate derived from limestone, granite, and volcanic material and is up to 2,400 ft thick (Halpenny, 1963, Gettings and Houser, 1997). Hydraulic conductivity values range from about 0.3 to 3.0 ft/d (Nelson, 2007).

Older alluvium consists of deposits of weakly cemented gravel, sand, and silt and overlies the Nogales Formation (Halpenny and Halpenny, 1988). The older alluvium is of Tertiary and Quaternary age and forms terraces that mark the old, inner valley of the Santa Cruz River south of the inferred fault. The terraces disappear along the edges of the inner valley north of the inferred fault (Halpenny and Halpenny, 1988). The older alluvium is up to 900 ft thick (Gettings and Houser, 1997), and hydraulic conductivity values range from about 1 to 50 ft/d (Nelson, 2007).

Younger alluvium is of Quaternary age and has been deposited along the Santa Cruz River. Younger alluvium is composed of gravel, sand, and occasional lenses of silt and ranges in thickness from about 30 to 150 ft thick (Halpenny and Halpenny, 1988; Carruth, 1995). The younger alluvium readily transmits water and has hydraulic conductivity values of 100 to 600 ft/d (Nelson, 2007).

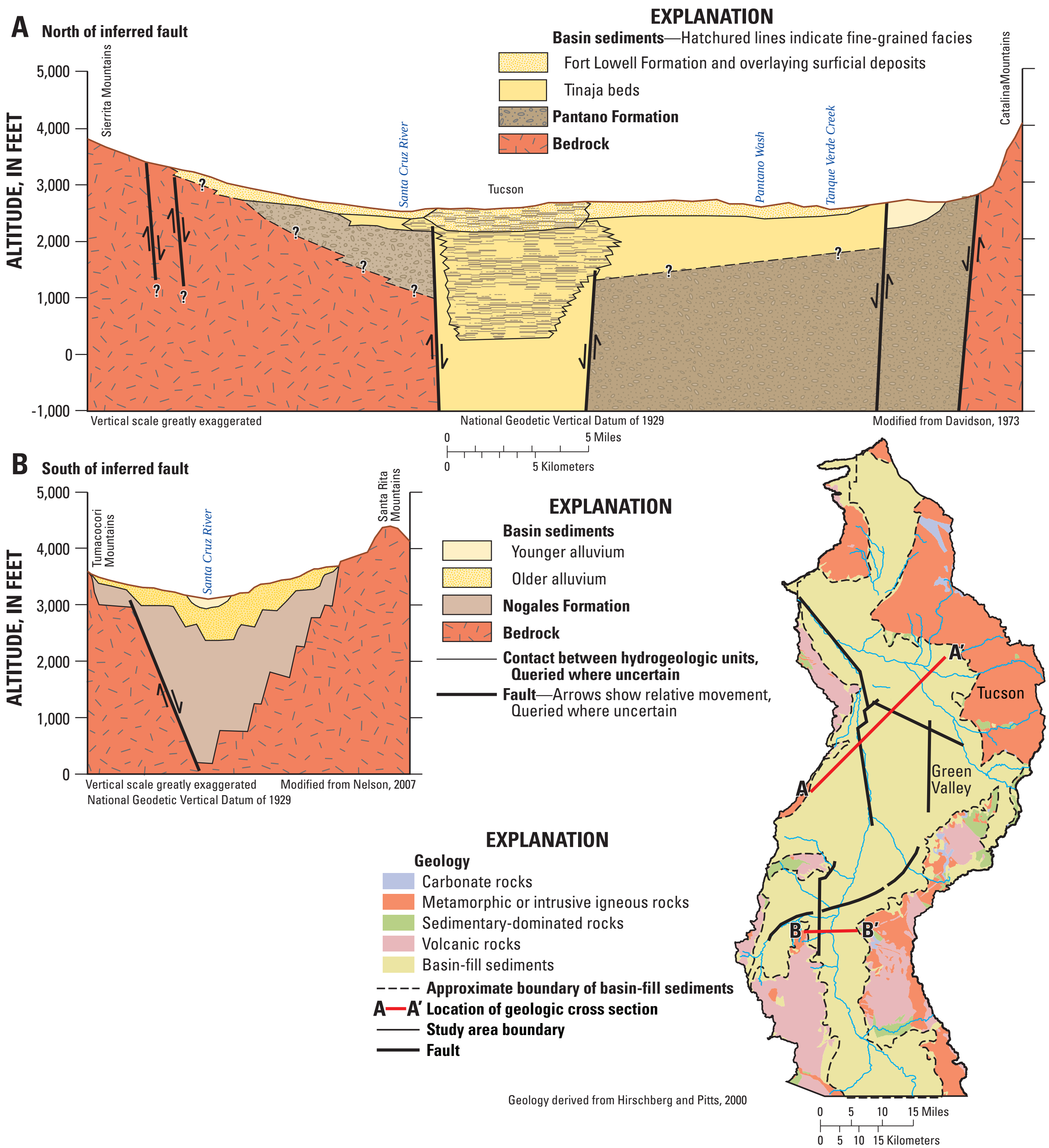


Figure 3. Generalized hydrogeologic sections of the Upper Santa Cruz Basin, Arizona. (*A*) North of the inferred fault, and (*B*) South of the inferred fault.

Conceptual Understanding of the Groundwater Flow System

The predevelopment groundwater flow system of the Upper Santa Cruz Basin resembles that of other Basin and Range systems in southern Arizona. Water levels in the basin-fill aquifer are generally parallel to the land surface of the valley floor and consequently groundwater flows from the basin margins towards the center and then northward along the basin axis. The aquifer is replenished primarily through mountain-front and mountain-block recharge, water losses from the channel of the Santa Cruz River, and as a consequence of water-resources development and use, incidental recharge from human activities. Groundwater leaves the aquifer primarily through evapotranspiration, and with water-resources development, through groundwater pumpage. Details of groundwater recharge, discharge, and flow are described in the following sections, along with the effects that water-resources development has had on the basin-fill aquifer.

Water Budget

A conceptual understanding of the primary and significant fluxes of water through the basin-fill aquifer is summarized in a groundwater budget for predevelopment and modern conditions in the Upper Santa Cruz Basin (fig. 4; table 1). Most water-budget components estimated in this study were derived by combining data reported for the area south of the inferred fault (Nelson, 2007) and the area north of the inferred fault (Mason and Bota, 2006). The water budget and groundwater flow system have changed significantly from the predevelopment period (prior to about 1900) to modern times (circa 2000; fig. 4; table 1).

The relative abundance of precipitation at higher elevations combined with the low permeability of mountain bedrock and the general permeable nature of basin-margin sediments leads to a conceptual model wherein a significant portion of groundwater recharges at the mountain fronts. The estimated mountain-front and mountain-block recharge for the Upper Santa Cruz Basin is 36,000 acre-ft/yr (table 1; Nelson, 2007; Mason and Bota, 2006), most of which originates within the Santa Catalina, Rincon, Santa Rita, and Sierrita Mountains. Streamflow infiltration is also an important recharge mechanism, although it may have varied in magnitude significantly through time. Streamflow was intermittent in the Santa Cruz River over most of its length prior to 1870 (Betancourt and Turner, 1993). In the years from the 1870s through 1890s, the main stem river channel became entrenched. That down cutting led to an increase in groundwater discharge to the streambed while the topmost part of the aquifer near the stream drained and established a new steady-state equilibrium condition (Hanson and Benedict, 1994). The estimated predevelopment streamflow infiltration for the Upper Santa Cruz Basin is 52,000 acre-ft/yr (table 1; Nelson, 2007; Mason and Bota, 2006). The contribution of streamflow infiltration for the area north of the inferred fault of 34,000 acre-ft/yr, however, may be overestimated because the water table was higher along the rivers during predevelopment conditions than during 1940, the year for which Mason and Bota (2006) reported the number. Areally distributed recharge is generally thought to be small or nonexistent in many desert environments (Scanlon and others, 1999) and was not estimated in previous studies of the Upper Santa Cruz Basin. Predevelopment subsurface inflow to the basin from Mexico was estimated to be about 5,000 acre-ft/yr (Nelson, 2007). The total predevelopment recharge for the Upper Santa Cruz Basin is estimated to be 93,000 acre-ft/yr (fig. 4; table 1).

The predominant mode of predevelopment groundwater discharge was evapotranspiration. For the area south of the inferred fault, Nelson (2007) estimated that evapotranspiration of shallow groundwater was about 15,000 acre-ft/yr. Estimates of evapotranspiration for the reach north of the inferred fault under steady state conditions are available for about 1940 (Mason and Bota, 2006). These values, however, are significantly lower than the evapotranspiration amount for predevelopment conditions because pumpage through 1940 had already lowered groundwater levels below the root zone of the predevelopment riparian ecosystem (Hanson and Benedict, 1994). For this study, predevelopment evapotranspiration of 64,000 acre-ft/yr for the area north of the inferred fault was estimated as the sum of evapotranspiration and pumpage under steady-state conditions in 1940 as reported by Mason and Bota (2006). Total predevelopment evapotranspiration is estimated to be 79,000 acre-ft/yr (table 1). The other primary process of groundwater discharge is underflow out of the basin, which is estimated to be about 14,000 acre-ft/yr (Mason and Bota, 2006). Total groundwater discharge under predevelopment conditions is estimated to be 93,000 acre-ft/yr (table 1), which is equal to groundwater recharge under the steady-state assumption.

The development of groundwater resources in the Upper Santa Cruz Basin from predevelopment (circa 1900) to modern (circa 2000) times has significantly changed several components of the water budget. The quantity of water pumped from the aquifer became significant after about 1920. From 1920–40, pumping was relatively steady, averaging about 34,000 acre-ft/yr, and by 1940 the aquifer system was probably in a new state of equilibrium with stable water levels but at lower altitudes relative to predevelopment times (Hanson and Benedict, 1994). Pumping in 1940 was about 50,000 acre-ft. As noted earlier, groundwater pumpage for the late 1990s was about 221,000 acre-ft/yr (table 1).

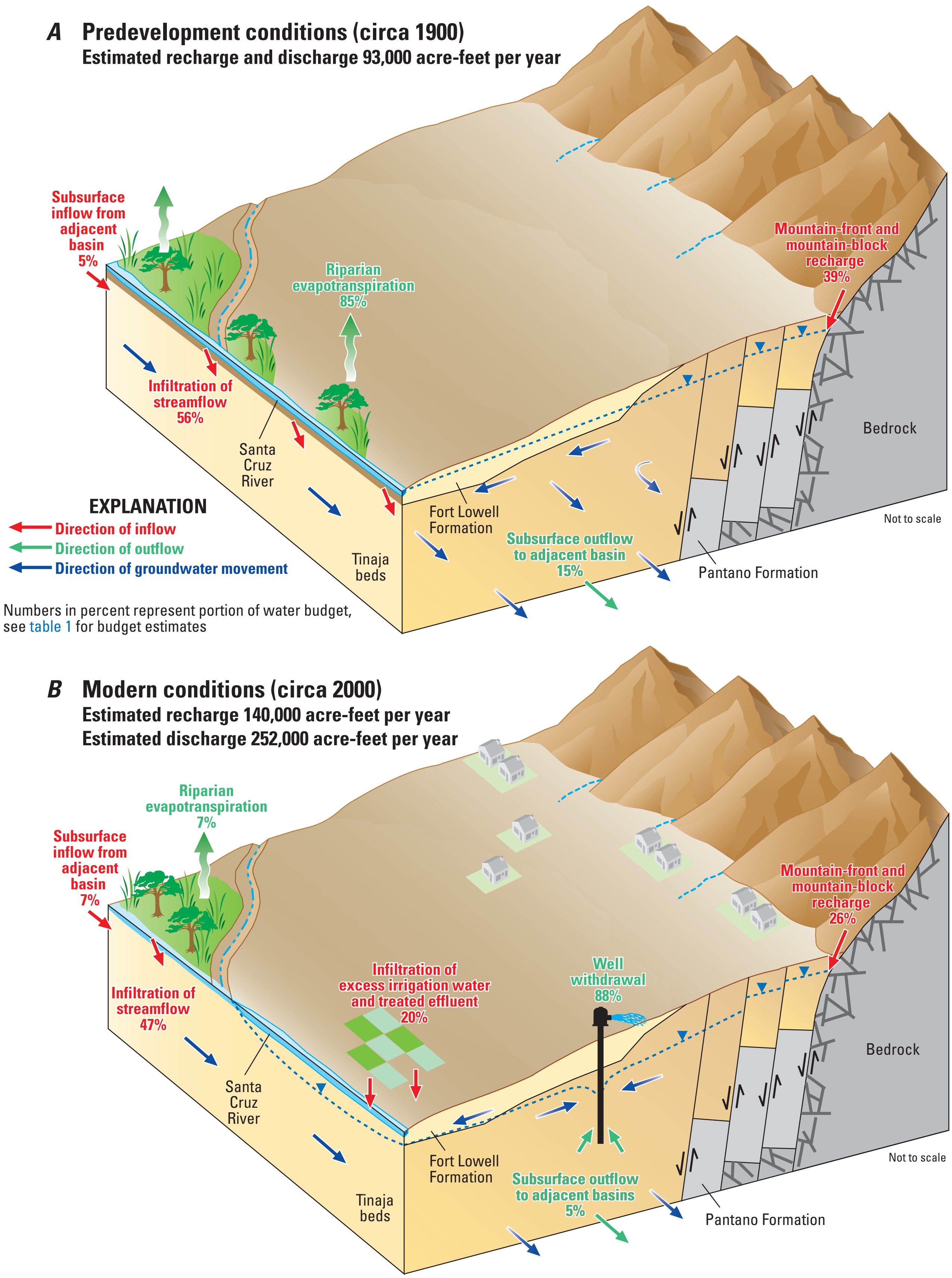


Figure 4. Generalized diagrams for the Upper Santa Cruz Basin, Arizona, showing components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin fill aquifer in the Upper Santa Cruz Basin, Arizona, under predevelopment and modern conditions.

[All values are in acre-feet per year (acre-ft/yr) and are rounded to the nearest thousand. Estimates of groundwater recharge and discharge for the area south of the inferred fault were derived from Nelson (2007). Estimates for the area north of the inferred fault were derived from Mason and Bota (2006), unless noted here. For the area north of the inferred fault, predevelopment evapotranspiration was computed as reported evapotranspiration plus pumpage for 1940 conditions. Infiltration of streamflow and evapotranspiration in the area north of the inferred fault may have been smaller than values reported here, which represent 1940 conditions, as a result of the groundwater table being lower along the river in 1940 than for predevelopment conditions. Modern infiltration of streamflow in the area north of the inferred fault is computed as predevelopment infiltration of streamflow plus 12,400 acre-ft/yr, a gain reported by Hanson and Benedict (1994) due to lowered groundwater levels near major streams. Estimates of recharge from excess irrigation water, sewage effluent, and industrial wastewater for the area north of the inferred fault are from Hanson and Benedict (1994). The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3. n/a, not applicable]

Budget component	Predevelopment conditions, circa 1900			Modern conditions, circa 2000			Change from predevelopment to modern conditions
	South of inferred fault	North of inferred fault	Total	South of inferred fault	North of inferred fault	Total	
Estimated recharge							
Subsurface inflow from adjacent basin	5,000	N/A	5,000	10,000	N/A	10,000	5,000
Mountain-block and mountain-front recharge	5,000	31,000	36,000	5,000	31,000	36,000	0
Infiltration of precipitation on basin	0	0	0	0	0	0	0
Infiltration of streamflow	18,000	34,000	52,000	20,000	46,000	66,000	14,000
Infiltration of excess irrigation water, sewage effluent, and industrial wastewater	0	0	0	2,000	26,000	28,000	28,000
Artificial recharge	0	0	0	0	0	0	0
Total recharge	28,000	65,000	93,000	37,000	103,000	140,000	47,000
Estimated discharge							
Subsurface outflow to adjacent basin	[1]0	14,000	14,000	[1]0	14,000	14,000	0
Evapotranspiration	15,000	64,000	79,000	15,000	2,000	17,000	-62,000
Discharge to streams	0	0	0	0	0	0	0
Discharge to springs and drains	0	0	0	0	0	0	0
Well withdrawals	0		0	16,000	205,000	221,000	221,000
Total discharge	15,000	78,000	93,000	31,000	221,000	252,000	159,000
Estimated change in storage (recharge-discharge)	N/A	N/A	0	N/A	N/A	-112,000	-112,000

[1]Flow occurs, but goes into area north of inferred fault or comes from area south of inferred fault.

Groundwater pumping has resulted in increases in recharge to and discharge from the aquifer. Pumping causes declines in water levels, which can lead to changes in the direction of flow or to the “capture” of water that under natural, predevelopment conditions was moving toward discharge areas. If the water-level declines are great enough, former discharge areas can become recharge areas. For the Upper Santa Cruz Basin, the lowering groundwater levels north of the inferred fault decreased evapotranspiration by 62,000 acre-ft/yr (table 1) and increased streamflow infiltration by 14,000 acre-ft/yr (table 1; Hanson and Benedict, 1994). At the international border, subsurface inflow into the Upper Santa Cruz Basin increased from predevelopment to modern conditions because pumping increased the hydraulic gradient of the aquifer in that area (Nelson, 2007). For this study, subsurface outflow data were not available for modern conditions, and the value for predevelopment conditions was used (table 1). The true modern subsurface outflow could be smaller than that shown (table 1) due to a decrease in hydraulic gradient or in the saturated cross section through which the water is moving.

As a consequence of the development and use of groundwater, recharge also occurred from incidental sources. North of the inferred fault, incidental recharge from excess irrigation water, effluent infiltration along the channel of the Santa Cruz River, and seepage from mine tailings ponds became major water sources replenishing the aquifer (Mason and Bota, 2006) and have significant implications with respect to water quality. The estimate of 26,000 acre-ft/yr for incidental recharge from these sources by Hanson and Benedict (1994) were used by Mason and Bota (2006) and in this study (table 1).

The cumulative effects of development have caused substantial changes in the groundwater flow system in the Upper Santa Cruz Basin (fig. 4; table 1). Comparison between predevelopment and modern conditions indicates considerable increases in water flowing in and out of the aquifer. Total inflows increased about 51 percent, from 93,000 to 140,000 acre-ft/yr, and outflows increased about 171 percent, from 93,000 to 252,000 acre-ft/yr (table 1). These increases in groundwater flux have implications for groundwater quality. As more water moves into the aquifer, especially if the water has been exposed to contaminant sources, the greater will be the intrinsic susceptibility of the aquifer to contamination. Reversing groundwater gradients so that former discharge areas become recharge areas, which has happened along the channel of the Santa Cruz River, creates new pathways to the aquifer for contaminant sources.

Groundwater Movement

Movement of groundwater in the aquifer of Upper Santa Cruz Basin is controlled by the locations and amounts of recharge and discharge and by the aquifer properties. Under predevelopment conditions, groundwater moved from upgradient mountain-front areas toward the river and then down valley to the north (fig. 5*A*). Changes to the aquifer system due to development can be characterized on a gross scale on the basis of water budget components, and they also can be characterized by changes in groundwater conditions and aquifer characteristics at a more local scale. In the Upper Santa Cruz Basin, such effects of development took the form of steeper vertical hydraulic gradients, thicker unsaturated zones, redirection of groundwater movement toward pumping centers, creation of perched water zones, reductions in aquifer transmissivity, land subsidence, and the capture of perennial streamflow and former riparian evapotranspiration along the channels of the Santa Cruz and Rillito Rivers (Coes and others, 2000).

North of the inferred fault, pumping for municipal, agricultural, industrial, and mining uses has lowered groundwater levels under the central area of Tucson, and also near Green Valley (fig. 5*C*; Arizona Department of Water Resources 1999a) and consequently the direction of groundwater movement has changed from that under predevelopment conditions (figs. 5*A* and 5*B*). By 1995, water-level declines caused by pumping were as great as 200 ft in the Tucson area, and as great as 150 ft in the Green Valley area (fig. 5*C*; Arizona Department of Water Resources 1999a; Anderson and others, 1992). The large water-level declines in the Tucson area have led to measured compaction of the aquifer (Hanson, 1989; Tucson Water, 1993) and predictions of subsidence potentially greater than 10 ft (Anderson, 1988). In less populated areas of the Upper Santa Cruz Basin, the declines have generally been limited to less than about 50 ft (fig. 5*C*; Arizona Department of Water Resources 1999a and 1999b; Anderson and others, 1992).

South of the inferred fault, depth to water has not changed significantly from predevelopment to modern conditions, and is generally less than 100 ft along the Santa Cruz River. North of the inferred fault depths to water are generally between 100 and 500 ft (fig. 5*D*; Arizona Department of Water Resources 1999a). While pumping can create downward hydraulic gradients and promote contaminant transport deep into the aquifer, the resulting water-level declines also dewater the upper part of the aquifer, thereby creating a longer travel path from land-surface sources to the water table and a greater opportunity for contaminant attenuation.

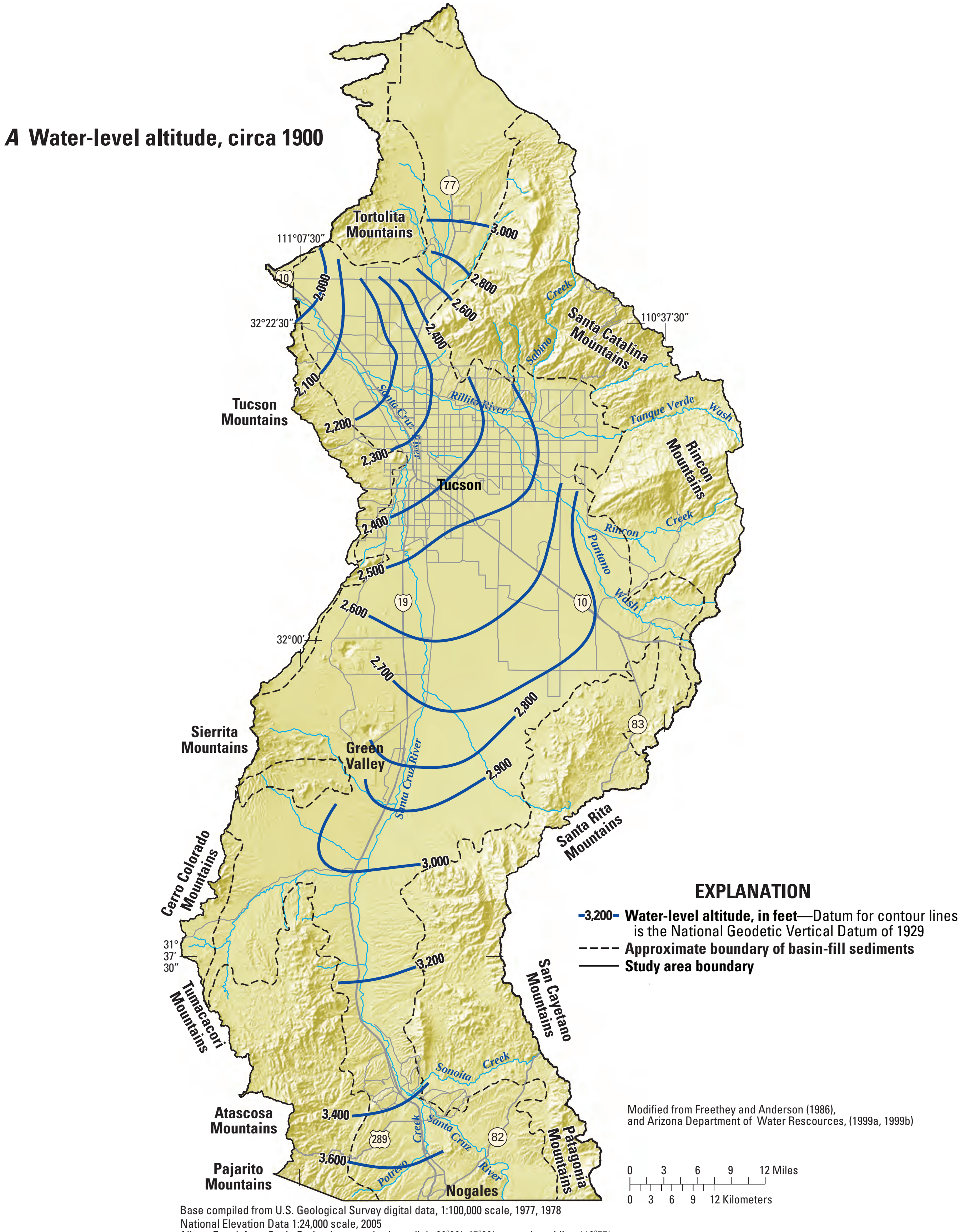


Figure 5. Water levels in the basin-fill aquifer of the Upper Santa Cruz Basin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1995. (*C*) Water-level change for 1900–1995. (*D*) Depth to water, 1995.

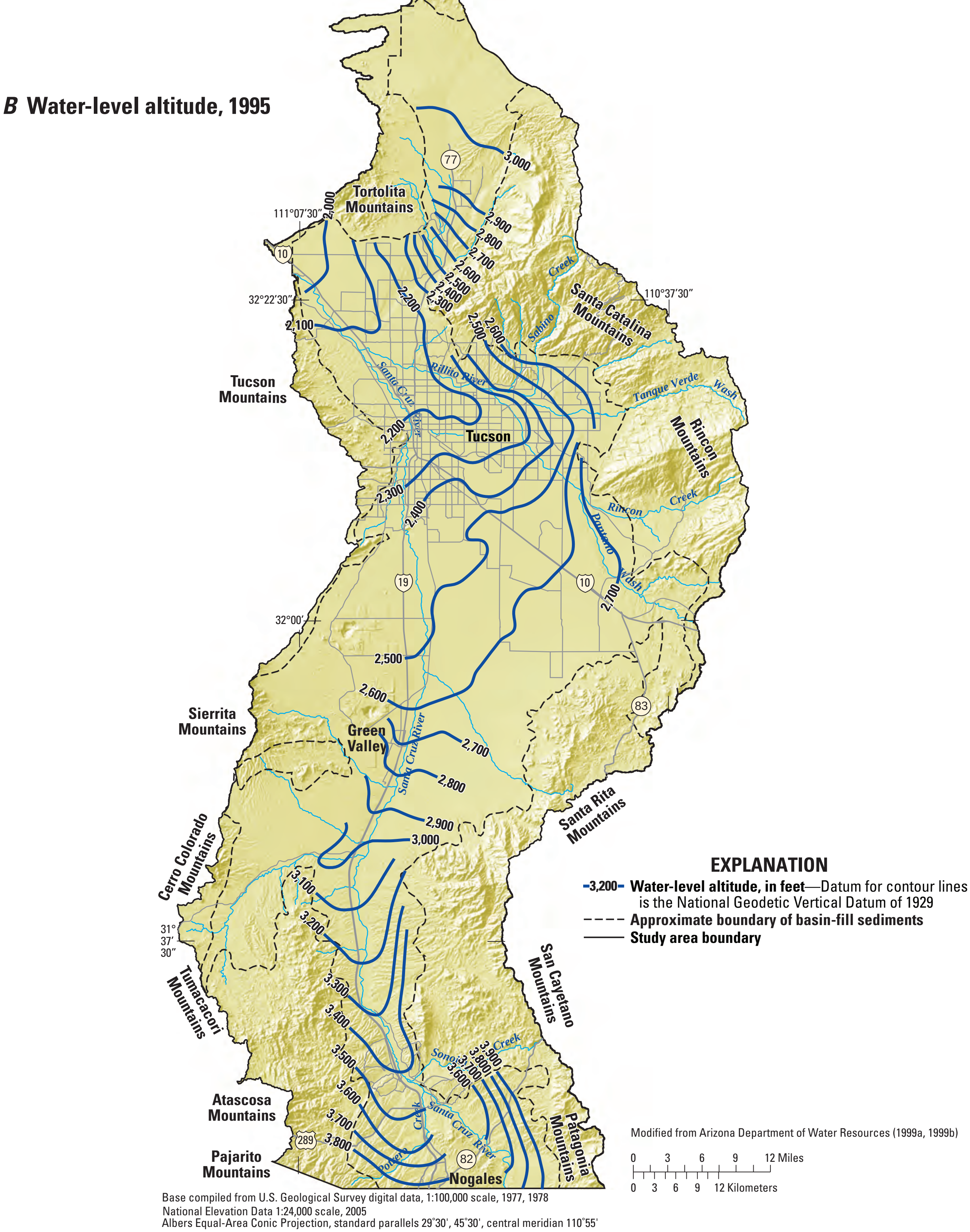


Figure 5. Water levels in the basin-fill aquifer of the Upper Santa Cruz Basin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1995. (*C*) Water-level change for 1900–1995. (*D*) Depth to water, 1995—Continued.

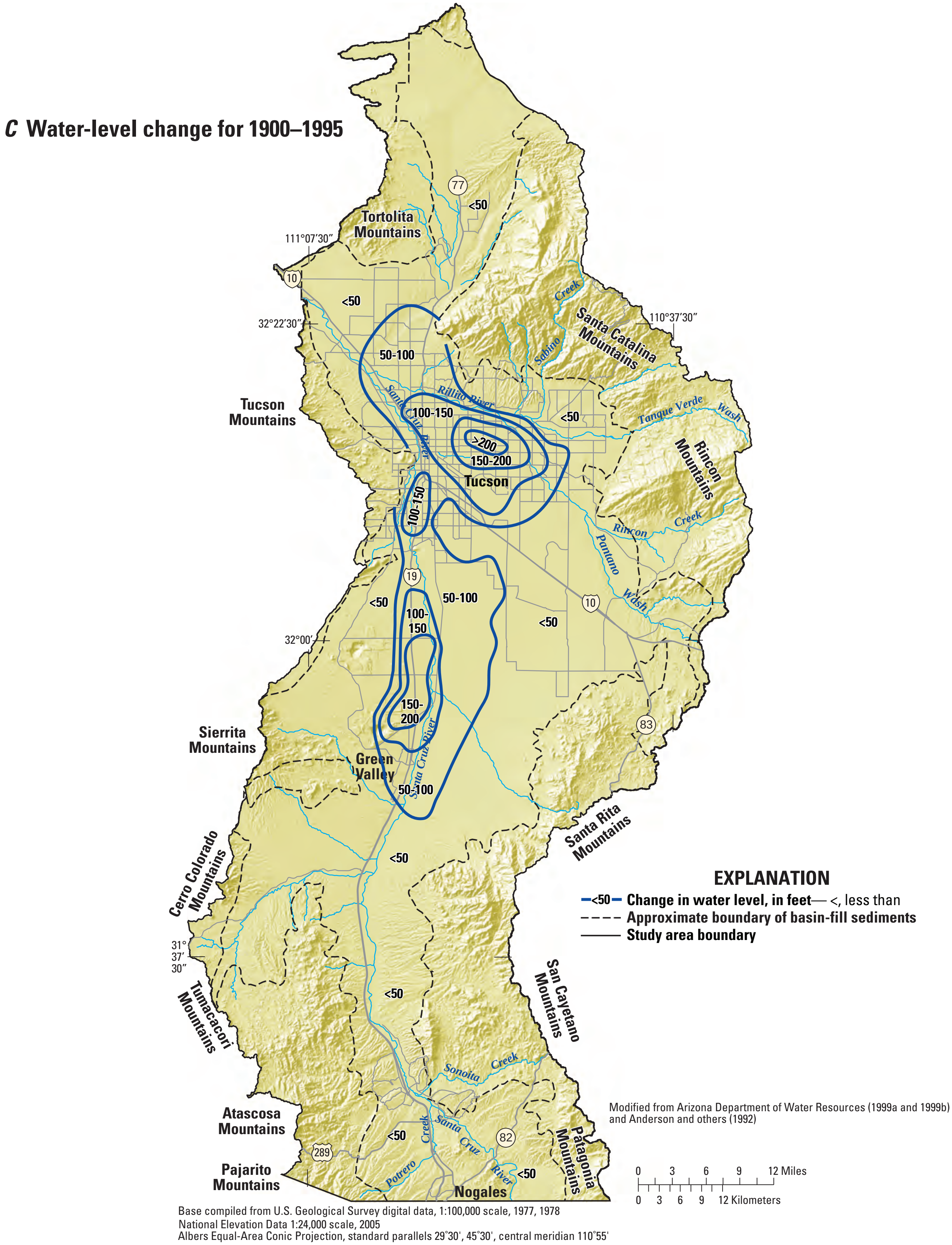


Figure 5. Water levels in the basin-fill aquifer of the Upper Santa Cruz Basin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1995. (*C*) Water-level change for 1900–1995. (*D*) Depth to water, 1995—Continued.

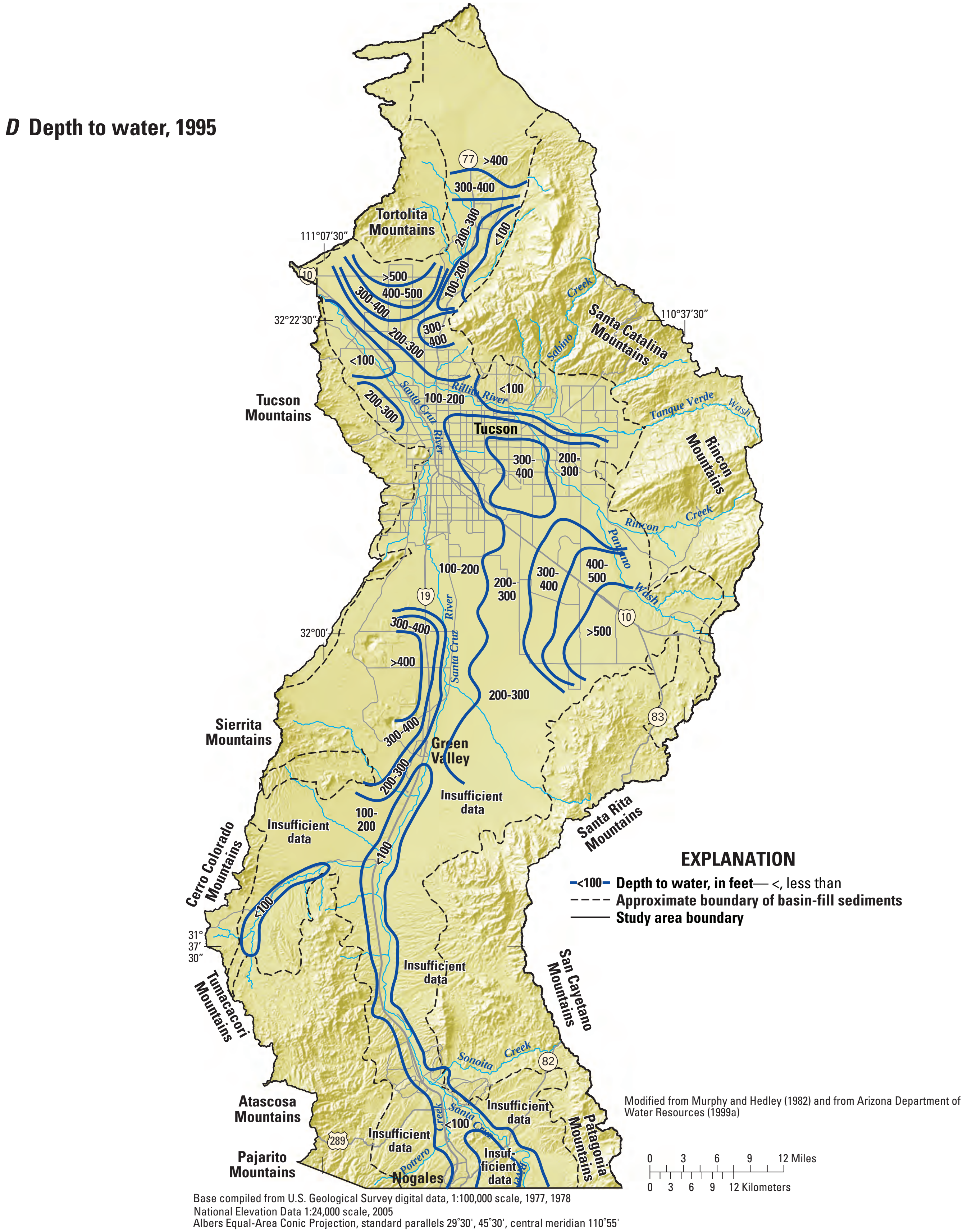


Figure 5. Water levels in the basin-fill aquifer of the Upper Santa Cruz Basin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 1995. (*C*) Water-level change for 1900–1995. (*D*) Depth to water, 1995—Continued.

Effects of Natural and Human Factors on Groundwater Quality

The quality of water in the basin-fill aquifer system in the Upper Santa Cruz Basin is affected by both natural and human-related factors. That water quality is characterized here on the basis of the analyses of samples collected from 58 wells in 1998 as part of the U.S. Geological Survey's (USGS) National Water Quality Assessment (NAWQA) Program (Coes and others, 2000; and Gellenbeck and Anning, 2002). Results of those analyses provided the information that enables an assessment of general chemical parameters, as well as the presence and concentrations of major ions, nutrients, trace constituents, pesticides, and volatile organic compounds relative to location, depth, land use, and geology. Half (29) of the wells were sampled by the ADEQ and the samples were analyzed for major ions, nutrients, trace constituents and tritium. The other 29 wells were sampled by NAWQA scientists and the samples were analyzed for the same constituents, as well as for pesticide and volatile organic compounds (fig. 6). The wells generally were used for domestic or commercial purposes, and all were completed within the developed part of the basin-fill aquifer.

General Groundwater-Quality Characteristics and Effects of Natural Factors

In a broad sense, groundwater in the Upper Santa Cruz Basin is suitable for industrial, agricultural, and municipal consumption, although some areas have water-quality concerns. Seventeen of the 58 wells sampled (29 percent) in 1998 contained one or more constituents at concentrations that exceeded a U.S. Environmental Protection Agency (USEPA) drinking-water standard (fig. 6; U.S. Environmental Protection Agency, 2009). Concentrations exceeded the USEPA primary drinking-water standards for arsenic (10 µg/L) in 7 wells, for fluoride (4 mg/L) in 1 well, and for nitrite plus nitrate (10 mg/L) in 5 wells (fig. 6). The USEPA secondary drinking-water standards were exceeded in 1 well each for iron (300 µg/L), and manganese (50 µg/L), in 2 wells each for pH (6.5 to 8.5 standard units), fluoride (2 mg/L), and sulfate (250 mg/L), and in 14 wells for dissolved solids (500 mg/L; fig. 6). Samples from 8 of 29 wells (28 percent) contained detectable levels of up to 5 pesticides. Volatile organic compounds (VOC) were detected in samples from 15 of 29 wells (52 percent; fig. 6). Analysis of the land use, hydrogeology, and water chemistry indicated that both natural and human-related factors influenced the presence and levels of contaminants in groundwater in the Upper Santa Cruz Basin.

The groundwater of the Upper Santa Cruz Basin is a calcium bicarbonate type, with a median dissolved-solids concentration of 305 mg/L (table 2). The water typically is slightly alkaline, and the median pH was 7.3 standard units. The middle 80 percent of the pH values, excluding the top and bottom 10 percent, were between 6.9 and 7.7 standard units (table 2). The median temperature was 77°F, and the middle 80 percent of the wells had temperatures between 67 and 86°F (table 2).

The most important natural control on groundwater quality in the Upper Santa Cruz Basin is its geology (Coes and others, 2000). Some natural controls, however, do not exhibit statistically significant relations to water quality. Concentrations of major ions, nitrate, and fluoride were not found to be statistically related to the mineralogy of the basin-fill unit or distance from the alluvium of the Santa Cruz River channel (Coes and others, 2000). Additionally, water samples from wells both north and south of the inferred fault exhibited statistically similar chemical characteristics.

Significant differences in the concentrations of dissolved solids, alkalinity, calcium, potassium, chloride, and sulfate were observed between wells less than 1.25 mi (2 km) from major faults in the basin fill and wells greater than 1.25 mi from those faults (faults shown in fig. 1; Coes and others, 2000). These findings corroborate those by Laney (1972), who noted that the concentrations of these constituents were elevated near the Santa Cruz Fault and attributed that fact to upward migration of water from gypsiferous mudstones of the Tinaja beds. Laney also found elevated concentrations of the same constituents downgradient of an area of gypsiferous rocks of the Pantano Formation in an area east of central Tucson.

The median arsenic concentration was 3 µg/L (table 2), and samples from seven wells exceeded the USEPA drinking-water standard for arsenic of 10 µg/L (fig. 6). The occurrence of arsenic in groundwater in Arizona is not considered unusual in that its source is presumed to be minerals in the basin-fill deposits that originated from hydrothermal sulfide and arsenide deposits in the surrounding mountains (Robertson, 1991). Six of the wells with elevated concentrations of arsenic are within about 3 mi of volcanic rocks, which can contain arsenic-bearing minerals (Coes and others, 2000; Welch and others, 1988) and which could be the parent rock for the basin-fill deposits (fig. 6).

The median fluoride concentration was 0.48 mg/L (table 2), and samples from two wells contained fluoride concentrations higher than the USEPA secondary drinking-water standard of 2 mg/L. In both cases, the likely source was attributed to dissolution and/or exchange reactions between groundwater and aquifer materials (Coes and others, 2000; Laney 1972). The Tucson Mountains are primarily volcanic in origin, and fluoride-bearing minerals are abundant in these rocks. Downgradient clays may have exchangeable fluoride adsorbed to ion-exchange sites.

Uranium was detected in samples from all but 4 of the 29 wells sampled for such analysis, and the median concentration was 3.1 µg/L (table 2; data from Tadayon and others, 1999). The largest concentration of uranium detected was 30 µg/L, which is the USEPA primary drinking-water standard. Geologic controls on uranium in basin-fill aquifers of the Upper Santa Cruz Basin, however, were not assessed.

Base compiled from U.S. Geological Survey digital data, 1:100,000 scale, 1977, 1978
National Elevation Data 1:24,000 scale, 2005
Albers Equal-Area Conic Projection, standard parallels 29°30', 45°30', central meridian 110°55'

Figure 6. Elevated concentrations and detections of selected compounds in groundwater samples from the Upper Santa Cruz Basin, Arizona, 1998.

Table 2. Summary of groundwater-quality data, Upper Santa Cruz Basin, Arizona, 1998.

[Constituents are dissolved. N/A, not applicable; mg/L, milligrams per liter; µg/L, micrograms per liter; MRL, minimum reporting level. Data from Coes and others (2000) and Gellenbeck and Anning (2002).]

	Number		Minimum reporting level		Percentiles				
	Wells	Detections	Highest	Lowest	10th	25th	50th	75th	90th
pH (standard units)	58	58	N/A	N/A	6.9	7.1	7.3	7.5	7.7
Temperature (degrees Fahrenheit)	58	58	N/A	N/A	67	71	77	82	86
Dissolved oxygen (mg/L)	26	26	N/A	0.1	1.5	3.1	4.3	4.8	6.2
Dissolved solids (mg/L)	58	58	10	1	169	218	305	478	621
Nitrate plus nitrite (mg/L as nitrogen)[1]	58	58	0.05	0.02	0.44	0.68	1.50	3.10	6.90
Phosphorus (mg/L)[1]	58	18	0.020	0.010	[3]0.0003	[3]0.001	[3]0.005	0.030	0.110
Arsenic (µg/L)[4]	55	27	10	1	[3]0.7	[2]2	[2]3	[2]6	12
Barium (µg/L)[4]	55	27	100	1.0	[2]7.1	[2]17	[2]27	[2]48	102
Chromium(µg/L)[1]	55	26	10	1.0	[2]1.6	[2]2.0	[2]2.4	[2]3.0	[2]3.6
Copper (µg/L)[4]	55	11	10	1.0	[2]0.4	[2]0.6	[2]0.9	[2]1.2	[2]1.8
Fluoride (mg/L)[4]	58	54	0.2	0.1	[2]0.17	0.35	0.48	0.65	1.2
Iron (µg/L)[4]	55	19	100	10	[3]1	[3]3	[2]11	[2]23	[2]55
Manganese (µg/L)[1]	55	17	50	1.0	[3]0.2	[3]0.5	[2]1.4	[2]4.5	[2]12
Zinc (µg/L)[4]	55	49	50	1.0	[2]25	[2]38	86	150	320
Uranium (µg/L)	28	24	1	1	[3]0.5	1.3	3.1	7.9	15.9

Notes on other constituents:

Trace constituents—More than 80 percent of the 55 wells with analyses were reported below the highest and lowest MRLs (in parentheses) antimony (5 and 1 µg/L), beryllium (1 and 0.5 µg/L), cadmium (1 and 1 µg/L), lead (5 and 1 µg/L), selenium (5 and 1 µg/L)), and silver (1 and 1 µg/L).

Pesticides—Of the 29 wells analyzed for 86 pesticide compounds, there were 5 compounds detected amongst 8 wells. Compounds included deethylatrazine (6 wells); atrazine (5 wells); and prometon, 2,4-D, and diuron (1 well each).

Volatile organic compounds—Of the 29 wells analyzed for 86 compounds, there were 11 compounds detected amongst 15 wells. Compounds included trichloromethane (7 wells); chloromethane and 1,4-dichlorobenzene (5 wells each); tetrachloroethylene (4 wells); methylbenzene (3 wells); bromodichloromethane and 1,2 dichlorobenzene (2 wells each); trichlorofluoromethane, dichlorodifluoromethane, trichloroethene, and 1,2,4-trimethylbenzene (1 well each).

[1]Summary statistics calculated using maximum likelihood estimation method (Cohen, 1959).

[2]Values are extrapolated between the two minimum reporting levels.

[3]Values are extrapolated below the lowest minimum reporting level.

[4]Summary statistics calculated using probability regression method (Cohen, 1959).

Potential Effects of Human-Related Factors

Human activities can influence groundwater quality, especially when altered land use is coincident with recharge areas. The recharge can carry dissolved contaminants to the aquifer that occur naturally or are introduced by activities at the land surface, such as the application of fertilizers to cropland or lawns. Groundwater quality can also be influenced by human activities in areas where recharge does not normally occur, especially when the contaminants are liquids, such as engine fuels or solvents used for commercial or industrial cleaning purposes.

Water samples were analyzed for tritium to identify wells that received recharge since the 1953 (See Section 1 of this report for a discussion of groundwater age and environmental tracers). Although not statistically tested, tritium detections tended to be in samples from wells near major streams or near the basin margins, where one would anticipate most recharge to the basin-fill aquifer takes place (fig. 6; table 1). Analysis of the locations of wells in which tritium, pesticide, and volatile organic compounds (VOCs) were detected demonstrates the susceptibility of the aquifer to contamination in areas that receive a component of recent recharge. Specifically, of the 12 wells with tritium detections and for which pesticide and VOC analyses are available, one or more pesticides or VOCs were detected in 9 wells (fig. 6). Therefore, 75 percent (9 of 12) wells that received recent recharge, as indicated by tritium detections, were contaminated with pesticides or VOCs.

The land uses that have the greatest potential to affect water quality in the Upper Santa Cruz Basin, on the basis of relative area, are urban and agriculture; mining also may play an important role, though the potential effects of mining on water quality were not evaluated by Coes and others (2000). For wells in urban areas, nitrate plus nitrite (as nitrogen) was elevated in samples of recently recharged water relative to concentrations in samples of "old" (pre-1950) groundwater. The urban areas offer several potential sources of nitrogen compounds, including treated wastewater effluent, lawn and garden fertilizers, and septic-tank systems. In addition, some areas that are currently urban were previously agricultural. Samples from two of five wells with concentrations of nitrate plus nitrite that exceeded the USEPA primary drinking-water standard for nitrate (10 mg/L) were associated with urban land use. One of these exceedences was likely related to wastewater released in Nogales Wash; the same well also contained manganese and dissolved-solids concentrations that exceeded the USEPA secondary drinking-water standards. The second well was in an urban area, but did not contain detectable tritium, and therefore the source of nitrate may be natural.

While the effects of the oxidation-reduction state of the groundwater samples collected in the Upper Santa Cruz River were not determined by Coes and others (2000), most of the nitrogen in the samples is in the form of nitrate, because the groundwater in the Upper Santa Cruz Basin is well oxygenated (data from Coes and others, 2000). The median dissolved-oxygen concentration was 4.3 mg/L (table 2), and concentrations of dissolved oxygen in all but 2 wells were greater than 1.0 mg/L. The oxidation-reduction state may also have affected concentrations of other constituents, such as arsenic, iron, and manganese.

Agriculture has long been practiced in the Upper Santa Cruz Basin, and although its effects on water quality are not widespread, pesticide detections were generally related to agricultural activities (Gellenbeck and Anning, 2002). Pesticides were detected in samples from 8 of 29 (8 percent) wells (fig. 6; table 2). The compounds detected included deethylatrazine, atrazine, prometon, 2,4-D, and diuron, although not all were found in each well. No pesticide concentrations exceeded any USEPA drinking-water standards. In 5 wells, both atrazine and and its degradation product, deethylatrazine, were detected. The herbicide atrazine is used both in agricultural and nonagricultural settings. Owing to their persistence and moderate to high mobility in the subsurface, detections of these two compounds are expected in areas where atrazine is applied. Three of the 5 wells in which atrazine and deethylatrazine were detected are co-located with historical agricultural areas where elevated concentrations of calcium, potassium, alkalinity, and dissolved solids also have been found (Coes and others, 2000). The remaining two wells are not directly adjacent to current or historically agricultural areas; the pesticides in water from those wells may have been transported from agricultural areas by the Santa Cruz River, or they may have come from pesticide use in urban areas.

VOCs are generally indicative of urban activities, and one or more compounds were found in samples from 15 of 29 wells (52 percent) analyzed for VOCs (fig. 6; table 2; Gellenbeck and Anning, 2002). Compounds detected included:

Trichloromethane (chloroform; 7 samples)	1,2 dichlorobenzene (2 samples)
Chloromethane (5 samples)	Trichlorofluoromethane (CFC-11; 1 sample)
1,4,-dichlorobenzene (5 samples)	Dichlorodifluoromethane (CFC-12; 1 sample)
Tetrachloroethylene (PCE; 4 samples)	Trichloroethene (TCE; 1 sample)
Methylbenzene (3 samples)	1,2,4-trimethylbenzene (1 sample)
Bromodichloromethane (2 samples)	

Two wells had samples with 5 VOC detections, while samples from the remaining wells had less detections. The concentration of trichloromethane in one well exceeded the drinking-water standards for that compound established by the USEPA.

Detections of VOCs were qualitatively related to land use in some, but not all, cases (Gellenbeck and Anning, 2002), yet such detections substantiate the potential for activities at the surface to cause the contamination of the underlying groundwater. For one well near the Mexico border, VOC detections were hypothesized to be related to its location near Nogales Wash, where VOCs including trichloroethene and many of its degradation products have been detected previously in surface-water samples. For another well, VOC detections were attributed to its location downgradient both from municipal wastewater releases to the Santa Cruz River and from a landfill near the river. Yet for a third well located in a newly developed residential area that was previously used for rangeland; no definitive sources of VOCs were identified.

Trichloromethane, also known as chloroform, was detected in samples from 7 wells. Chloroform, which is used as a solvent, is also a byproduct of the chlorination of water delivered for public supply. It may enter the ground through lawn irrigation, leaking water mains, and sewers (Squillace and others, 1999).

Analysis of data on major ions, nutrients, and selected trace constituents for six wells sampled annually from the 1980s to 1998 indicated notable trends (Coes and others, 2000). Concentrations of nitrate plus nitrite increased at one well in an area of continued agriculture and decreased in a well where urban development had replaced agriculture. Nitrate plus nitrite concentrations increased, however, at a well where land was converted from rangeland to urban use; lawn fertilizers are thought to contribute to this trend. Concentrations of constituents did not change significantly at three additional wells: one located where land use has been consistently agricultural, one where land use has changed from rangeland to urban, and one where land use has changed from agricultural to urban.

Summary

The Upper Santa Cruz Basin in the Basin and Range Physiographic Province of south central Arizona consists of an elongated sediment-filled valley surrounded by mountain ranges. The basin has a warm arid to semiarid climate, but wide variations in elevation cause large differences in precipitation and temperature. Land use in the basin is predominantly rangeland and urban, with other land uses and land covers including agriculture being minor. In the late 1990s, estimated groundwater withdrawals were about 221,000 acre-ft/yr, of which about 55 percent was for municipal uses, 26 percent for industrial uses, and 19 percent for agricultural uses. Other water sources include treated municipal wastewater and water imported from the Colorado River by the Central Arizona Project.

The basin-fill aquifer north of an inferred fault across the valley consists of unconsolidated to semiconsolidated sediments of the Pantano Formation, Tinaja beds, and Fort Lowell Formations. South of the inferred fault, the basin is filled by unconsolidated to semiconsolidated sediments of the Nogales Formation, older alluvium, and younger alluvium. Water levels in the basin-fill aquifer generally reflect the configuration of the valley floor, and consequently groundwater flows from the basin margins toward the center and then northward along the basin axis. The aquifer is replenished primarily through mountain-front recharge, mountain-block recharge, water losses from the channel of the Santa Cruz River, and with water-resources development, incidental recharge from human activities. Water leaves the aquifer primarily through evapotranspiration, and with water-resources development, through pumping from wells.

The cumulative effects of development have caused substantial changes in the groundwater flow system. Estimated total recharge in the basin-fill aquifer has increased by about 50 percent, from 93,000 to 140,000 acre-ft/yr, and discharge has increased about 170 percent, from 93,000 to 252,000 acre-ft/yr as a result of the development. These increases in flux have implications for groundwater quality. The more water moving into the aquifer, especially if exposed to contaminant sources, the greater intrinsic susceptibility to contamination. Reversing groundwater gradients and thereby changing an area from a discharge area to recharge area which has occurred along the Santa Cruz River, creates new entryways to the aquifer for contaminants. The effects of development took the form of steeper vertical hydraulic gradients, thicker unsaturated zones, redirection of groundwater movement toward pumping centers, creation of perched water zones, reductions in aquifer transmissivity, land subsidence, and the capture of perennial streamflow and former riparian evapotranspiration along the channels of the Santa Cruz and Rillito Rivers.

Analyses of samples collected from 58 wells in 1998 as part of a cooperative investigation by the National Water-Quality Assessment Program and the Arizona Department of Environmental Quality indicates that the water in the basin-fill aquifer of the Upper Santa Cruz Basin generally is suitable for industrial, agricultural, and municipal uses, although some areas have water-quality concerns. About 29 percent of the wells samples contained one or more constituents or properties (such as arsenic, fluoride, nitrate, iron, manganese, pH, or dissolved solids) that exceeded a state or federal water-quality standard. In addition, samples from 28 percent of the wells contained detectable levels of pesticides and samples from 52 percent of the wells contained detectable levels of one or more volatile organic compounds (VOCs).

Analysis of the land use, hydrogeology, and the chemistry of groundwater in the Upper Santa Cruz Basin indicated that both natural and human-related factors influenced the presence and levels of contaminants in the water (table 3). Natural factors, primarily basin geology and the geochemical processes between the groundwater and basin-fill sediments, were attributed as the cause of elevated concentrations of dissolved solids, alkalinity, calcium, potassium, chloride, sulfate, arsenic, and fluoride. The increase in groundwater flux through the aquifer as a consequence of the development, use, and disposal of water has increased the intrinsic susceptibility of the aquifer to contamination from sources present or generated at the land surface. For example, the use of chemical compounds in urban and agricultural areas that receive focused recharge from the infiltration of excess irrigation water has resulted in the presence of pesticides and VOCs in the basin-fill aquifer.

Table 3. Summary of documented effects of natural and human-related factors on groundwater quality in the Upper Santa Cruz Basin, Arizona.

Groundwater-quality effect	Cause	General location(s)	Reference(s)
		Primarily natural factors	
Elevated concentrations of dissolved solids, alkalinity, calcium, potassium, chloride, and sulfate	Upward migration of groundwater from gypsiferous mudstones of the Tinaja beds	Within about 1.25 miles of major faults in basin-fill sediments, such as the Santa Cruz Fault	Laney (1972), Coes and others (2000)
Elevated concentrations of dissolved solids, alkalinity, calcium, potassium, chloride, and sulfate	Movement of groundwater through gypsiferous sediments of the Pantano Formation	Vail to central Tucson	Laney (1972)
Elevated concentrations of arsenic	Geochemical reactions between the groundwater and compounds in the basin fill that are presumed to come from hydrothermal sulfide and arsenide deposits in the surrounding mountains	Along the Santa Cruz River and near volcanic rocks in the mountains along the basin margins	Robertson (1991), Coes and others (2000)
Elevated concentrations of fluoride	Geochemical reactions between the groundwater and compounds in the basin fill that are presumed to come from fluoride-bearing minerals in volcanic rocks	Localized parts of basin	Laney (1972), Coes and others (2000)
		Primarily human-related factors	
Elevated concentrations of nitrate	Application of nitrogen fertilizers and irrigation of crops and urban landscaped areas; infiltration from septic tanks and treated wastewater released to the Santa Cruz River	Localized parts of basin	Coes and others (2000)
Occurrence of pesticides	Application of pesticide compounds to croplands and urban landscaped areas	Agricultural and urban areas	Gellenbeck and Anning (2002)
Occurrence of volatile organic compounds	Urban and industrial activities on the land surface and subsequent transport of compounds to aquifer	Urban areas	Gellenbeck and Anning (2002)
Occurrence of pesticides and volatile organic compounds	Urban or agricultural use of organic compounds	Areas susceptible to contamination, especially those which receive modern (post-1950) focused recharge, such as along streams and irrigated areas	This study

References Cited

Adams, D.K., and Comrie, A.C., 1997, The North American monsoon: Bulletin of the American Meteorological Society, v. 78, p. 2197–2213.

Anderson, S.R., 1987, Cenozoic stratigraphy and geologic history of the Tucson Basin, Pima County, Arizona: U.S. Geological Survey Water-Resources Investigations Report 87–4190 28 p.

Anderson, S.R., 1988, Potential for aquifer compaction, land subsidence, and earth fissures in the Tucson basin, Pima County, Arizona: U.S. Geological Survey Hydrologic Investigations Atlas HA–713, 3 sheets.

Anderson, T.W., Freethey, G.W., and Tucci, P., 1992, Geohydrology and water resources of alluvial basins in south-central Arizona and parts of adjacent states: U.S. Geological Survey Professional Paper 1406–B, 67 p., 3 plates, scale 1:250,000.

Anning, D.W., and Duet, N.R., 1994, Summary of ground-water conditions in Arizona, 1987–1990: U.S. Geological Survey Open File Report 94–476, 1 plate.

Arizona Department of Water Resources, 1999a, Third Management Plan 2000–2010 for the Tucson Active Management Area, Chapter 2: Overview of Water Resources, Arizona Department of Water Resources, accessed December 29, 2008 at http://www.azwater.gov/dwr/content/Publications/files/ThirdMgmtPlan/tmp_final/tucson/ch2-tuc.pdf.

Arizona Department of Water Resources, 1999b, Third Management Plan 2000–2010 for the Santa Cruz Active Management Area, Chapter 2: Overview of Water Resources, Arizona Department of Water Resources accessed Dec. 29, 2008 at http://www.azwater.gov/dwr/content/Publications/files/ThirdMgmtPlan/tmp_final/santa_cruz/ch2-sc.pdf.

Betancourt, J.L., and Turner, R.M., 1993, Tucson's Santa Cruz River and the Arroyo Legacy: University of Arizona.

Carruth, R.L., 1995, Cross-sectional area of the shallow alluvial aquifer at selected sites along the Upper Santa Cruz River, Arizona: U.S. Geological Survey Water-Resource Investigations Report 95–4112.

Central Arizona Project, 2007, About CAP—History, accessed on December 1, 2010 at http://www.cap-az.com/about-cap/history/.

Coes, A.L., Gellenbeck, D.J., Towne, D.C., and Freark, M.C., 2000, Ground-water quality in the Upper Santa Cruz Basin, Arizona, 1998: U.S. Geological Survey Water-Resources Investigations Report 00–4117, 55 p. Available at http://az.water.usgs.gov/pubs/WRIR00-4117intro.html.

Cohen, A.C., 1959, Simplified estimators for the normal distribution when samples are singularly censored or truncated: Technometrics, v. 1, no. 3. p. 217–237.

Davidson, E.S., 1973, Geohydrology and water resources of the Tucson Basin, Arizona: U.S. Geological Survey Water-Supply Paper 1939–E, 81 p.

Fenneman, N.M., 1931, Physiography of western United States: New York, McGraw-Hill, 534 p.

Freethey, G.W., and Anderson, T.W., 1986, Predevelopment hydrologic conditions in the alluvial basins of Arizona and adjacent parts of California and New Mexico: U.S. Geological Survey Hydrologic Investigations Atlas HA–664, 3 sheets.

Gellenbeck, D.J., and Anning, D.W., 2002, Occurrence and distribution of pesticides and volatile organic compounds in ground water and surface water in Central Arizona Basins, 1996–98, and their relation to land use: U.S. Geological Survey Water-Resources Investigations Report 01–4144, 101 p. Available at http://az.water.usgs.gov/pubs/WRIR01-4144intro.html.

Gelt, J., Henderson, J., Seasholes, K., Tellman, B., and Woodard, G., 2006, Water in the Tucson area: Seeking sustainability: Water Resources Research Center on-line report, accessed January 18, 2006 at http://ag.arizona.edu/AZWATER/publications/sustainability/.

Gettings, M.E., Houser, B.B., 1997, Basin geology of the Upper Santa Cruz Valley, Pima and Santa Cruz Counties, southeastern Arizona: U.S. Geological Survey Open-File Report 97–676.

Halpenny, L.C., 1963, Geophysical and geohydrological investigations of Santa Cruz River Valley, Arizona, international boundary to mouth of Sonoita Creek: Tucson, Arizona, Water Development Corporation, 57 p.

Halpenny, L.C., and Halpenny, P.C., 1988, Review of the hydrogeology of the Santa Cruz Basin in the vicinity of the Santa Cruz-Pima County line: Tucson, Arizona, Water Development Corporation, 59 p.

Hanson, R.T., 1989, Aquifer-system compaction, Tucson Basin and Avra Valley, Arizona: U.S. Geological Survey Water-Resources Investigations Report 88–4172, 75 p.

Hanson, R.T., and Benedict, J.F., 1994, Simulation of Ground-water flow and potential land subsidence, Upper Santa Cruz Basin, Arizona: U.S Geological Survey Water-Resources Investigations Report 93–4196, 46 p.

Hirschberg, D.M., and Pitts, S.G., 2000, Digital geologic map of Arizona: a digital database from the 1983 printing of the Wilson, Moore, and Cooper 1:500,000 scale map of Arizona: U.S. Geological Survey Open-File Report 2000–409. Available at http://geopubs.wr.usgs.gov/open-file/of00-409/.

Klimas, T., Williams, C., and Thiel, J.H., 2006, Future directions: part 6. Canals, AZ BB:13:481 *in* J. Homer Thiel and Jonathan B. Mabry (editors), Rio Nuevo Archaeology Program, 2000–2003: Investigations at the San Agustin Mission and Mission Gardens, Tucson Presidio, Tucson Pressed Brick Company, and Clearwater Site, Center for Desert Archaeology, Tucson, Arizona,Technical Report No. 2004–11.

Laney, R.L., 1972, Chemical quality of the water in the Tucson Basin: U.S Geological Survey Water-Supply Paper 1939–D, 46 p.

Mason, D.A., and Bota, L., 2006, Regional groundwater flow model, Tucson Active Management Area, Tucson, Arizona: simulation and application: Arizona Department of Water Resources Modeling Report 13, 159 p.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the Southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

Murphy, B.A., and Hedley, J.D., 1984, Maps showing groundwater conditions in the Upper Santa Cruz Basin Area, Pima, Santa Cruz, Pinal, and Cochise Counties, Arizona—1982: Arizona Department of Water Resources Hydrologic Map Series Report Number 11, 3 plates, scale 1:125,000.

Nelson, K., 2007, Groundwater flow model of the Santa Cruz Active Management Area along the effluent-dominated Santa Cruz River, Santa Cruz and Pima Counties, Arizona: Arizona Department of Water Resources Modeling Report 14, 154 p.

Nelson, K., and Erwin, G., 2001, Santa Cruz active management area 1997–2001 hydrologic monitoring report: Arizona Department of Water Resources Monitoring Report, 44 p.

Robertson, S. J., 1991, Geochemistry of ground water in alluvial basins of Arizona and adjacent parts of Nevada, New Mexico, and California: U.S. Geological Survey Professional Paper 1406–C, 90 p.

Scanlon, B.R., Langford, R.P., and Goldsmith, R.S., 1999, Relationship between geomorphic settings and unsaturated flow in an arid setting: Water Resources Research, v. 35, p. 983–999.

Squillace, P.J., Moran, M.J., Lapham, W.W., Price, C.V., Clawges, R.M., and Zogorski, J.S., 1999, Volatile organic compounds in untreated ambient ground water of the United States, 1985–95: Environmental Science and Technology, v. 33, no. 23, p. 4176–4187.

Tadayon, S., Duet, N.R., Fisk, G.G., McCormack, H.F., Partin, C.K., Pope, G.L., and Rigas, P.D., 1999, Water resources data, Arizona, water year 1998: U.S. Geological Survey Water-Data Report AZ–98–1, 454 p.

Thiel, J.H., and Diehl, M.W., 2006, Cultural history of the Tucson Basin and the project area *in* J. Homer Thiel and Jonathan B. Mabry (editors), Rio Nuevo Archaeology Program, 2000–2003: Investigations at the San Agustin Mission and Mission Gardens, Tucson Presidio, Tucson Pressed Brick Company, and Clearwater Site, Center for Desert Archaeology, Tucson, Arizona,Technical Report No. 2004–11.

Tucson Water, 1993, Annual static water level basic data report, Tucson Basin and Avra Valley, Pima County, Arizona: City of Tucson, Tucson Water Planning and Technical Services Division report, 134 p.

Unland, H.E., Arain, A. M., Harlow, C., Houser, P.R., Garatuza-Payan, J., Scott, P., Sen, O. L., and Shuttleworth, J., 1998, Evaporation from a riparian system in a semi-arid environment: Hydrological Processes, v. 12, p. 527–542.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed March 3, 2009 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

Welch, A.H., Lico, M.S., and Hughes, J.L., 1988, Arsenic in Ground Water of the Western United States: Gound Water v. 26 no. 3, pp. 333–347.

Western Regional Climate Center, 2007, Climate data for Tucson, Arizona, accessed January 16, 2007 at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?az8800.

Section 9.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the Sierra Vista Subbasin of the Upper San Pedro Basin, Arizona

By David W. Anning and James M. Leenhouts

Basin Overview

The Sierra Vista subbasin (fig. 1) hosts a growing human population as well as a remarkable riparian ecosystem along the San Pedro River in southeastern Arizona. Groundwater in this subbasin is important because it is the primary source of water for the residents and also because it sustains the base flow of and the riparian ecosystem along the San Pedro River. Groundwater development to support the population and the economic and cultural activities over the past century has caused substantial changes in the basin-fill aquifer. These changes include a 38 percent increase in recharge and a 103 percent increase in discharge. These and other changes to the aquifer have resulted in an increase in its intrinsic susceptibility to contamination, and the effects of both natural and human-related factors on groundwater quality are discussed in this section of the report.

The Sierra Vista subbasin is a 1,826-mi^2 hydrogeologic area defined in McKinney and Anning (2009) and is roughly coincident with the Sierra Vista groundwater subbasin defined by the state of Arizona in the southern part of the upper San Pedro Basin. The United States-Mexico border forms the southern study area boundary and excludes about 700 mi^2 of the subbasin in Mexico that drains northward (fig. 1). The state of Arizona further divides the Sierra Vista groundwater subbasin into two surface-water drainages: the Sierra Vista subwatershed to the south and the Benson subwatershed to the north (fig. 1). The results of several hydrologic studies of these two subwatersheds are integrated in this discussion.

The San Pedro River drains both the surface and groundwater systems of the Sierra Vista subbasin and is perennial for about 11 mi (Leenhouts and others, 2006). One tributary to the San Pedro River, the Babocomari River, also includes perennial reaches. Nearly all other reaches of, and tributaries to, the San Pedro, with the exception of short reaches in the mountains, are ephemeral. An act of Congress in 1988 formally protected much of the riparian ecosystem as the San Pedro Riparian National Conservation Area (fig. 1), which is now managed by the U.S. Bureau of Land Management. The biological importance of the river stems from the ecosystem contrast between the riparian corridor and the surrounding area. The riparian corridor supports a diverse biota and is a primary corridor for migrating birds. The riparian corridor provides habitat for more than 400 bird species, and the Sierra Vista subbasin supports the second highest known number of mammal species in the world (Goodrich and others, 2000).

The climate in the Sierra Vista subbasin is semiarid, but a wide range in altitude causes significant variations in precipitation and temperature. Altitude along the river ranges from 4,300 ft at the United States-Mexico border in the south to 3,300 ft at the downstream end of the basin in the north, and the highest altitudes extend to 9,500 ft in the Huachuca Mountains. Annual rainfall averages about 30 in. in the mountains and about 12 in. on the low basin floor (Leenhouts and others, 2006).

Temperatures in the Sierra Vista subbasin range from a mean maximum temperature of 80°F to a mean minimum temperature of 45°F (1971–2000 averages recorded in Benson). Annual precipitation amounts for 1971–2000 are 12.3 in. in Benson, 14 in. in Tombstone, and 15.2 in. in Sierra Vista, though rainfall in this area is highly variable, both spatially and temporally. About 25 percent of the average annual precipitation is attributed to winter frontal storms during November through February that typically are longer in duration and less intense than storms during the remainder of the year. During winter, most of the vegetation is inactive and nighttime frosts are common. During April through June, days are typically dry and hot. During July through September, the Sierra Vista subbasin is under the influence of the North American Monsoon (Adams and Comrie, 1997), which brings in moist subtropical air that combines with intense surface heating to generate high intensity, typically short duration convective storms. About 60 percent of the annual precipitation in the valley occurs during the monsoon (Goodrich and others, 2000).

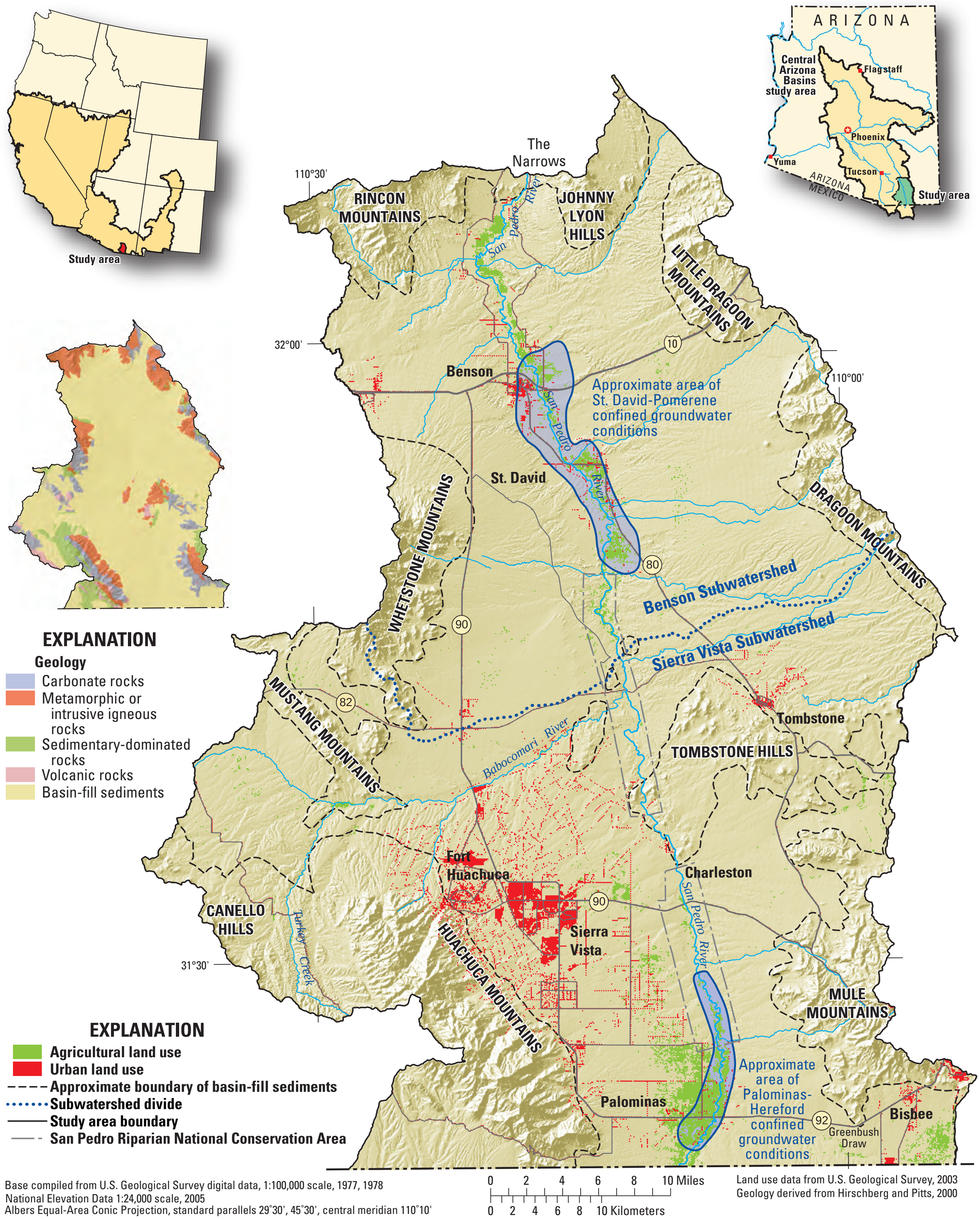


Figure 1. Physiography, land use, and generalized geology of the Sierra Vista subbasin, Arizona.

As is typical for semiarid to arid regions, potential evaporation in the Sierra Vista subbasin exceeds normal annual precipitation. A pan evaporation study at a site along the San Pedro River measured a total open-water evaporation rate of 46 in. for 2003, whereas the potential evapotranspiration was 70 in. (Scott and others, 2006). As a result of the evaporation excess, recharge is generally thought not to occur through the open desert floor, but instead is concentrated at the mountain fronts or at other locations where water collects, even if only temporarily (Scanlon and others, 1999).

Land use in the alluvial basin, excluding the surrounding mountainous areas, includes about 3 percent urban and 3 percent agricultural lands (McKinney and Anning, 2009). Land use patterns on the valley floor of the Sierra Vista subbasin have become increasing urban over the last several decades, particularly in the area of Sierra Vista and Fort Huachuca (fig. 1). Some of this urbanization has occurred to provide housing and support services for Fort Huachuca. In addition, the pleasant climate and environs have made the area a retirement destination. Sierra Vista was incorporated in 1956 with 1,671 people, and in 2000 hosted a population of 37,775 (includes Fort Huachuca; City of Sierra Vista, 2006). On the basis of 2000 data, the total population of the Sierra Vista subbasin is estimated to be about 80,000 people (McKinney and Anning, 2009).

Water Development History

The Sierra Vista subbasin has a long history of human habitation and water development. Although the earliest Paleo-Indian sites date from the late Pleistocene (Haynes, 1987), the human activities that have affected the water resources of the region likely occurred within the past 400 to 500 years. During this period, the valley was explored, settled, and exploited primarily by three cultures—the Spanish, Mexican, and Anglo (Trischka, 1971; Hereford, 1993). The area has also hosted Native American populations such as the Sobaipuri Indians, who grew crops in irrigated fields.

The Spanish exploratory expeditions of the 1600s and 1700s kept the first records of water use in the Sierra Vista subbasin. Around this time, a population of about 2,000 Sobaipuri Indians was observed to be farming using diversions from the San Pedro River for irrigation (Arizona Department of Water Resources, 2005a).

Padre Eusebio Francisco Kino led the first Christian missions into the area in the late 1600s and early 1700s, and is credited with establishing cattle ranching in the San Pedro Valley (Hereford, 1993). Attempts to attract settlers to the area in significant numbers were unsuccessful until the late 19th century owing to Apache depredations (Hereford, 1993), although some ranching and farming were practiced through the 1800s. Although these agricultural industries undoubtedly used water, information about the amounts, locations, and character of such use is scarce. Fort Huachuca was established in 1877 to provide a base for protection of settlers and has remained an important part of the area's population to the current day.

Utilization of water began to increase significantly in the late 1800s and was dominated until about the mid 1980s by two industries: mining and agriculture. The discovery of lead, copper, and silver deposits near Tombstone and Bisbee in the late 1870s initiated significant settlement and development of the Sierra Vista subbasin (Rodgers, 1965). Although Tombstone's boom was short-lived, around 1880 it was briefly the largest town in Arizona with a population of about 15,000 (Arizona Department of Water Resources, 1991). During development of Tombstone's mine in 1881, workers struck water at 520 ft below land surface. Water was removed from the mine at an estimated 1,000 gallons per minute; Tombstone Mayor John Clum urged residents to water their lawns with the excess water (Arizona Department of Water Resources, 1991). Removal of water from the mine ended temporarily in 1886 when a fire destroyed the pump works. Pumping was reinitiated from about 1902 to 1911, with maximum withdrawals of about 6,000 acre-ft in 1910. A brief period of minor pumping from the mine occurred around 1955. The other significant mining operation was the Copper Queen mine in Bisbee. Rich ore deposits were first discovered in the Mule Mountains near Bisbee in the late 1870s. Withdrawals of water from the mine began in 1905 and quickly increased to about 6,000 acre-ft/yr. Maximum annual withdrawals exceeded 10,000 acre-ft/yr in the 1940s, and pumping ceased in about 1987 (Pool and Dickinson, 2007). The area of the Copper Queen mine straddles the Upper San Pedro and Douglas Basin divide, and it is likely that some portion of water drawn from the mine was Douglas-Basin water. Mining was a major industry in the United States portion of the basin through about 1985 and played a role in the establishment of several communities. Another large copper mine has pumped groundwater upgradient of the Sierra Vista subbasin near Canenea, Mexico (Pool and Dickinson, 2007).

Agricultural water use increased in the Sierra Vista subbasin from the late 1800s to about 1985, but generally decreased through 2006. The bulk of irrigated acreage, mostly alfalfa, has historically been in the northern half of the basin in the Benson subwatershed. The Arizona Department of Water Resources (1991) estimated that about 3,500 acres of land were under cultivation in 1899. By 1934, total cultivated acreage had increased to 4,200, of which about 3,300 acres were irrigated by diversions from the San Pedro River near St. David and Benson (Bryan and others, 1934). At this time, about 650 acres of alfalfa were irrigated near Bisbee using groundwater pumped from the copper mines. Areas of land irrigated using diversions from the San Pedro River were also noted in the Palominas-Hereford area, but were not quantified (Bryan and others, 1934).

In 1952, the area of cultivated land in the Sierra Vista subbasin was estimated at about 5,600 acres, with a net demand for groundwater of 14,500 acre-ft/yr (Heindl, 1952). All other uses were estimated at about 3,800 acre-ft/yr, for a total estimated basin use of 18,300 acre-ft/yr. By 1968, estimated total annual basin groundwater use was 35,300 acre-ft, including 22,100 acre-ft (62.6 percent) for agriculture, 6,600 acre-ft (18.7 percent) for mining and industrial uses, and 6,600 acre-ft (18.7 percent) for municipal and all other purposes (Roeske and Werrell, 1973). About 28,300 acre-ft of groundwater were pumped in the Sierra Vista subbasin in 1985, with about 13,300 acre-ft (47.0 percent) supporting agriculture, 13,000 acre-ft (45.9 percent) used for municipal purposes, and 2,000 acre-ft (7.1 percent) for industrial and other purposes (Arizona Department of Water Resources, 2005a).

After 1985, groundwater use for irrigation in the Sierra Vista subbasin declined owing to retirement of agricultural lands. Increases in population, however, caused increased groundwater pumping for municipal purposes. In 2002, total water use was 31,100 acre-ft, of which 27,800 acre-ft (89.4 percent) was supplied by groundwater (Arizona Department of Water Resources, 2005b). Total agricultural use was estimated at about 9,800 acre-ft (31.5 percent of total), with about 7,500 acre-ft (76.5 percent) being supplied by groundwater and 2,300 acre-ft (23.5 percent) from San Pedro River diversions. Total municipal water use was about 18,900 acre-ft (60.7 percent of total), of which 17,900 acre-ft (94.7 percent) was supplied by groundwater and the remaining 1,000 acre-ft (5.3 percent) supplied by surface water and treated municipal effluent. Other water use, including that by industry, was about 2,400 acre-ft (7.7 percent of total) and was supplied by groundwater.

Future water development in the Sierra Vista subbasin will likely be highly influenced by Section 321 of Public Law 108-136, a congressional directive to the residents of the Sierra Vista subwatershed that they determine and attain a sustainable yield of groundwater withdrawals by 2011.

Hydrogeology

The San Pedro River flows through typical basin and range physiography. Basins have formed in the grabens between block-faulted mountain ranges and have filled with Miocene through early Pleistocene sediments eroded from the uplifted blocks. The result is a series of roughly linear and parallel northwest-trending complexes of mountains and basins (Brown and others, 1966; fig. 1).

The Sierra Vista subbasin is bounded on the east by the Mule and Dragoon Mountains and on the west by the Huachuca and Whetstone Mountains (fig. 1). The Huachuca, Whetstone, and Dragoon Mountains, as well as the Rincon Mountains at the northwestern edge of the subbasin, are composed largely of granite, limestone, dolomite, conglomerate, and claystone ranging in age from Precambrian to Cretaceous (fig. 1; Drewes, 1996). The Mule Mountains consist of early Precambrian schist unconformably overlain by Mesozoic conglomerate, red mudstone, siltstone, and limestone (fig. 1; Hayes, 1970).

The earliest sedimentary unit in the basin is the Oligocene to lower Miocene Pantano Formation (fig. 2; Gettings and Houser, 2000). It is described by Brown and others (1966) as semiconsolidated brownish-red to brownish-grey conglomerate, and according to Gettings and Houser (2000), it is as much as 2,300-ft thick at the southern end of the study area. The Pantano Formation yields water through fractures to many wells in the Sierra Vista area and is an important water-bearing unit in some locations (Pool and Coes, 1999).

Alluvial sediments that are as much as about 750-ft thick overlie the Pantano Formation (Pool and Coes, 1999) and, for the purposes of this study, are subdivided into three groups: basin-fill sediments, terrace deposits, and stream alluvium (fig. 2). The basin-fill sediments are further divided into upper and lower units. The lower basin fill is Miocene to Pliocene in age and consists largely of interbedded gravel and sandstone, but can include clay, siltstone, and silt (Pool and Coes, 1999). Sorting in gravel beds and sandstones is generally poor, and the degree of cementation is variable (Brown and others, 1966). In most of the basin, the lower basin-fill sediments serve as an important water-bearing unit; its thickness ranges from about 150 to 350 ft. In the southern part of the subbasin, hydraulic conductivity values of the lower basin fill average about 3.2 ft/d for sand and gravel, about 2.6 ft/d for interbedded sand and gravel, and about 0.016 ft/d for silt and clay (Pool and Dickinson, 2007).

The upper basin-fill sediments consist of Pliocene- to Pleistocene-age reddish-brown clay, silt, sand, and gravel that are generally weakly cemented (Pool and Coes, 1999). The lithology grades from gravels with high permeability in the fan deposits along the flank of the Huachuca Mountains to relatively impermeable silts and clays near Charleston. Aquifer thickness is 400 ft or less. In the southern part of the subbasin, hydraulic conductivity values of the upper basin fill average about 11 ft/d for sand and gravel, about 2.9 ft/d for interbedded sand and gravel, and about 0.75 ft/d for silt and clay (Pool and Dickinson, 2007).

The terrace deposits began forming in the middle Pleistocene when changes in the climatic regime caused a transition from deposition to erosion (Brown and others, 1966). These deposits mark the location of the San Pedro River through the process of several episodes of downcutting, and extend from the base of the mountains to the San Pedro's current flood plain. The terrace deposits form a veneer near the mountains, but can be as much as 50 to 100 ft thick in erosional channels near the current San Pedro River (Pool and Coes, 1999). The sediments are a poorly sorted mixture of gravel, sand, and clay from local sources (Brown and others, 1966).

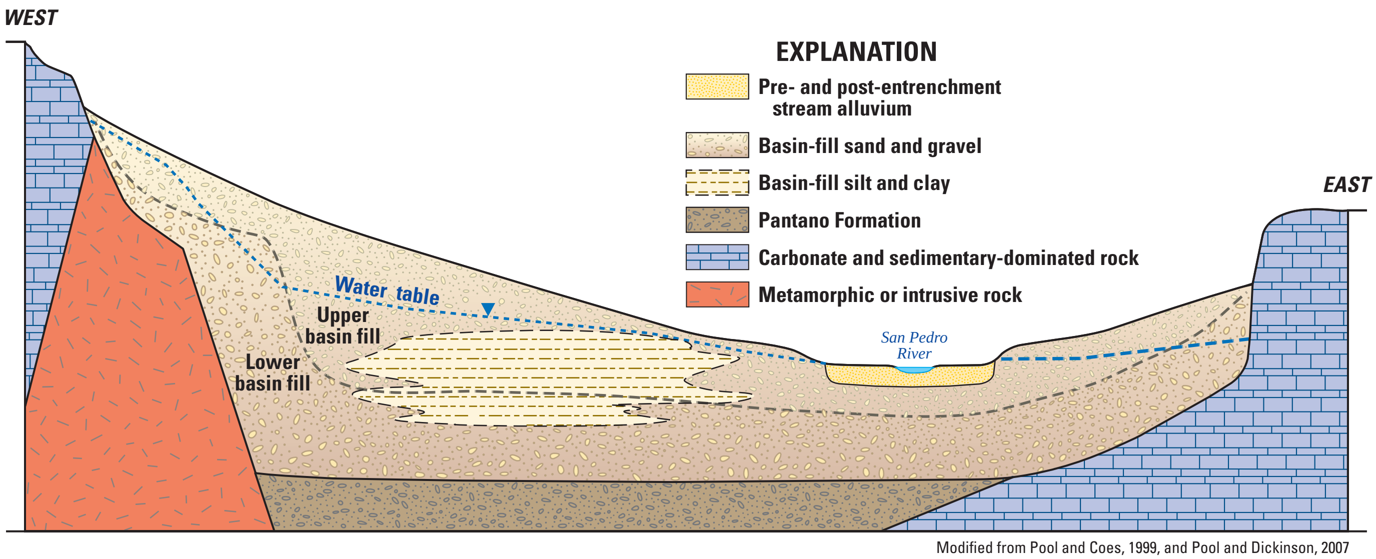


Figure 2. Generalized hydrogeologic cross section, Sierra Vista subbasin, Arizona.

The youngest terrace deposit comprises the modern stream alluvial sediments. The modern stream alluvium is subdivided into the pre-entrenchment and post-entrenchment units (fig. 2). They are Holocene in age, generally 20 ft or less in thickness, as much as 1 mi wide, and have average hydraulic conductivity values of about 25 ft/d (Pool and Dickinson, 2007). The post-entrenchment alluvium is equivalent to the present-day flood plain. The pre-entrenchment alluvium is at a higher altitude, is only rarely flooded, and is basically flat lying. The pre-entrenchment alluvium is also called the terrace. Portions of the pre-entrenchment terrace in the San Pedro Riparian National Conservation Area were cleared for agricultural use in the mid-20th century.

Conceptual Understanding of the Groundwater Flow System

The predevelopment groundwater flow system of the Sierra Vista subbasin resembles other basin and range systems in southern Arizona. Water levels in the basin-fill aquifer are generally parallel to the land surface of the valley floor, and consequently, groundwater flows from the basin margins toward the center and then northward along the basin axis. The aquifer is replenished primarily through mountain-front recharge, mountain-block recharge, water losses from the San Pedro River, and with water-resources development, incidental recharge from human activities. Water leaves the aquifer primarily through evapotranspiration, and with water-resources development, through groundwater pumping. Details of groundwater recharge, discharge, and flow are described in the following sections, along with the effects that water-resources development has had on the basin-fill aquifer.

Water Budget

A conceptual understanding of the primary and significant fluxes through the basin-fill aquifer is summarized in the groundwater budget for predevelopment and modern conditions in the Sierra Vista subbasin (fig. 3; table 1). Natural predevelopment flow in the aquifer is characterized by a predominance of recharge from stream-channel infiltration near the contact between the low-permeability rocks of the mountains and basin fill, and discharge along the San Pedro River either as contributions to stream baseflow or as evapotranspiration. This pattern is consistent with many arid to semiarid environs where orographically induced precipitation leads to excess available water in and near the mountains. Recent evidence, however, has suggested that about 12 to 19 percent of total recharge in the Sierra Vista subwatershed of the Sierra Vista subbasin occurs at some distance from the mountain front in ephemeral stream channels where runoff water is concentrated (Coes and Pool, 2005). Relatively deep groundwater levels across most of the basin prevent direct access by plant roots except near the river. In addition, a small fraction of total groundwater discharge occurs through springs.

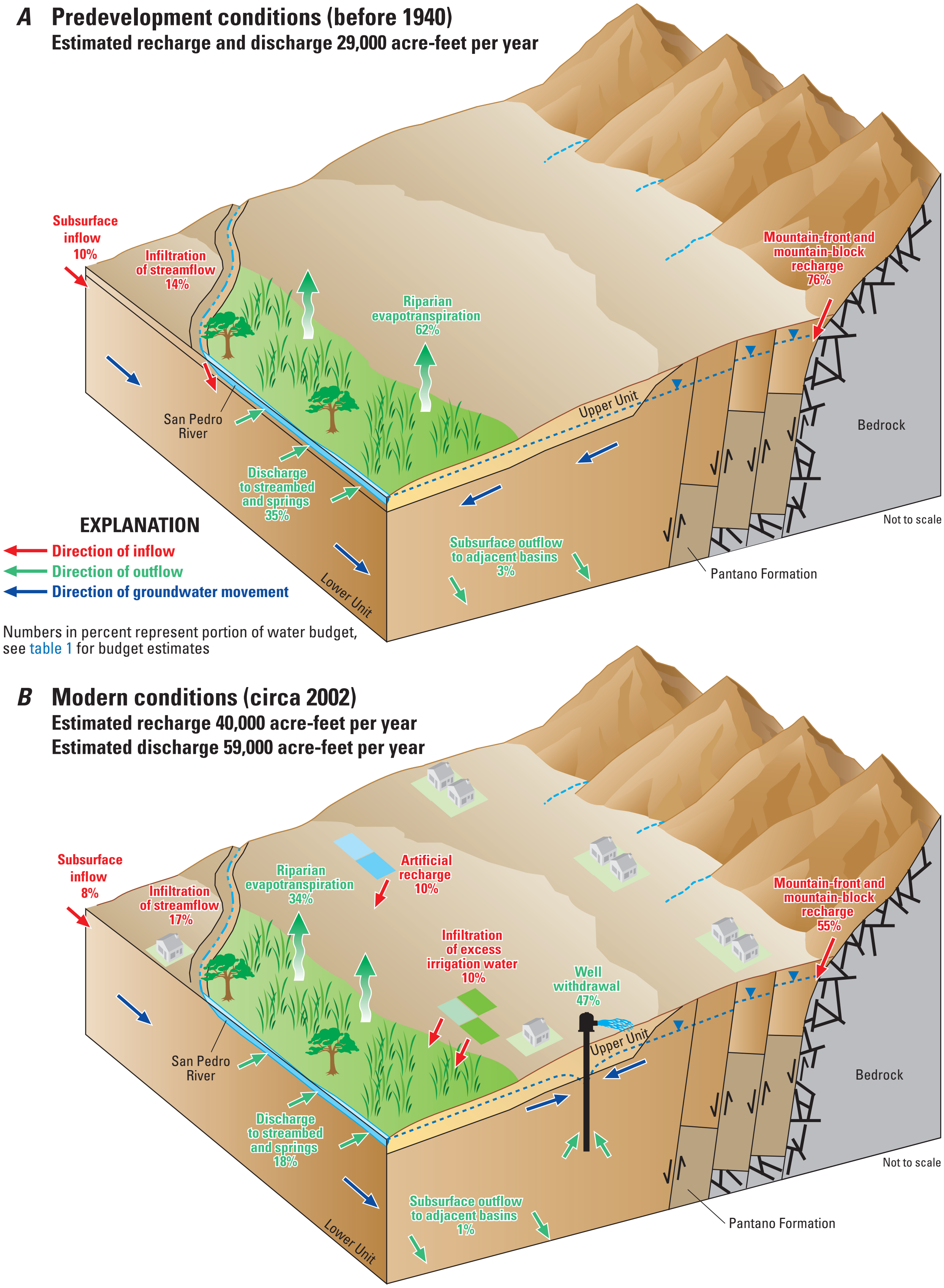


Figure 3. Generalized diagrams for the Sierra Vista subbasin, Arizona, showing components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 1. Estimated groundwater budget for the basin-fill aquifer in the Sierra Vista subbasin, Arizona, under predevelopment and modern conditions.

[All values are in acre-feet per year (acre-ft/yr) and are rounded to the nearest thousand. Estimates of groundwater recharge and discharge under predevelopment and modern conditions were derived from the footnoted sources. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components. Percentages for each water budget component are shown in figure 3. <, less than]

Budget component	Predevelopment conditions (before 1940)	Modern conditions (2002)	Change from predevelopment to modern conditions
	Estimated recharge		
Subsurface inflow	[1]3,000	[1]3,000	0
Mountain-front and mountain-block recharge	[1]22,000	[1]22,000	0
Infiltration of precipitation on basin	0	0	0
Infiltration of streamflow	[2]4,000	[2]7,000	3,000
Infiltration of excess irrigation water	0	[3]4,000	4,000
Artificial recharge	0	[1,4]4,000	4,000
Total recharge	29,000	40,000	11,000
	Estimated discharge		
Subsurface outflow to adjacent basins	[1]<1,000	[1]<1,000	0
Evapotranspiration	[1,5]18,000	[1,6]20,000	2,000
Discharge to streams	[7]10,000	[7]10,000	0
Discharge to springs	[8]<1,000	[8]<1,000	0
Well withdrawals	0	[1]28,000	28,000
Total discharge	29,000	59,000	30,000
Estimated change in storage	0	-19,000	-19,000

[1]From Arizona Department of Water Resources (2005a).

[2]Includes flood recharge in San Pedro River from Coes and Pool (2005). Modern conditions includes additional 3,000 acre-ft/yr because of increased runoff from urban areas.

[3]Based on net pumpage from Arizona Department of Water Resources (2005a) and an assumption of a 34 percent loss to groundwater system.

[4]Artifical recharge from municipal effluent recharge facilities, turf facility, and septic tank return flows.

[5]Combined value of Sierra Vista subwatershed evapotranspiration from Pool and Dickinson (2007) and Benson subwatershed evapotranspiration from Arizona Department of Water Resources (2005b).

[6]From Leenhouts and others (2006).

[7]Computed as residual of total discharge reported by Anderson and Freethey (1995) minus other discharge terms listed here.

[8]From Pool and Dickinson (2007); value for Sierra Vista subwatershed only.

The flux of water through the basin-fill aquifer of the Sierra Vista subbasin under predevelopment conditions has been estimated on the basis of available streamflow records and groundwater flow model calibration, and most of these efforts have focused on the Sierra Vista subwatershed (Freethey, 1982; Vionnet and Maddock, 1992; Anderson and Freethey, 1995; Corell and others, 1996; Thomas and Pool, 2006; Pool and Dickinson, 2007). One effort has been completed that utilized geochemical tracers to quantify recharge from the Huachuca Mountains (Wahi, 2005). Dickinson and others (2004) used inverse analysis of time-varying groundwater levels to infer recharge from the Huachuca Mountains. Generally, it is assumed that the natural portion of recharge is unchanged since predevelopment times, although work by Pool (2005) has related temporal variations in climate to changes in recharge.

Groundwater fluxes for the basin-fill aquifer of the Sierra Vista subbasin is estimated at 29,000 acre-ft annually for predevelopment conditions (fig. 3; table 1). Mountain-front and mountain-block recharge are the largest inflows to the aquifer, about 22,000 acre-ft/yr (Arizona Department of Water Resources, 2005a). Other sources of recharge include about 3,000 acre-ft/yr subsurface inflow to the aquifer along the United States-Mexico international boundary (Arizona Department of Water Resources, 2005a) and about 4,000 acre-ft/yr stream loss (Coes and Pool, 2005).

Evapotranspiration is the major pathway of groundwater discharge from the basin-fill aquifer and has been studied in detail for modern conditions in the Sierra Vista subwatershed (Leenhouts and others, 2006) and in much less detail for modern and predevelopment conditions in the Benson subwatershed. Pool and Dickinson (2007) calculated evapotranspiration from groundwater for the Sierra Vista subwatershed and the portion of the Upper San Pedro River Basin in Mexico as about 8,000 acre-ft/yr for predevelopment conditions. Arizona Department of Water Resources (2005b) estimated evapotranspiration from groundwater in the Benson subwatershed for modern times as about 10,000 acre-ft/yr, but did not make a predevelopment estimate. Assuming predevelopment and modern values are the same for the Benson subwatershed, total predevelopment evapotranspiration from the basin-fill aquifer was about 18,000 acre-ft/yr. Pool and Dickinson (2007) estimated less than 1,000 acre-ft/yr of discharge from the basin-fill aquifer as springflow, and Arizona Department of Water Resources (2005a) also estimated subsurface underflow out of the basin to be less than 1,000 acre-ft/yr. Given that the total groundwater recharge is estimated as 29,000 acre-ft/yr, stream gain in the Sierra Vista subbasin computed as a residual of the groundwater budget is about 10,000 acre-ft/yr. This estimate is confirmed by steady-state modeling performed separately for the Sierra Vista and Benson subwatersheds (Anderson and Freethey, 1995) from which an estimate of about 12,000 acre-ft/yr of net stream gain was simulated.

The act of developing a groundwater system changes an assumed initial steady-state condition into a transient condition. As a result, the development conditions are a function of the time period of interest. For this discussion, 2002 is taken to represent the modern condition.

Total groundwater recharge increased about 38 percent, from about 29,000 to 40,000 acre-ft/yr, as a result of water-resources development (table 1). Subsurface inflow, natural recharge through the mountain fronts, mountain blocks, and ephemeral stream channels is assumed to be invariant through time. An estimated 3,000 acre-ft/yr of recharge, however, was added to stream losses for modern conditions as a result of increases in the number and extent of impermeable surfaces in urban areas that generate more runoff that subsequently infiltrates the channel beds of ephemeral washes. Recharge under modern conditions was increased by about 8,000 acre-ft/yr as a result of artificial recharge facilities and incidental recharge from irrigated agricultural and urban lands and from septic tanks.

Total discharge nearly doubled, from about 29,000 to 59,000 acre-ft/yr, as a result of water-resources development (table 1). This increase was caused largely by groundwater pumping, which as discussed previously was nearly 28,000 acre-ft in 2002 (Arizona Department of Water Resources, 2005b). Estimated evapotranspiration increased by about 2,000 acre-ft/yr from predevelopment to modern conditions, but this increase may be an artifact of the different techniques used for the estimates provided by the different data sources. A quantitative evaluation of evapotranspiration changes through time has not been completed.

Groundwater Movement

The general pattern of groundwater movement under predevelopment conditions is of flow from the bounding mountains toward the San Pedro River and downgradient with the river (fig. 4*A*; Freethey and Anderson, 1986). This pattern of movement is about the same under modern conditions (fig. 4*B*; Arizona Department of Water Resources, 2005b). Differences between the location of the 100-ft water-level altitude contours from 1900 and 2001, shown in figures 4*A* and 4*B*, are more likely a result of differences in the interpretation of data by different studies rather than actual changes in water levels. Anderson and others (1992) indicate that water levels have not declined more than 50 ft from predevelopment conditions through 1980.

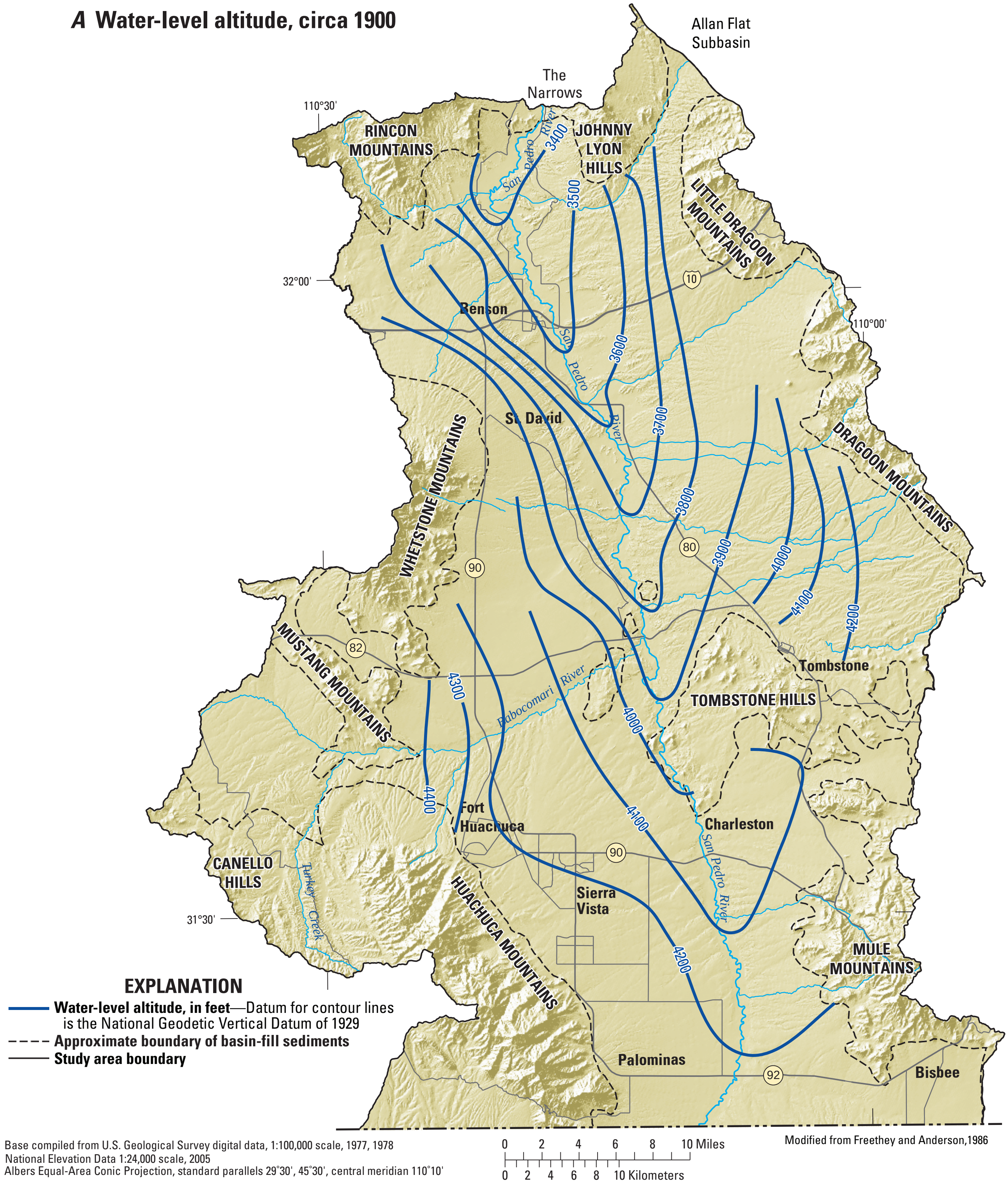


Figure 4. Water levels in the basin-fill aquifer of the Sierra Vista subbasin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 2001. (*C*) Water-level change for 1990–2001. (*D*) Depth to water, 2001.

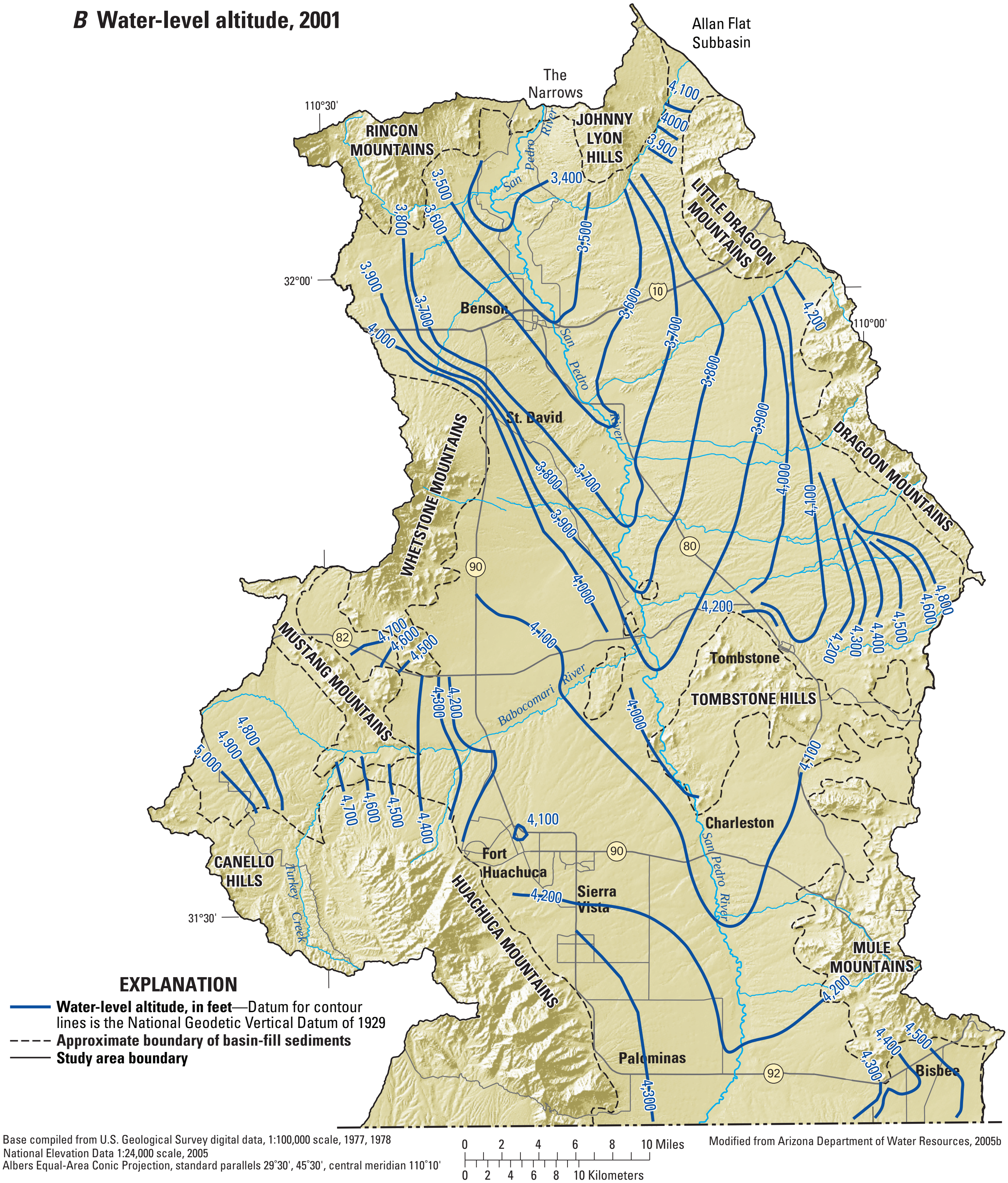


Figure 4. Water levels in the basin-fill aquifer of the Sierra Vista subbasin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 2001. (*C*) Water-level change for 1990–2001. (*D*) Depth to water, 2001—Continued.

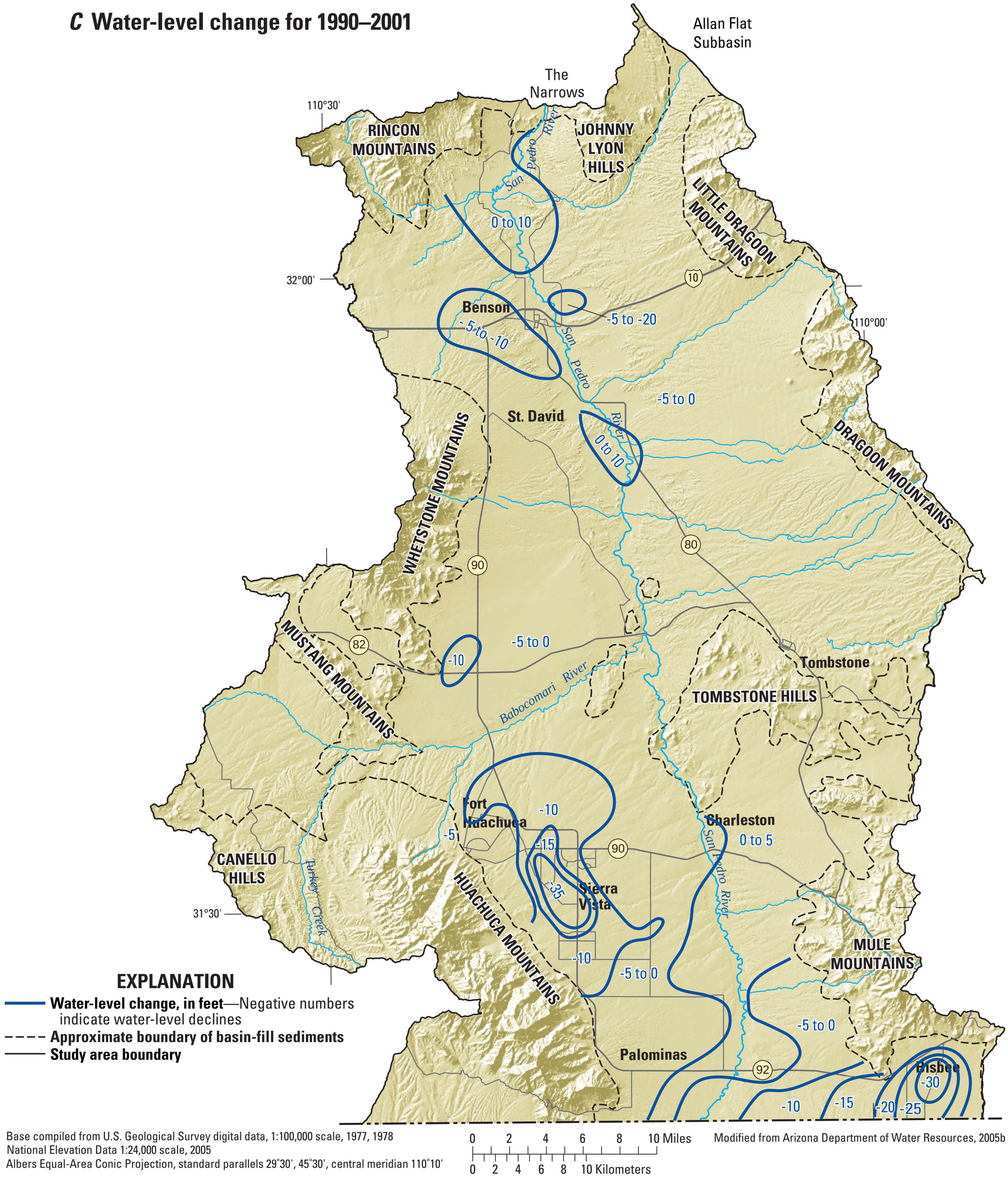


Figure 4. Water levels in the basin-fill aquifer of the Sierra Vista subbasin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 2001. (*C*) Water-level change for 1990–2001. (*D*) Depth to water, 2001—Continued.

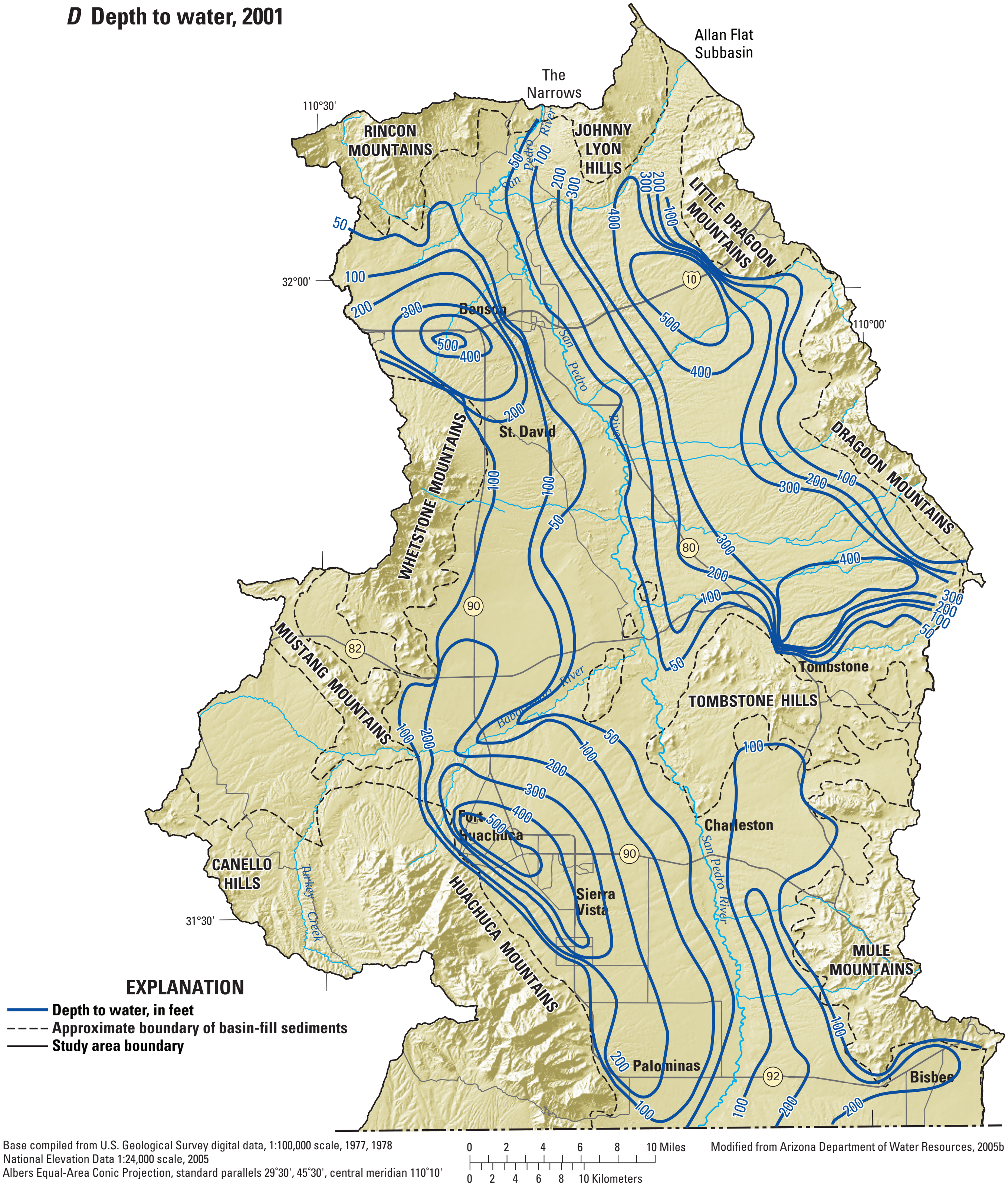


Figure 4. Water levels in the basin-fill aquifer of the Sierra Vista subbasin, Arizona. (*A*) Water-level altitude, circa 1900. (*B*) Water-level altitude, 2001. (*C*) Water-level change for 1990–2001. (*D*) Depth to water, 2001—Continued.

Locally, however, groundwater withdrawals have resulted in depressed water levels near Benson and Sierra Vista and in the extreme southeastern part of the subbasin. These locales are detectable through comparison of water-level measurements made in a common set of wells for 1990 and 2001 (fig. 4*C*; Arizona Department of Water Resources, 2005b). In a few areas of the basin along the San Pedro River, however, water levels rose during the period 1990–2000 (fig. 4*C*; Arizona Department of Water Resources, 2005b). Another change to the groundwater system besides water-level changes is attributed to the effects of irrigation, specifically, the process of recharging excess water from the surface while pumping from wells completed deep within the aquifer serves to redistribute water from deeper to shallower zones in the aquifer system.

Areas with shallow depths to water can be more susceptible to contamination than areas with deeper water levels, especially where vertical gradients are not upward in the aquifer and in unconfined areas. Depths to water typically are less than 100 ft along the basin-fill margins, increase to several hundred feet toward the center of the basin, and then decrease to less than 100 ft along the basin axis (figs. 2 and *4D*; Arizona Department of Water Resources 2005b). Areas of confined groundwater with upward gradients occur along the San Pedro River in the Palominas-Hereford area, and also in the St. David-Pomerene area (fig. 1).

Effects of Natural and Human Factors on Groundwater Quality

The quality of water in the basin-fill aquifer system in the Sierra Vista subbasin, which is affected by both natural and human-related factors, was cooperatively investigated by the U.S. Geological Survey's (USGS) National Water Quality Assessment (NAWQA) Program and the Arizona Department of Environmental Quality (ADEQ) in a 1996–97 study that sampled 39 wells. The results of this study were published in several reports. Coes and others (1999) provided a basin-wide assessment of general chemical parameters, major ions, nutrients, and trace constituents relative to location, depth, land use, and geology. Gellenbeck and Anning (2002) build on information from Coes and others (1999) by providing an assessment of pesticides and volatile organic compounds (VOCs) in the Sierra Vista subbasin and other areas. Cordy and others (2000) synthesized water-quality data from the Sierra Vista Subbasin, the West Salt River Valley, and the Upper Santa Cruz Basin. The following summary draws from these reports.

For the purposes of generalizing water-quality information, the Sierra Vista subbasin may be divided into four quadrants, with the east-west dividing line running roughly along the Sierra Vista and Benson subwatershed delineation (fig. 1) and the north-south line following the San Pedro River. A total of 39 wells in the subbasin were sampled during 1996–97, with each quadrant represented. Twenty of the wells were completed in unconfined basin-fill aquifer, 5 in confined basin-fill aquifer, 13 in water-bearing bedrock, and 1 in both water-bearing bedrock and unconfined basin-fill aquifer. Nineteen wells were sampled by the USGS, and 20 were sampled by the ADEQ. Coes and others (1999) determined that the datasets from the two agencies were comparable on the basis of replicate samples, and used both datasets in their analysis (fig. 5). Samples collected by the two agencies were analyzed for major ions, nutrients, and trace elements; those sampled by the USGS were additionally analyzed for tritium, pesticides, and VOCs. Groundwater in the Sierra Vista subbasin is, in most locations, suitable for all general human uses; relatively few sites had water in which any U.S. Environmental Protection Agency (USEPA) specific primary and secondary maximum contaminant levels (MCLs) were exceeded (fig. 5; U.S. Environmental Protection Agency, 2009).

Groundwater in the Sierra Vista subbasin is predominantly a calcium bicarbonate type, and generally is alkaline and of low salinity. Water in samples from 38 of the 39 wells had a pH above 7.0, and the median value was 7.4 standard pH units (table 2). Dissolved-solids concentrations are generally low—the median concentration in all wells was 262 mg/L, and the concentrations in samples from only two wells exceeded the USEPA secondary drinking-water standard of 500 mg/L (fig. 5). Sulfate concentrations in samples from these same two wells also exceeded the USEPA secondary drinking-water standard of 250 mg/L for sulfate.

Major-ion chemistry of the well samples was spatially correlated by quadrants (Coes and others, 1999). Sodium concentrations were higher in the northern half of the study area and were highest in the northeastern quadrant. Potassium concentrations were also generally higher in the northeastern quadrant than in other areas. Sodium concentration was related to aquifer type; concentrations were lower in unconfined areas of the aquifer than in the confined part in the St. David-Pomerene area. Sodium was also more concentrated in water-bearing bedrock units. Chloride concentrations were higher in water-bearing bedrock than in the basin-fill units. The spatial distribution of these ion concentrations are likely related to the varied mineralogy of the rocks in the mountains surrounding the Sierra Vista subbasin. Specifically, the higher concentrations of sodium and potassium in the northeastern quadrant are likely controlled by sodium- and potassium-bearing intrusive rocks of the Dragoon and Little Dragoon Mountains. Similarly, high concentrations of sulfate in the northwestern quadrant are likely related to deposits of gypsum interbedded with siltstone and dolomite in the Whetstone Mountains.

Figure 5. Elevated concentrations and detections of selected compounds in groundwater samples from the Sierra Vista subbasin, Arizona, 1996–97.

Table 2. Summary of groundwater-quality data, Sierra Vista subbasin, Arizona, 1996–97.

[Constituents are dissolved. N/A, not applicable; mg/L, milligrams per liter; µg/L, micrograms per liter; MRL, minimum reporting level. Data from Coes and others (1999) and Gellenbeck and Anning (2002)]

	Number of wells	Minimum reporting level		Percentiles				
		Highest	Lowest	10th	25th	50th (median)	75th	90th
pH (standard units)	39	N/A	N/A	7.2	7.3	7.4	7.6	7.9
Dissolved oxygen (mg/L)	19	N/A	N/A	1.7	3.5	5.0	6.1	6.5
Dissolved solids (mg/L)	39	10	1	174	222	262	316	419
Nitrate plus nitrite (mg/L as nitrogen)[1]	38	0.1	0.05	[2]0.02	0.47	0.78	1.40	3.9
Nitrogen ammonia (mg/L as nitrogen)[3]	38	0.1	0.015	[2]0.001	[2]0.003	[2]0.014	[2]0.030	[2]0.053
Arsenic (µg/L)[1]	39	10	1	[2]0.19	[2]0.48	[2]1.3	[2]3.5	[2]8.5
Barium (µg/L)[3]	39	100	1	[2]26	[2]36	[2]99	240	450
Fluoride (mg/L)[3]	39	0.2	0.1	0.2	0.2	0.5	1.0	2.7
Uranium (µg/L)	19	1	1	1.0	1.0	2.5	5.3	6.4

Notes on other constituents:

Trace constituents—More than 80 percent of the 39 wells were reported below the highest and lowest MRLs (in parentheses) for phosphorus (0.1 and 0.01 mg/L), iron (100 and 1 µg/L), lead (5 and 1 µg/L), manganese (50 and 1 µg/L), and selenium (5 and 1 µg/L). More than 100 percent of the 39 wells were reported below the highest and lowest MRLs (in parentheses) for antimony (5 and 1 µg/L), beryllium (1 and 0.5 µg/L), cadmium (1 and 1 µg/L), and silver (1 and 1 µg/L).

Pesticides—Of the 19 wells analyzed for 47 pesticide compounds, there were no pesticide detections.

Volatile organic compounds—Of the 19 wells analyzed for 87 compounds, there were 11 compounds detected amongst 14 wells. Compounds included 1,2,4 trimethylbenzene (10 wells), tetrachloroethylene (3 wells), chloromethane, dichlorodifluoromethane, and carbon disulfide (2 wells each), and bromodichloromethane, tribromomethane, benzene, chlorobenzene, acetone, and tetrahydrofuran (1 well each).

[1]Summary statistics calculated using maximum likelihood estimation method (Cohen, 1959).

[2]Values are extrapolated between the two minimum reporting levels.

[3]Summary statistics calculated using probability regression method (Cohen, 1959).

Of the 38 wells with nutrient analyses, concentrations were above the minimum reporting level (MRL) in 36 wells for nitrate plus nitrite (0.05 mg/L), but in only 11 wells for ammonia (0.015 mg/L), in 2 wells for phosphorus (0.01 mg/L) and in no wells for nitrite (0.010 mg/L). The low nitrite concentrations are likely a result of well oxygenated waters—all but one well had a dissolved-oxygen concentration greater than 1.0 mg/L. The median nitrate plus nitrite concentration was 0.78 mg/L, and 90 percent of the wells had concentrations less than 3.9 mg/L (table 2). The USEPA primary drinking-water standard for nitrate of 10 mg/L was not exceeded in any well. Statistical relations (using the Kruskall-Wallis test statistic) were found between the concentration of nitrate plus nitrite and well location. Specifically, concentrations were significantly higher in the southeastern quadrant than in the northeast quadrant. Both quadrants host minimal agricultural activity, and adequate data are not available to relate concentrations to sources.

With the exceptions of fluoride, arsenic, barium, and uranium, trace constituents were detected in filtered samples from fewer than 8 (20.5 percent) of the 39 wells (table 2). The median fluoride concentration was 0.5 mg/L, the USEPA secondary drinking-water standard for fluoride (2 mg/L) was exceeded in 7 wells (fig. 5). In one well, the concentration also exceeded the USEPA primary drinking-water standard for fluoride (4 mg/L). Fluoride concentrations were found to be higher in the northeastern quadrant than other parts of the subbasin and also higher in the confined parts of the St. David-Pomerene area (fig 5). Coes and others (1999) hypothesized that the cause of the higher concentrations was fluoride-bearing minerals in the Pinal Schist of the Dragoon, Little Dragoon, and Whetstone Mountains.

The median arsenic concentration is 1.3 µg/L, and the USEPA drinking-water standard for arsenic (10 µg/L) was exceeded in samples from 4 wells (fig. 5; Coes and others, 1999). For groundwater in southern Arizona, Robertson (1991) found a plausible arsenic source to be minerals in the basin-fill deposits that originated from hydrothermal sulfide and arsenide deposits in the surrounding mountains.

The median uranium concentration in samples was 2.5 μg/L; however, no samples exceeded the USEPA drinking-water standard for uranium of 30 μg/L (fig. 5; Tadayon and others, 1998). Manganese and iron concentrations were below the detection limit for most samples; however, concentrations in one sample from a well completed in bedrock exceeded the USEPA secondary drinking-water standard for these two constituents (fig. 5).

A comparison of major ion and trace constituent data for historical (1950–65) and 1996–97 conditions using the Wilcoxon rank-sum test found that no significant changes occurred between these periods in spite of a large increase in human population. This finding suggests that although exceptions occur locally, human activities have not had a widespread effect on groundwater chemistry in the subbasin.

Groundwater and surface waters in the Sierra Vista subbasin were sampled for analysis of pesticides and VOCs. Within the NAWQA studied basins in Arizona, the Sierra Vista subbasin represents a minimally developed basin as compared with other investigated areas (Gellenbeck and Anning, 2002). Consistent with this development status, analyses for a suite of 47 pesticides in water sampled from 19 wells in the Sierra Vista subbasin resulted in zero detections (table 2). Likewise, there were no detections for 86 pesticides that were analyzed in surface-water samples collected from the San Pedro River at the U.S. Geological Survey's stream-gaging station at Charleston (station number 09471000).

Detections of VOCs, however, belied the minimally developed designation of the basin. Eleven of 87 VOCs analyzed were detected in 14 (74 percent) of 19 groundwater samples (table 2; Gellenbeck and Anning, 2002). No VOC concentrations exceeded standards for those compounds established by the USEPA. The 14 samples in which VOCs were detected were from wells distributed about the subbasin (fig. 5) in areas of both urban land use and rangeland, suggesting anthropogenic impacts under a variety of land-use patterns. Detected compounds include:

1,2,4 trimethylbenzene (10 samples)	Tribromomethane (1 sample)
Tetrachloroethylene (PCE; 3 samples)	Benzene (1 sample)
Chloromethane (2 samples)	Chlorobenzene (1 sample)
Dichlorodifluoromethane (CFC-12; 2 samples)	Acetone (1 sample)
Carbon disulfide (2 samples)	Tetrahydrofuran (1 sample)
Bromodichloromethane (1 sample)	

Specific natural or human sources for the VOCs detected could not be identified; however, 1,2,4-trimethylbenzene is used in dyes and perfumes, as well as in trimetallic anhydride production. Tetrachloroethylene (PCE) and dichlorodifluoromethane may have been present in one well because of its location near a land fill. Two of the VOCs detected may have originated from natural sources—chloromethane and carbon disulfide can enter groundwater from fungi and, less likely for the Sierra Vista subbasin, from volcanic gases.

Tritium was detected in samples from 9 (47 percent) of 19 wells (fig. 5; Tadayon and others, 1998), although this occurrence was not discussed by Coes and others (1999) or by Gellenbeck and Anning (2002). The presence of tritium indicates a post-1953 recharge source for at least some component of the groundwater in those wells (See Section 1 of this report for a discussion of groundwater age and environmental tracers). Of the nine wells with tritium detections, seven also had one or more VOC detections (fig. 5). This indicates that areas with recent groundwater recharge can be expected to have a higher susceptibility to contamination. Other studies that have performed radioisotope dating (Wahi, 2005) found that the youngest waters were generally near the mountain fronts and that the ages increased toward the San Pedro River; this pattern fits the conceptual model of flow through the basin-fill aquifer. Wahi (2005) found uncorrected data indicated ages of greater than 18,000 radiocarbon years in deep wells near the basin center. In spite of these age patterns, the presence of VOCs in wells near the basin center suggests the presence of alternative, shorter transport pathways of anthropogenic chemicals to well screens that make the older groundwater vulnerable to contamination.

Summary

The Sierra Vista subbasin hosts a growing population as well as a substantial riparian ecosystem along the San Pedro River in southeastern Arizona. Groundwater is the primary source of water for the residents of the basin and it also sustains the baseflow and riparian ecosystem of the San Pedro River.

The subbasin has typical basin and range physiography and geology, and was formed in the grabens between block faulted mountain ranges. Miocene through early Pleistocene sediments eroded from the uplifted blocks and filled the basin. The basin fill includes the Pantano Formation, upper and lower basin-fill sediments, terrace deposits, and stream alluvium. The upper and lower basin-fill sediments hold the principal aquifer in the basin.

Under natural predevelopment conditions, the basin-fill aquifer in the Sierra Vista subbasin was primarily recharged along the mountain fronts, and groundwater flow was toward the San Pedro River where it discharged through the streambed or was removed through evapotranspiration.

Groundwater recharge and discharge increased as a result of water-resources development. Recharge increased primarily through greater stream losses, the infiltration of excess irrigation water, and artificial recharge. Discharge, which has nearly doubled in magnitude, increased primarily from groundwater pumping.

The general pattern of groundwater movement for predevelopment conditions is of flow from the bounding mountains toward the San Pedro River and downgradient with the river. While the general pattern of movement for modern conditions is about the same, groundwater withdrawals have resulted in locally depressed water levels near Benson and Sierra Vista, and in the extreme southeastern part of the subbasin.

Groundwater quality of the subbasin was cooperatively investigated by the U.S. Geological Survey's National Water-Quality Assessment Program and the Arizona Department of Environmental Quality in a 1996–97 study that sampled 39 wells. Groundwater in the Sierra Vista subbasin is, in most locations, suitable for all general human uses. The relatively few exceedences of U.S. Environmental Protection Agency standards include those of the primary maximum contaminant level (MCL) for fluoride (1 sample) and the secondary MCLs for pH (2 samples), both sulfate and dissolved solids (2 samples), fluoride (7 samples), and both iron and manganese (1 sample).

Variation in concentrations of major ions and trace constituents were attributed mostly to natural factors rather than human-related factors (table 3). Specifically, the presence and concentrations of major ions such as sodium, potassium, and sulfate, and trace elements such as fluoride were correlated with the occurrence of certain geological materials. The absence of pesticides in the water was attributed to the small amount of crop production in the subbasin. The frequent occurrence of volatile organic compounds in tandem with tritium detections (14 of 19 wells), however, emphasizes the vulnerability of the aquifer to contamination from sources at the land surface, especially in parts of the subbasin where there are pathways for recent recharge to enter the groundwater system.

Table 3. Summary of documented effects of natural and human-related factors on groundwater quality in the Sierra Vista subbasin, Arizona.

Groundwater-quality effect	Cause/source	General location(s)	Reference(s)
	Primarily natural factors		
Spatial variation in major ion chemistry	Spatial variation in different geologic materials	Entire basin	Coes and others, 1999
Elevated concentrations of sodium and potassium	Mineralogy of volcanic and intrusive rocks in the Dragoon and Little Dragoon Mountains	Northeastern quadrant of subbasin	Coes and others, 1999
Elevated concentrations of sulfate	Gypsum deposits interbedded with siltstone and dolomite	Northwestern quadrant of subbasin	Coes and others, 1999
Elevated concentrations of nitrate	Unknown, but unlikely human causes	Southeastern and northeastern quadrants of subbasin	Coes and others, 1999
Elevated concentrations of fluoride	Fluoride bearing minerals in the Pinal Schist	Northeastern quadrant of subbasin and in confined parts of aquifer near St. David and Pomerene	Coes and others, 1999
Detections of chloromethane and carbon disulfide	Possibly from fungi or volcanic gasses	Not pervasive in any area	Coes and others, 1999
	Primarily human-related factors		
Lack of pesticide detections in well samples	Relatively small amount of crop production compared with other basins in the southwestern United States	Entire basin	Gellenbeck and Anning, 2002
Occurrence of volatile organic compounds	Urban or agricultural use of volatile organic compounds	Areas susceptible to contamination, especially those that receive modern (post-1950) focused recharge, such as that along streams and irrigated areas	This study

References Cited

Adams, D.K., and Comrie, A.C., 1997, The North American monsoon: Bulletin of the American Meteorological Society, v. 78, p. 2197–2213.

Anderson, T.W., and Freethey, G.W., 1995, Simulation of ground-water flow in alluvial basins in south-central Arizona and parts of adjacent states: U.S. Geological Survey Professional Paper 1406–D.

Anderson, T.W., Freethey, G.W., and Tucci, P., 1992, Geohydrology and water resources of alluvial basins in south-central Arizona and parts of adjacent states: U.S. Geological Survey Professional Paper 1406–B, 67 p., 3 plates, scale 1:250,000.

Arizona Department of Water Resources, 1991, Hydrographic survey report for the San Pedro River Watershed: Volume 1: General Assessment, 546 p.

Arizona Department of Water Resources, 2005a, Groundwater resources of the Upper San Pedro Basin, Arizona: Technical report to the Upper San Pedro Basin AMA report, 85 p.

Arizona Department of Water Resources, 2005b, Upper San Pedro Basin Active Management Area review report, 219 p.

Brown, S.G., Davidson, E.S., Kister, L.R., and Thomsen, B.W., 1966, Water resources of Fort Huachuca Military Reservation, southeastern Arizona: U.S. Geological Survey Water-Supply Paper 1819–D, p. D1–D57.

Bryan, K., Smith, G.E., and Waring, G.A., 1934, Groundwater supplies and irrigation in the San Pedro Valley, Arizona: U.S. Geological Survey Open-File Report, 167 p.

City of Sierra Vista, 2006, accessed October 24, 2006 at http://www.ci.sierra-vista.az.us/cms1//resources/PIO/FAQ/Community%20Profile.pdf.

Coes, A.L., Gellenbeck, D.J., and Towne, D.C., 1999, Ground-water quality in the Sierra Vista Subbasin, Arizona, 1996–97: U.S. Geological Survey Water-Resources Investigations Report 99–4056, 50 p.

Coes, A.L., and Pool, D.R., 2005, Ephemeral-stream channel and basin-floor infiltration and recharge in the Sierra Vista subwatershed of the Upper San Pedro Basin, southeastern Arizona: U.S. Geological Survey, Open-File Report 2005–1023, 84 p. Available at http://pubs.usgs.gov/of/2005/1023/.

Cohen, A.C., 1959, Simplified estimators for the normal distribution when samples are singularly censored or truncated: Technometrics, v. 1, no. 3. P. 217–237.

Cordy, G.E., Gellenbeck, D.J., Gebler, J.B., Anning, D.W., Coes, A.L., Edmonds, R.J., Rees, J.A.H., and Sanger, H.W., 2000, Water quality in the central Arizona basins, Arizona, 1995–98: U.S. Geological Survey Circular 1213, 38 p. Available at http://pubs.usgs.gov/circ/circ1213.

Corell, S.W., Corkhill, F., Lovvik, D., and Putnam, F., 1996, A groundwater flow model of the Sierra Vista subwatershed of the Upper San Pedro Basin—Southeastern Arizona, in Modeling report no. 10: Phoenix, Arizona Department of Water Resources, Hydrology Division, 107 p.

Dickinson, J.E., Hanson, R.T., Ferre, T.P.A., and Leake, S.A., 2004, Inferring time-varying recharge from inverse analysis of long-term water levels: Water Resources Research, v. 40, 15 p.

Drewes, H., 1996, Geologic maps of the Coronado National Forest, southeast Arizona and southwest New Mexico: U.S. Geological Survey Bulletin 2083–B, p. 17–41.

Freethey, G.W., 1982, Hydrologic analysis of the Upper San Pedro Basin from the Mexico-United States boundary to Fairbank, Arizona: U.S. Geological Survey Open-File Report 82–752, 64 p.

Freethey, G.W., and Anderson, T.W., 1986, Predevelopment hydrologic conditions in the alluvial basins of Arizona and adjacent parts of California and New Mexico: U.S. Geological Survey Hydrologic Investigations Atlas 664, 3 plates.

Gellenbeck, D.J., and Anning, D.W., 2002, Occurrence and distribution of pesticides and volatile organic compounds in ground water and surface water in central Arizona basins, 1996–98, and their relation to land use: U.S. Geological Survey Water-Resources Investigations Report 01–4144, 101 p. Available at http://az.water.usgs.gov/pubs/WRIR01-4144intro.html.

Gettings, M.E., and Houser, B.B., 2000, Depth to bedrock in the Upper San Pedro Valley, Cochise County, southeastern Arizona: U.S. Geological Survey Open-File Report 00–138. Available at http://pubs.usgs.gov/of/2000/of00-138/.

Goodrich, D.C., Chehbouni, A., and others, 2000, Semi-arid land-surface-atmosphere (SALSA) program special issue: Agricultural and Forest Meteorology, v. 105, nos. 1–3, p. 3–20.

Hayes, P.T., 1970, Mesozoic stratigraphy of the Mule and Huachuca Mountains, Arizona: U.S. Geological Survey Professional Paper 658–A, 28 p.

Haynes, C.V., Jr., 1987, Curry Draw, Cochise, Arizona: A late Quaternary stratigraphic record of Pleistocene extinction and paleo-Indian activities, *in* Geologic diversity of Arizona and its margins: Excursions to choice areas: Tucson, Arizona Bureau of Geology and Mineral Technology Special Paper 5, p. 23–28.

Heindl, L.A., 1952, Upper San Pedro Basin, Cochise County Arizona, in Halpenny, L.D., Ground water in the Gila River Basin and adjacent areas, Arizona—A summary: U.S. Geological Survey Open File Report no. 29, p. 69–86.

Hereford, R. 1993, Entrenchment and widening of the upper San Pedro River, Arizona—Special Paper 282: Boulder, Colorado, Geological Society of America, 42 p.

Hirschberg, D.M., and Pitts, S.G., 2000, Digital geologic map of Arizona: a digital database from the 1983 printing of the Wilson, Moore, and Cooper 1:500,000 scale map of Arizona: U.S. Geological Survey Open-File Report 2000–409. Available at http://geopubs.wr.usgs.gov/open-file/of00-409/.

Leenhouts, J. M., Stromberg, J.C., and Scott, R.L., eds., 2006, Hydrologic requirements of and consumptive ground-water use by riparian vegetation along the San Pedro River, Arizona: U.S. Geological Survey Scientific Investigations Report 2005–5163, 154 p. Available at http://pubs.usgs.gov/sir/2005/5163/.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

Pool, D.R., 2005, Variations in climate and ephemeral channel recharge in southeastern Arizona, United States: Water Resources Research, v. 41, 24 p.

Pool, D.R., and Coes, A.L., 1999, Hydrogeologic investigations of the Sierra Vista subwatershed of the Upper San Pedro Basin, Cochise County, southeast Arizona: U.S. Geological Survey Water-Resources Investigations Report 99–4197, 41 p. Available at http://az.water.usgs.gov/pubs/WRIR99-4197intro.html.

Pool, D.R., and Dickinson, J.E., 2007, Ground-water flow model of the Sierra Vista subwatershed and Sonoran portions of the Upper San Pedro Basin, southeastern Arizona, United States, and northern Sonora, Mexico: U.S. Geological Survey Scientific Investigations Report 2006–5228, 48 p. Available at http://pubs.usgs.gov/sir/2006/5228/.

Robertson, F.N., 1991, Geochemistry of ground water in alluvial basins of Arizona and adjacent parts of Nevada, New Mexico, and California: U.S. Geological Survey Professional Paper 1406–C, 90 p.

Roeske, R.H., and Werrell, W.H., 1973, Hydrologic conditions in the San Pedro River Valley, Arizona, 1971: Arizona Water Commission Bulletin 4, 76 p.

Rodgers, W.M., 1965, Historical land occupance of the upper San Pedro River since 1970: M.S. thesis, University of Arizona, Tucson, 167 p.

Scanlon, B.R., Langford, R.P., and Goldsmith, R.S., 1999, Relationship between geomorphic settings and unsaturated flow in an arid setting: Water Resources Research, v. 35, p. 983–999.

Scott, R.L., Williams, D.G., Goodrich, D.C., Cable, W.L., Levick, L.R., McGuire, R., Gazal, R.M., Yepez, E.A., Ellsworth, P., and Huxman, T.E., 2006, Determining the riparian ground-water use within the San Pedro Riparian National Conservation Area and the Sierra Vista subwatershed, Arizona, chapter D *of* Leenhouts, J. M., Stromberg, J.C., and Scott, R.L., eds., Hydrologic requirements of and consumptive ground-water use by riparian vegetation along the San Pedro River, Arizona: U.S. Geological Survey Scientific Investigations Report 2005–5163, 154 p. Available at http://pubs.usgs.gov/sir/2005/5163/.

Tadayon, S., Duet, N.R., Fisk, G.G., McCormack, H.F., Pope, G.L., and Rigas, P.D., 1998, Water resources data, Arizona, water year 1997: U.S. Geological Survey Water-Data Report AZ–97–1, 416 p.

Thomas, B.E., and Pool, D.R., 2006, Trends in streamflow of the San Pedro River, southeastern Arizona, and regional trends in precipitation and streamflow in southeastern Arizona and southwestern New Mexico: U.S. Geological Survey Professional Paper 1712, 79 p. Available at http://pubs.usgs.gov/pp/pp1712/.

Trischka, C., 1971, A history of Cochise County Arizona: Cochise Quarterly, v. 1, p. 3–21.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed March 3, 2009 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003b, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

Vionnet, L.B., and Maddock, T., 1992, Modeling of groundwater flow and surfacewater/groundwater interactions in the San Pedro River Basin—Part I—Canenea, Mexico to Fairbank, Arizona: Tucson, Ariz., University of Arizona, Department of Hydrology and Water Resources Technical Report HWR 92–010, 236 p.

Wahi, A.K., 2005, Quantifying mountain system recharge in the Upper San Pedro Basin, Arizona, using geochemical tracers: M.S. thesis, University of Arizona, Department of Hydrology and Water Resources, 122 p.

Section 10.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the San Luis Valley, Colorado and New Mexico

By Laura M. Bexfield and Scott K. Anderholm

Basin Overview

The San Luis Valley in southern Colorado and northern New Mexico (fig. 1) has extensive areas of irrigated agriculture overlying a shallow aquifer of relatively high intrinsic susceptibility to contamination that is used for public, domestic, and agricultural supply. Defined for the purposes of this study by the boundary of the basin-fill deposits within the San Luis surface-water basin (fig. 1), the San Luis Valley includes the internally drained San Luis closed basin at the far northern end of the area, along with the northernmost surface-water and alluvial basins drained by the Rio Grande, which enters the San Luis Valley from the west. By means of the Rio Grande, the San Luis Valley is hydraulically connected at its southern end to the Española Basin. Altitudes of the alluvial basins within the San Luis Valley, which cover about 4,900 mi^2, range from about 5,800 ft where the Rio Grande drains the valley at its southern end to nearly 8,900 ft along the margins of the San Juan Mountains on the west and the Sangre de Cristo Mountains on the east (fig. 1). This section will focus on that part of the San Luis Valley within Colorado and particularly on the Alamosa Basin (an alluvial basin that includes both the San Luis closed basin and areas drained by the Rio Grande), which contains most of the valley's agricultural area. The Alamosa Basin lies within the Southern Rocky Mountains Physiographic Province and is considered part of the Rio Grande aquifer system (Robson and Banta, 1995), but has hydrogeologic characteristics similar to those of alluvial basins in the Basin and Range aquifer system of the southwestern United States.

The San Luis Valley is categorized as having an arid to semiarid climate, characterized by abundant sunshine, low humidity, and a high rate of evaporation that substantially exceeds the generally low rate of precipitation. Mean annual precipitation for 1948–2006 was only 7.1 in. at Alamosa (Western Regional Climate Center, 2006a), although mean annual precipitation for 1957–2005 was 45.4 in. at Wolf Creek Pass in the San Juan Mountains, which border the basin to the west (Western Regional Climate Center, 2006b). Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 11.0 in. over the alluvial basin area of the San Luis Valley as a whole (McKinney and Anning, 2009). About 44 percent of precipitation within the alluvial basin falls between July and September; winter storms make a large contribution to annual precipitation in the surrounding mountains. Evapotranspiration from a class-A pan during April through October for years 1960 to 1980 at Alamosa averaged 57 in. (Leonard and Watts, 1989). The mean monthly maximum temperature for 1948–2006 at Alamosa was 37.4°F in January and 82.1°F in July (Western Regional Climate Center, 2006a).

In 2000, the total population of the six major counties that lie within the San Luis Valley was about 75,300 (U.S. Census Bureau, 2001a, 2001c), a 33 percent increase from the population of about 56,600 in 1980. The largest cities and towns in 2000 were Alamosa, Colorado; Taos, New Mexico; and Monte Vista, Colorado (U.S. Census Bureau, 2001b, 2001d). Analysis of LandScan population data for 2000 (Oak Ridge National Laboratory, 2005) indicated a population of about 70,200 for the alluvial basin area of the San Luis Valley as a whole (McKinney and Anning, 2009). Areas classified as urban cover less than one percent of the valley. The National Land Cover Database dataset for 2001 (U.S. Geological Survey, 2003) indicated that the dominant land-use types are rangeland, which makes up about 70 percent of the area, and agriculture, which makes up about 20 percent. Most agriculture is concentrated in the western part of the Alamosa Basin (fig. 1). The high rate of evapotranspiration relative to precipitation requires that crops be irrigated throughout the growing season. The most abundant crops grown in the San Luis Valley are alfalfa, native hay, barley, wheat, potatoes, and other vegetables, with rotation of barley or alfalfa and potatoes being common (Anderholm, 1996).

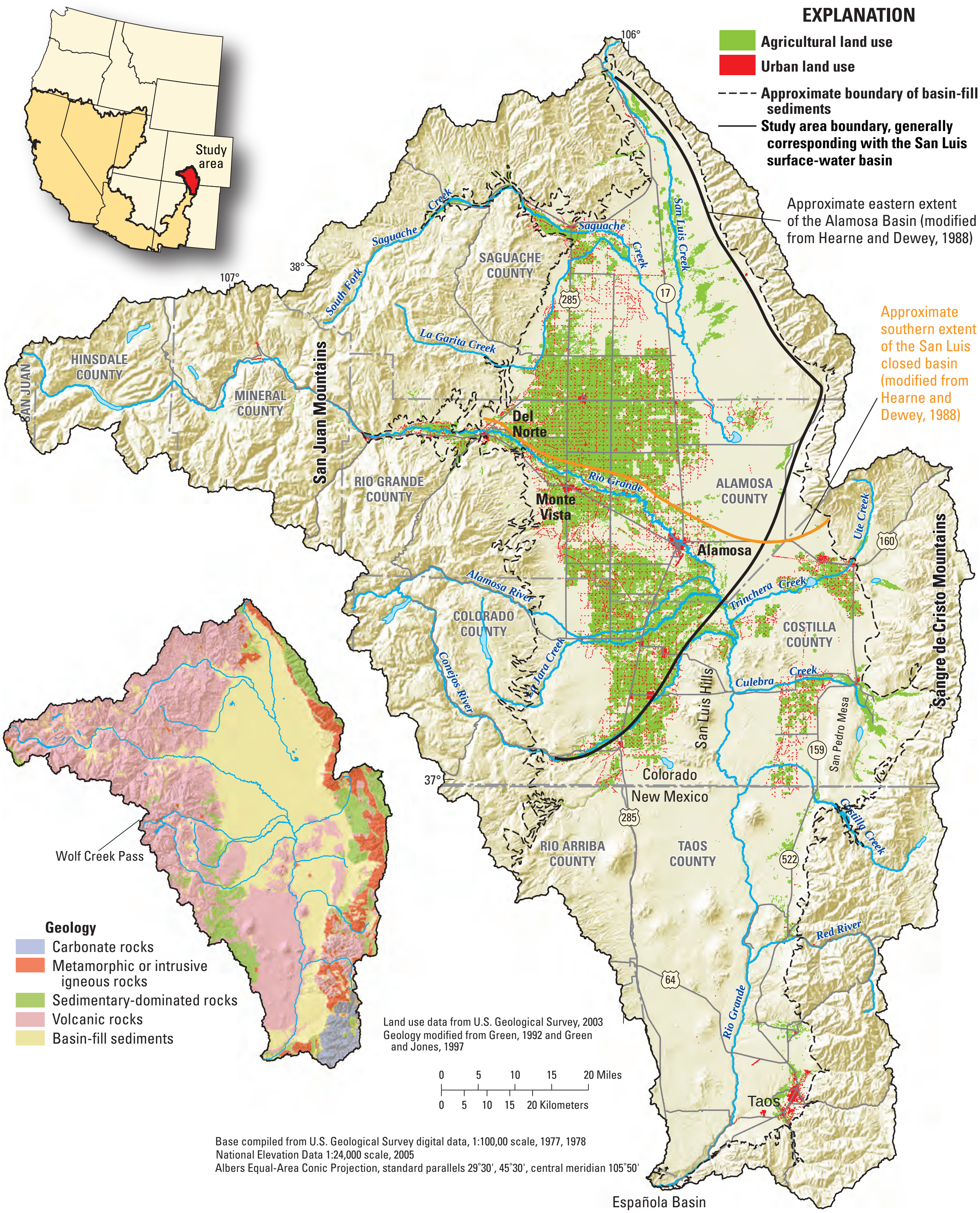


Figure 1. Physiography, land use, and generalized geology of the San Luis Valley, Colorado and New Mexico.

Irrigated agriculture is the largest water user within the San Luis Valley. Water-use estimates by the U.S. Geological Survey (USGS) for 2000 (http://water.usgs.gov/watuse/) indicate that water use for public supply was less than 1 percent of total use. About 54 percent of the water used for irrigated agriculture was surface water, which is diverted from the Rio Grande and smaller tributary streams. Most of the wells pumped for irrigation water are completed in the shallow unconfined aquifer, although the deeper confined aquifer also is commonly used as a source of irrigation water (Emery and others, 1969; Emery and others, 1971a, 1971b; Hearne and Dewey, 1988; Stogner, 2001). USGS estimates indicate that more than 90 percent of all water demand for public supply in the San Luis Valley in 2000 was met by groundwater withdrawals. Public-supply wells pump primarily from the confined aquifer (Emery and others, 1971b), whereas domestic wells commonly pump from the unconfined aquifer (Stogner, 2001). Development of water resources in the San Luis Valley for agricultural and urban purposes has substantially altered processes that recharge the groundwater system and has also affected groundwater movement and discharge.

Groundwater-quality issues that were identified for the San Luis Valley include both naturally occurring contaminants and anthropogenic compounds. As described later in this section, concentrations of dissolved solids are larger than the U.S. Environmental Protection Agency (USEPA) non-enforceable guideline of 500 mg/L (U.S. Environmental Protection Agency, 2009; each time a drinking-water standard or guideline is mentioned in this section, it denotes the citation "U.S. Environmental Protection Agency, 2009") and as large as 20,000 mg/L in parts of the unconfined aquifer; dissolved-solids concentrations also can exceed 500 mg/L in upper parts of the confined aquifer. Nitrate concentrations in shallow groundwater of the unconfined aquifer exceed the background concentration of 3 mg/L (Stogner, 2001)—and even the USEPA drinking-water standard of 10 mg/L—across large areas in the western part of the Alamosa Basin, likely as the result of the application of fertilizer to crops. Naturally elevated concentrations of uranium and(or) gross alpha activities have been detected in groundwater from shallow monitoring wells completed in the unconfined aquifer, as have elevated arsenic concentrations. With respect to anthropogenic compounds, pesticides (both agricultural and nonagricultural herbicides and insecticides) have been detected at generally low concentrations in shallow parts of the unconfined aquifer beneath agricultural areas.

Water Development History

Although irrigated agriculture has been practiced in the Alamosa Basin at least since the arrival of Spanish settlers in the 1630s, irrigated acreage remained small until the 1880s (Hearne and Dewey, 1998). From about 1880 to 1890, extensive networks of canals and irrigation structures were built to divert water from the Rio Grande and its tributaries, resulting soon afterward in the diversion of all available natural flow from these streams to irrigate agricultural fields, primarily in the central part of the Alamosa Basin (Powell, 1958; Hearne and Dewey, 1988; Stogner, 2001). Several reservoirs also were constructed on the Rio Grande, the Conejos River, and other tributaries starting in the early 1900s to help match water supplies to irrigation needs, typically by storing water during spring months and releasing it to canals late in the summer (Colorado Division of Water Resources, 2004). Before the 1970s, a common method of irrigation using surface water was subirrigation, whereby sufficient water was applied to raise the water table to the root zone of the growing crops, about 1 to 3 ft below land surface (Powell, 1958; Hearne and Dewey, 1988; Stogner, 2001). However, subirrigation soon resulted in waterlogging and alkali damage of soils, forcing a shift in irrigated agriculture to higher land to the west by about 1915 (Powell, 1958). Another consequence of irrigating the west side of the basin with surface water was substantial rise in water levels, estimated to be as great as 50 to 100 ft across areas of the Rio Grande alluvial fan (Powell, 1958). The groundwater divide that separates the San Luis closed basin from areas to the south might have developed as a result of irrigation on the Rio Grande alluvial fan, and the location of the divide probably migrates north and south partly in response to changes in irrigation-return flow in the area (Hearne and Dewey, 1988).

Groundwater also has been used to irrigate crops in the Alamosa Basin since at least 1904, when water from the confined aquifer was noted to be of economic importance for agriculture in several areas of the basin (Powell, 1958). At the time of his study, Powell (1958) had documented 586 flowing wells and 61 pumped wells that were completed in the confined aquifer for use in irrigation. Irrigation water reportedly has been pumped from the unconfined aquifer since 1903 (Powell, 1958). However, utilization of water from the unconfined aquifer did not increase markedly until severe droughts of the 1930s and 1950s forced farmers to supplement surface-water supplies (Stogner, 2001). The number of irrigation wells completed in the unconfined aquifer rose from 176 by 1936 to about 1,300 by 1952 and more than 2,300 by 1980 (Powell, 1958; Stogner, 2001). By the late 1960s, subirrigation was no longer effective because the increase in withdrawals from the unconfined aquifer had lowered the water table (Stogner, 2001). A dramatic increase in the use of center-pivot sprinkler systems for irrigation started in the 1970s; the number of such systems rose from 262 in 1973 to 1,541 in 1980 and almost 2,000 in 1990 (Hearne and Dewey, 1988; Stogner, 2001). These systems, which generally rotate overhead sprinklers around a point in the center of a 160-acre field, allow more precise application of water. They generally use groundwater pumped from the unconfined aquifer and have largely replaced flood irrigation in the southern part of the San Luis closed basin, where they are particularly common (Anderholm, 1996).

The amount of surface water used to irrigate crops in the Alamosa Basin varies from year to year depending on total irrigated acreage and climatic factors that affect crop water requirements and surface-water availability (fig. 2) (Wilson, 2004). Irrigated acreage in the Colorado part of the Rio Grande drainage generally increased between 1950 and 2002, averaging about 581,000 acres (Wilson, 2004). The average annual surface-water diversion for irrigation between 1950 and 2002 was about 1,077,000 acre-ft; the annual supply-limited consumptive use averaged about 395,000 acre-ft (Wilson, 2004). The use of groundwater to help meet irrigation requirements increased steadily between 1950 and 2002, with the average annual diversion of groundwater at about 543,000 acre-ft and the average annual consumptive use at about 365,000 acre-ft (Wilson, 2004). During periods when surface water supplied to individual farms exceeds crop demands, some irrigation districts and ditch companies in the area encourage diversion of surface water into recharge pits at the edges of agricultural fields, thereby helping to maintain water levels in the unconfined aquifer (Miller and others, 1993).

Most of the water used for drinking, domestic purposes, and stock needs in the Alamosa Basin is groundwater. Powell (1958) reported that the first wells used for these purposes were completed in the unconfined aquifer; however, only four years after confined conditions were discovered by accident in 1887, an estimated 2,000 flowing wells had been completed in the confined aquifer, mostly for domestic water uses. At the time of his study, Powell (1958) documented a total of six public-supply wells completed in the confined aquifer, producing about 720 million gallons annually from depths ranging from 365 ft to 1,802 ft below land surface. Emery and others (1971b) found records of 11 wells completed in the confined aquifer and four wells completed in the unconfined aquifer in use for public supply in the Colorado portion of the San Luis Valley. All of the larger cities and several of the smaller towns in the area now rely on public-supply wells (Colorado Division of Water Resources, 2004); at least 76 municipal supply wells have been permitted in the Colorado part of the San Luis Valley (Harmon, 2000). Using city and county population estimates combined with representative per capita use by month and a consumptive use factor of 0.4, Wilson (2000) estimated total consumptive water use in 1995 for combined municipal, domestic, commercial, and public purposes to be about 5,800 acre-ft, which equates to withdrawals of about 14,000 acre-ft. Harmon (2000) indicated that about 600 wells had been permitted for domestic or related uses in the Colorado part of the San Luis Valley and estimated that these wells pump about 530 acre-ft/yr.

Groundwater also is pumped from the San Luis Valley in association with the Closed Basin Project. This project pumps water from the unconfined aquifer in areas of natural groundwater discharge in the San Luis closed basin and delivers the water to the Rio Grande through a series of channels and pipelines. The Closed Basin Project is designed to “salvage” water that otherwise would have been lost to “nonbeneficial” evapotranspiration by lowering the water table (Leonard and Watts, 1989; Harmon, 2000). The salvaged water is then used to help meet Colorado’s obligations under the Rio Grande Compact of 1929. Pumping for the Closed Basin Project averaged about 22,560 acre-ft/yr between 1986 (the first year of operation) and 1997, when 168 wells were included in the project (Harmon, 2000).

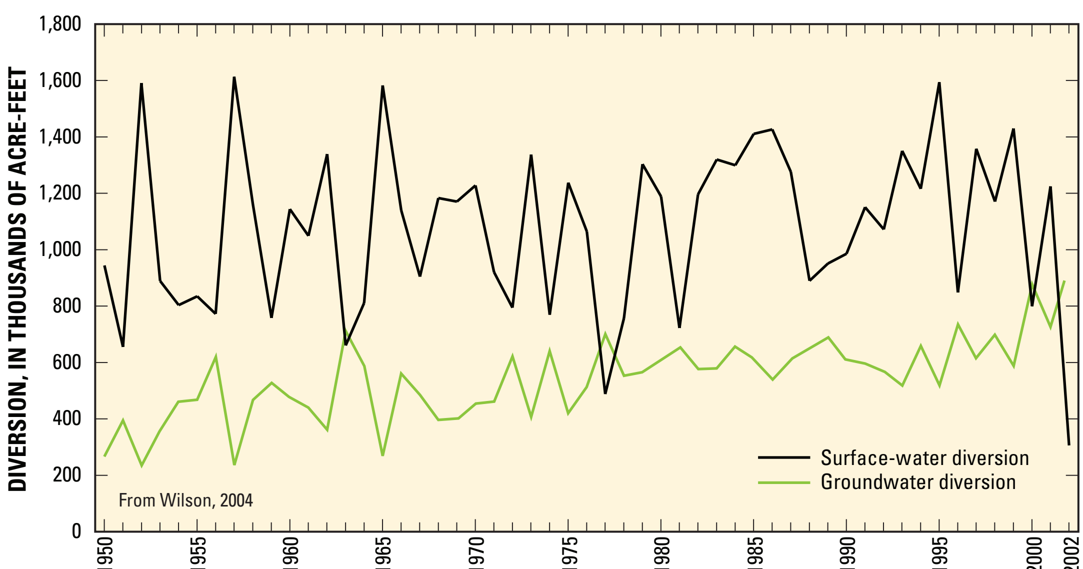


Figure 2. Estimated annual diversions of surface water and groundwater for irrigation in the Colorado part of the San Luis Valley, 1950 to 2002.

Hydrogeology

The San Luis Valley is a major physiographic and structural feature formed by crustal extension along the generally north-south trending Rio Grande Rift. The valley is downfaulted along the Sangre de Cristo Mountains that border the valley on the east and hinged along the San Juan Mountains that border the valley on the west (Emery and others, 1971a) (fig. 1). The Sangre de Cristo Mountains consist largely of Precambrian, Paleozoic, and Mesozoic igneous and metamorphic rocks, whereas the San Juan Mountains consist of a thick sequence of volcanic rocks that underlie and intertongue with sedimentary rocks within the San Luis Valley (Hearne and Dewey, 1988).

The Alamosa Basin at the north end of the San Luis Valley is divided into eastern and western subbasins, the Baca and Monte Vista grabens, respectively, by an uplifted fault block known as the Alamosa horst (fig. 3). Except where indicated, the following information on the valley-fill deposits of the Alamosa Basin is derived from the discussion by Leonard and Watts (1989), which incorporates the conclusions of several previous investigations. The maximum thickness of valley-fill deposits is about 10,000 ft in the western subbasin, about 5,400 ft over the Alamosa horst, and about 19,000 ft in the eastern subbasin. As mentioned previously, the Alamosa Basin includes the San Luis closed basin on the north. At the southern end, the Alamosa Basin is hydraulically separated from the Costilla Plains and the Taos Plateau by the San Luis Hills (Hearne and Dewey, 1988). Faults, which are common in the Alamosa Basin, might affect groundwater movement by acting as barriers to horizontal flow (Huntley, 1976) and(or) as conduits for vertical movement (Mayo and Webber, 1991).

The basin fill of the Alamosa Basin comprises alluvial sedimentary rocks and Tertiary volcanic rocks (fig. 3). The oldest sequence of alluvial sedimentary rocks is the Eocene to Oligocene deposits of the Vallejo Formation, which is present only in the western part of the basin and consists of reddish fluvial clay, silt, sand, and gravel. The Vallejo Formation is overlain by an eastward-thinning wedge of heterogeneous volcanic and volcaniclastic rocks of the Oligocene Conejos Formation (McCalpin, 1996; Mayo and others, 2007), which is in turn overlain by the Fish Canyon and Carpenter Ridge Tuffs of Oligocene age (Leonard and Watts, 1989; Mayo and others, 2007).

The basin-fill deposits of the Santa Fe and Los Pinos Formations, which range in age from Oligocene to Pliocene, are as thick as 10,000 ft in the eastern subbasin of the Alamosa Basin (McCalpin, 1996). The Los Pinos Formation forms an eastward thickening wedge along the eastern border of the San Juan Mountains, consisting of sandy gravel with interbedded volcaniclastic sandstone and tuffaceous material. The Santa Fe Formation, which is predominant in the eastern part of the Alamosa Basin and intertongues with the Los Pinos Formation, consists of buff to pinkish-orange clays with interbedded, poorly to moderately sorted silty sands.

The Alamosa Formation of Pliocene and Pleistocene age overlies the Santa Fe and(or) Los Pinos Formations across most of the Alamosa Basin. The Alamosa Formation, which is up to about 2,050 ft thick, consists of discontinuous beds of clay, silt, sand, and gravel of mixed fluvial, lacustrine, and eolian origin (Leonard and Watts, 1989; McCalpin, 1996); these deposits generally become more fine grained toward the topographic low of the San Luis closed basin. Within the Alamosa Formation, the position of the uppermost blue clay or fine-grained sand, the top of which is generally between 60 and 120 ft below land surface, typically is used to assign the division between the shallow, unconfined aquifer and the deeper, confined aquifer. Pleistocene and Holocene deposits that overlie the Alamosa Formation are similar in lithology and represent eolian reworking of valley floor deposits, alluvial fan deposition, and deposition in stream channels (McCalpin, 1996; Mayo and others, 2007).

Conceptual Understanding of the Groundwater System

The groundwater system of the San Luis Valley has been most thoroughly studied—and is most intensely utilized—in Colorado, and particularly in the Alamosa Basin. In general terms, the groundwater system of the Alamosa Basin includes two main aquifers—the shallow, unconfined aquifer that is present across the entire basin, and the deeper, confined aquifer that is present everywhere except along the basin margins. Although the division between the two aquifers usually is defined by the top of the uppermost blue clay or fine-grained sand in the Alamosa Formation, Hearne and Dewey (1988) emphasize that groundwater conditions in the basin are complex because the overall aquifer system is really a heterogeneous mixture of aquifers and leaky confining beds, each of limited areal extent. Two separate flow systems also are present in the basin—one in the San Luis closed basin and one in the part of the Alamosa Basin that is drained by the Rio Grande.

Depths to water are fairly small throughout the Alamosa Basin. In 1969, Emery and others (1973) measured depths to water of 12 ft or less throughout much of the basin, although depths to water exceeded 12 ft along the basin margins. During 1997–2001, depths to water in the intensively cultivated area north of the Rio Grande ranged from less than 5 ft to more than 25 ft (fig. 4). Depths to water throughout the basin respond to seasonal variations in recharge and discharge (Emery and others, 1973; Leonard and Watts, 1989; Stogner, 2005).

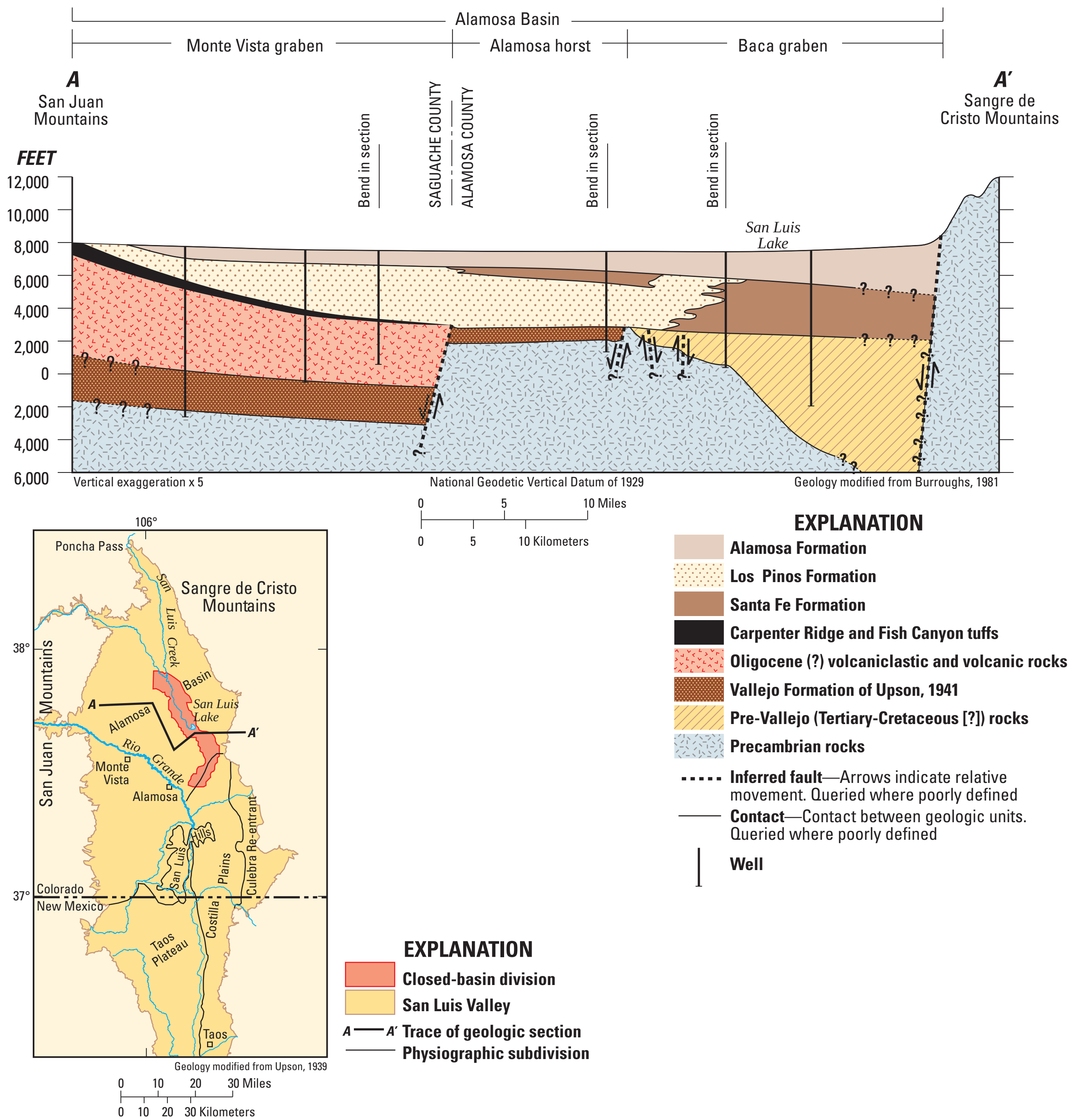


Figure 3. Generalized geologic section for the Alamosa Basin, Colorado.

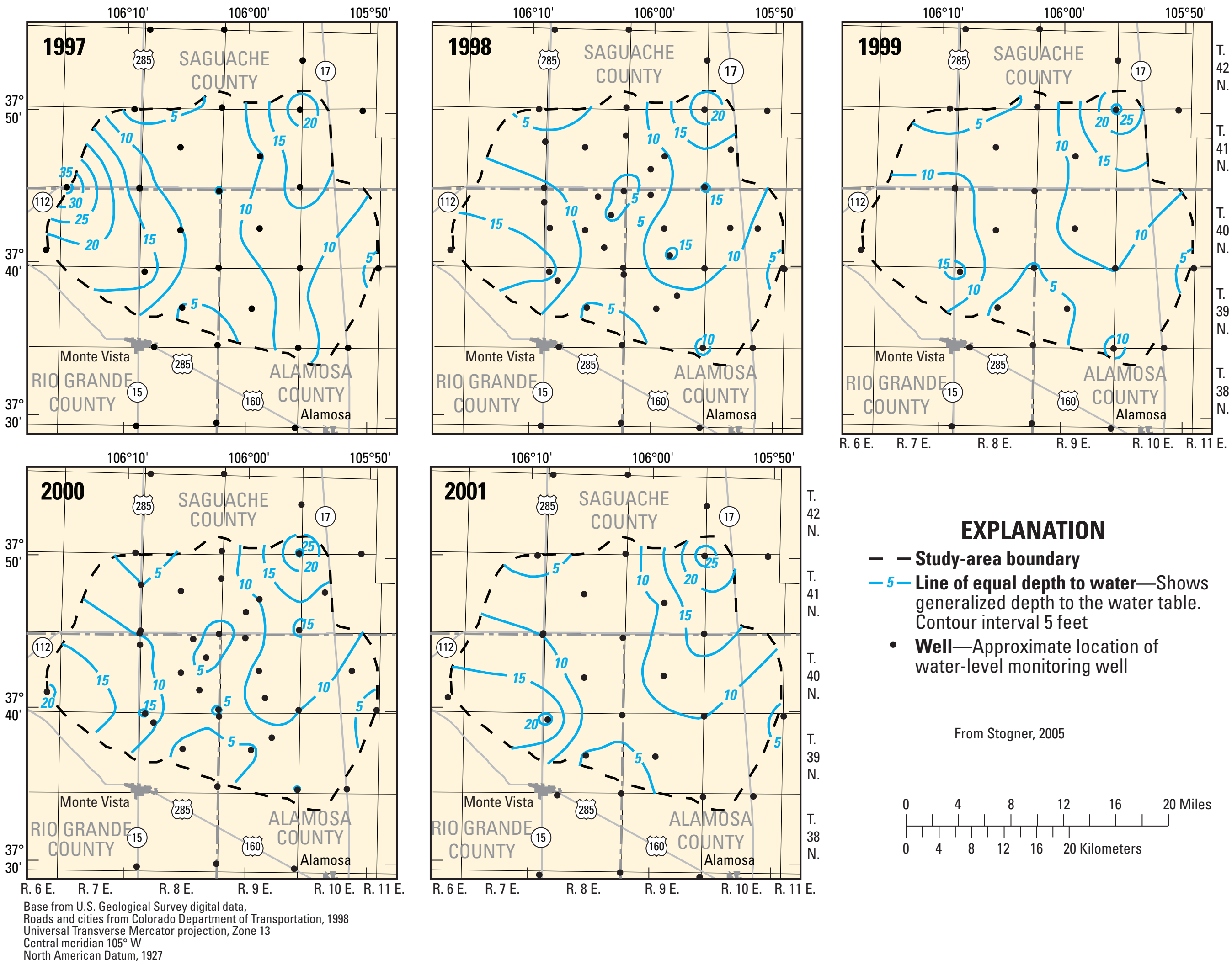


Figure 4. Depths to water in the unconfined aquifer in June 1997–2001 for part of the Alamosa Basin, Colorado.

On the basis of results of aquifer tests and specific-capacity tests, Emery and others (1973) estimated transmissivity values in the unconfined aquifer in most of the Alamosa Basin from about 700 ft^2/d near basin margins to about 30,200 ft^2/d in the western part of the basin, north of the Rio Grande; estimated transmissivity values in the confined aquifer generally ranged from about 13,400 to more than 160,800 ft^2/d. The Colorado Division of Water Resources (2004) groundwater-flow model (hereinafter, "the CDWR model") for the Colorado part of the San Luis Valley assigns individual values of horizontal hydraulic conductivity to each of four layers in 26 "parameter zones" and to a single fifth layer that represents the lower Santa Fe Formation. Horizontal hydraulic conductivity values range from 0.1 ft/d for the lower Santa Fe Formation (layer 5) to 400 ft/d for coarse Rio Grande alluvium and alluvial fan deposits (both layer 1); most values range between 5 and 100 ft/d. Ratios of horizontal to vertical hydraulic conductivity in the CDWR model range from 2:1 to 10,000:1.

Water Budget

Investigators have developed water budgets for modern conditions in the groundwater system in various parts of the San Luis Valley. Three of these budgets represent overall inflow to and outflow from the area of study, rather than the groundwater system alone: Emery and others (1973) presented a budget for the Colorado part of the San Luis Valley, Huntley (1976) presented a budget for the San Luis closed basin, and Hearne and Dewey (1988) presented a budget for the Alamosa Basin. An estimated budget compiled for the CDWR groundwater-flow model of the Colorado part of the San Luis Valley represents inflow to and outflow from the groundwater system of the modeled area (table 1), and is, therefore, the focus of this discussion.

Also discussed in this section is a newly estimated predevelopment water budget for the groundwater system of the Colorado part of the San Luis Valley (table 1).

Table 1. Estimated modern water-budget components for the Colorado Division of Water Resources (2004) groundwater-flow model of the San Luis Valley, Colorado, and predevelopment water-budget components newly derived from documentation of the Colorado Division of Water Resources (2004) groundwater-flow model.

[All values are in acre-feet per year and are rounded to the nearest thousand. The predevelopment budget is intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and does not represent a rigorous analysis of individual recharge and discharge components. Percentages of water-budget components are illustrated on figure 5. GIS, geospatial information system; CDWR, Colorado Division of Water Resources]

	Predevelopment	Modern (1970-2002)	Change from predevelopment to modern
Budget component		**Recharge**	
Canal and lateral leakage (including canals without GIS data)	0	290,000	290,000
Surface-water irrigation	0	291,000	291,000
Groundwater irrigation	0	158,000	158,000
Rim recharge (margin streams and creeks not explicitly modeled as streams)	[1]166,000	166,000	0
Precipitation	[2]50,000	70,000	20,000
Surface-water runoff from irrigation	0	17,000	17,000
Streams (natural streams, drains and canals modeled as streams)	[3]71,000	124,000	53,000
Constant flux (Subsurface inflow from eastern and western boundaries)	[1]113,000	113,000	0
Wells	0	2,000	2,000
Flowing wells and springs	0	45,000	45,000
Total recharge	[4]399,000	[4]**1,275,000**	**876,000**
Budget component		**Discharge**	
Streams (natural streams, drains and canals modeled as streams)	[4]57,000	77,000	20,000
Constant flux (Subsurface outflow through layers 1-3 of southern boundary)	[1]36,000	36,000	0
General head (flow from layer 4 of southern boundary)	[1]13,000	13,000	0
Wells	0	623,000	623,000
Flowing wells and springs	[1]4,000	75,000	71,000
Subirrigation meadow	0	97,000	97,000
Subirrigation alfalfa	0	32,000	32,000
Native evapotranspiration	[5]289,000	389,000	100,000
Total discharge	**399,000**	[4]**1,341,000**	[4]**942,000**
Change in aquifer storage (total recharge minus total discharge)	0	-66,000	-66,000

[1]Value assumed to have changed insignificantly between predevelopment and modern conditions; equivalent to value used in the CDWR (2004) groundwater flow model.

[2]Value calculated by adjusting the formula used in the CDWR (2004) groundwater flow model to include no irrigated lands; possible elevation differences between irrigated and non-irrigated lands were not considered.

[3]Value equivalent to results of "no pumping" scenario of the CDWR (2004) groundwater flow model.

[4]Values do not total up exactly because of rounding.

[5]Value estimated by applying a reduction that maintained the same ratio used in the CDWR (2004) model between stream gain and native evapotranspiration, while balancing total discharge with total recharge.

This budget is intended only to provide a basis for comparison of overall magnitudes of recharge and discharge between predevelopment and modern conditions, and does not represent a rigorous analysis of individual recharge and discharge components. The budget was developed by changing the CDWR model budget as deemed appropriate to remove influences of development on the system. All explicitly anthropogenic recharge sources were removed (that is, all recharge from irrigation and irrigation infrastructure). Rim recharge, which results from infiltration of water from the channels of stream along the basin margins that were not explicitly modeled as streams, and constant fluxes representing groundwater inflow along the basin margins were assumed to be unchanged. Precipitation input was adjusted by changing the 10 percent infiltration of precipitation assumed by the CDWR method for irrigated lands during the growing season to the 3 percent infiltration assumed by the same method for non-irrigated lands, without attempting to account for differences in precipitation resulting from elevation. Stream infiltration was adjusted to match the infiltration simulated by the "no pumping" scenario of the CDWR model. This quantity of infiltration is believed to be reasonable because most stream infiltration under natural conditions probably occurred in areas near the basin margins, where groundwater levels generally have not risen enough as a result of irrigation to substantially affect stream infiltration.

Discharge components of the CDWR water budget were removed or adjusted to represent predevelopment conditions. In particular, all components related to wells (both pumped and flowing), subirrigation, and flow to drains were removed. The constant flux and general head components representing flow into New Mexico at the south end of the model were assumed to be virtually unchanged. Because stream gains simulated by the "no pumping" scenario of the CDWR model are likely to be greatly influenced by the simulated application of irrigation water, these simulated gains were not used in the estimated predevelopment budget. Instead, the reduction in discharge required to balance the overall recharge to the groundwater system was applied to both stream gain and native evapotranspiration using the ratio between these two values in the original CDWR budget. This method, therefore, assumes that neither stream gain nor native evapotranspiration increased disproportionally to the other as a result of development; both discharge components would be likely to have increased in the study area as a whole because agricultural development occurred both within and outside the San Luis closed basin. A small spring discharge component was maintained to reflect average annual spring discharges estimated for the CDWR model. Individual components of recharge to and discharge from the groundwater system under both predevelopment and modern conditions are illustrated in the conceptual diagrams of regional groundwater flow in figure 5.

Under natural conditions, subsurface inflow—primarily from the relatively permeable rocks of the San Juan Mountains on the west—was one of the largest sources of recharge to the groundwater system of the Alamosa Basin (Huntley, 1976; Hearne and Dewey, 1988). Almost 90 percent of the total 113,000 acre-ft/yr groundwater inflow simulated by constant-flux boundaries (table 1) in the CDWR model is from the San Juan Mountains. The quantity of recharge to both the unconfined and confined aquifers from subsurface inflow probably has not changed substantially between predevelopment and modern conditions. Leakage of groundwater upward from the confined aquifer in the central part of the basin was an additional source of recharge to the unconfined aquifer under natural conditions; despite changes in hydraulic head caused by withdrawals from both aquifers, upward leakage continues to be a source of water to the unconfined aquifer. Using a hydrologic budget for the San Luis closed basin, Emery and others (1975) estimated leakage from the confined to the unconfined aquifer across the area to be about 0.6 ft/yr.

Although depths to water are small throughout most of the Alamosa Basin, low precipitation rates combined with high evaporation rates result in only a small contribution of precipitation to groundwater recharge (Emery and others, 1973; Leonard and Watts, 1989). For the CDWR model, higher percentages of infiltration were assumed for precipitation falling on irrigated lands during the growing season (10 percent) and for the sand dune area (21 percent) than for irrigated and non-irrigated lands outside the growing season (3 percent). Also, by taking elevation into account, the resulting value for recharge from precipitation was about 70,000 acre-ft/yr (table 1). Adjustments to represent predevelopment conditions, when no irrigation-wetted lands would be present to enhance infiltration, resulted in a value of about 50,000 acre-ft/yr (table 1). Some water—including precipitation—that infiltrates along the margins of the valley migrates downward to recharge the confined aquifer that underlies the central part of the valley (Leonard and Watts, 1989; Anderholm, 1996) (fig. 5).

Infiltration of surface water is an important source of recharge to both the unconfined and confined aquifers of the San Luis Valley. The Rio Grande, which had a mean annual discharge of about 648,000 acre-ft for the period 1909–2006 where it enters the Alamosa Basin near Del Norte (USGS digital data for 1909–2006), gains water along most of its course through the valley (Hearne and Dewey, 1988). However, several smaller streams enter the valley from the surrounding mountains and lose substantial quantities of water to the aquifer, particularly near the basin margins (fig. 5).

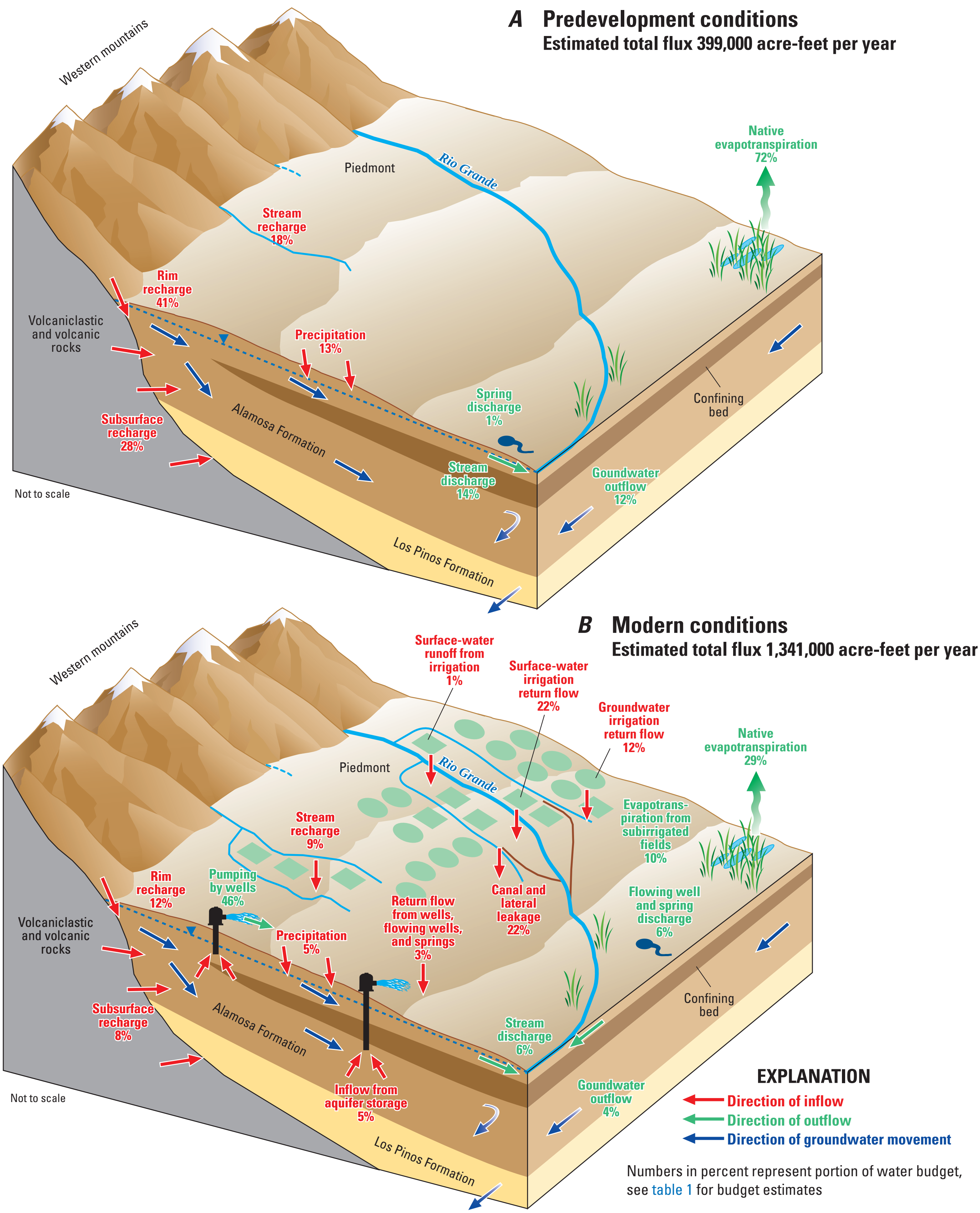


Figure 5. Generalized diagrams for the Alamosa Basin, Colorado, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

In the San Luis closed basin, these streams include Saguache, San Luis, and La Garita Creeks (Anderholm, 1996). In the part of the Alamosa Basin drained by the Rio Grande, important streams include the Conejos River, Alamosa River, and La Jara Creek, all originating in the San Juan Mountains, and Trinchera Creek, which flows out of the Sangre de Cristo Mountains. Along the margins of the valley, downward flow of water that originated as stream infiltration to the unconfined aquifer is a substantial component of recharge to the confined aquifer (Leonard and Watts, 1989; Anderholm, 1996).

Infiltration of water from streams flowing across the margins of the San Luis Valley was the primary source of surface-water recharge to the groundwater system under predevelopment conditions. For the CDWR model, this component of recharge was represented in part by a "rim recharge" term for all streams and creeks that were not explicitly represented in the model (table 1). This term, which should be unchanged between predevelopment and modern conditions, was estimated to be about 166,000 acre-ft/yr from information about precipitation rates and the drainage areas of surface-water basins along the valley margins. An unspecified portion of stream infiltration near the valley margins also is included in the approximately 124,000 acre-ft/yr of recharge from streams that are explicitly represented in the model (table 1); data provided in the CDWR model documentation indicate that about two-thirds of this amount likely represents infiltration from natural streams (as compared with canals and drains).

The development of irrigated agriculture has resulted in combined infiltration of applied irrigation water and canal leakage as the primary means through which surface water recharges the groundwater system—primarily the unconfined aquifer—under modern conditions (table 1 and fig. 5). Through surface-water diversions for irrigation, water from the Rio Grande is delivered throughout much of the Alamosa Basin; more than 180,000 acre-ft is diverted annually into the Rio Grande Canal that feeds the San Luis closed basin (Colorado Division of Water Resources, 2004). Most or all natural flow in tributaries is diverted for irrigation as well, resulting in recharge through canals and fields across broad areas, rather than at focused points along the mountain fronts. Leonard and Watts (1989) and Emery and others (1971a) state that return flow of irrigation water is now the single largest source of recharge to the unconfined aquifer in the Alamosa Basin. The CDWR model indicates that irrigation water applied to fields results in about 466,000 acre-ft/yr of recharge to the aquifer; canal and lateral leakage adds about 290,000 acre-ft/yr of recharge.

Agricultural and urban development has introduced additional sources of recharge to the groundwater system in the Alamosa Basin. Agricultural development has added infiltration of water that is pumped from the unconfined or confined aquifer and then applied to crops. Likely minor sources of recharge resulting from urbanization include seepage from septic tanks, sewer and water-distribution lines, and turf irrigation. Based on the estimated predevelopment flux through the groundwater system of about 399,000 acre-ft/yr through the Colorado part of the San Luis Valley (table 1), activities and practices associated with agricultural and urban development have more than tripled fluxes of water through the system.

Prior to the start of groundwater pumping, discharge from the unconfined aquifer of the Alamosa Basin took place primarily through evapotranspiration (Hearne and Dewey, 1988); in the San Luis closed basin, evapotranspiration was the only substantial means of discharge (Huntley, 1976 and 1979). Because the water table is close to the land surface throughout large areas of the Alamosa Basin, evapotranspiration can occur through direct evaporation of groundwater as well as through transpiration by phreatophytes. Most evapotranspiration is focused in the central, topographically low part of the Alamosa Basin, and particularly in the "ancestral sump" area of the San Luis closed basin (although groundwater pumping for the Closed Basin Project has recently lowered water levels and reduced evapotranspiration in this area). Because application of irrigation water has raised water levels across broad areas (Powell, 1958; Hearne and Dewey, 1988), evapotranspiration from the groundwater system of the San Luis Valley has increased overall as a result of agricultural development. The CDWR model simulates native evapotranspiration as about 389,000 acre-ft/yr and evapotranspiration from subirrigated meadows and crops as 129,000 acre-ft/yr. For the estimated predevelopment water budget of table 1, adjustment of the evapotranspiration component to balance groundwater inflows resulted in an estimated evapotranspiration of about 289,000 acre-ft/yr.

Direct groundwater discharge from the Alamosa Basin as underflow to areas to the south is believed to be small because of the relative impermeability of the San Luis Hills. Similarly, underflow from the southern tip of the San Luis Valley (as defined in fig. 1) to the Española Basin probably is relatively small; the hydrologic connection is primarily by means of the Rio Grande. Because the southern boundary of the CDWR model is the state line between Colorado and New Mexico, a component of groundwater underflow across that boundary is required to balance the model's budget for the groundwater system. The value of 49,000 acre-ft/yr (table 1) for this underflow is not likely to have changed substantially between predevelopment and modern conditions. The CDWR model also simulates discharge to springs, which is one means of discharge from the confined aquifer of the valley. The relatively small discharge of 4,000 acre-ft/yr for two major springs that are not explicitly represented as streams was not changed for the estimated predevelopment budget of table 1. Natural discharge from the confined aquifer to the unconfined aquifer through upward leakage is not explicitly represented in the budgets of table 1.

Besides evapotranspiration by native vegetation, the other relatively large component of discharge from the groundwater system of the San Luis Valley under natural conditions was outflow from the unconfined aquifer to streams. This component applies only to areas south of the San Luis closed basin, where the Rio Grande (the major surface-water feature) generally gains water as it traverses the valley (Hearne and Dewey, 1988). The overall quantity of discharge to surface-water features has increased under modern conditions as a result of larger fluxes of water through the groundwater system and locally higher water levels associated with crop irrigation, although pumping has likely intercepted some groundwater that would otherwise have discharged to the Rio Grande. The CDWR model simulates about 77,000 acre-ft/yr of groundwater flowing to streams and agricultural drains. For the estimated predevelopment water budget of table 1, adjustment of the stream discharge component to balance groundwater inflows resulted in an estimated flow of about 57,000 acre-ft/yr to streams.

The substantial use of groundwater in the San Luis Valley since about the 1950s, primarily for crop irrigation, has resulted in pumping becoming the largest component of discharge from the groundwater system under modern conditions. Documentation for the CDWR model indicates that about 52 percent of all wells in the San Luis Valley are completed in the unconfined aquifer and about 25 percent are completed in the upper few hundred feet of the confined aquifer. Despite state-imposed moratoriums on the construction of new high-capacity wells in the confined aquifer in 1972 and in the unconfined aquifer in 1981 (Colorado Division of Water Resources, 2004), the CDWR model simulates discharge by “pumping” wells at about 623,000 acre-ft/yr (table 1). Flowing wells cause a much smaller net discharge of water from the aquifer because a portion of flowing well discharge is unconsumed and is assumed to recharge the unconfined aquifer. Ultimately, because a portion of the large quantity of groundwater applied to crops is lost to evapotranspiration, the net effect of application of groundwater for irrigation is a decrease in the quantity of groundwater in the basin (Anderholm, 1996). As a result of this development of the groundwater resource, the CDWR model simulates a 66,000 acre-ft annual reduction in the quantity of water in aquifer storage.

Groundwater Flow

Water-level maps for 1968 conditions (Emery and others, 1971a) (fig. 6) and 1980 conditions (Crouch, 1985) in the unconfined aquifer of the San Luis Valley illustrate that groundwater flows generally from the eastern, western, and northern margins of the valley (the primary predevelopment recharge areas) toward its central axis. In the San Luis closed basin, flow is toward the topographic low known as the “ancestral sump” area, where natural saline lakes and salt deposits provide evidence of a great quantity of evapotranspiration. Small quantities of groundwater might also flow across the southern boundary of the closed basin (Leonard and Watts, 1989). In the southern part of the Alamosa Basin, groundwater flows primarily toward the Rio Grande, where most discharge occurs, and southward toward the Costilla Plains and Taos Plateau. Contours of the potentiometric surface in the confined aquifer (Emery and others, 1973) indicate that horizontal groundwater-flow directions are similar to those in the unconfined aquifer. Although the vertical flow of groundwater is downward in the recharge area around the perimeter of the Alamosa Basin (Hearne and Dewey, 1988), in the central part of the basin, hydraulic heads in the confined aquifer are higher than in the unconfined aquifer, resulting in upward leakage (Emery and others, 1973; Hearne and Dewey, 1988).

Groundwater pumping from the unconfined aquifer in the Alamosa Basin has caused some decline in water levels, particularly during years when surface water is in short supply for irrigation. Declines were apparent as early as 1980 in parts of the closed basin (Crouch, 1985). These declines are also evidenced by the reduction in groundwater storage simulated by the CDWR groundwater-flow model for the Colorado part of the San Luis Valley and by the calculations of Stogner (2005) indicating that the volume of water in the unconfined aquifer in part of the San Luis closed basin was about 10 percent less during 1997–2001 than it was during 1948–49. Maps of the 1997–2001 conditions in the unconfined aquifer in part of the closed basin (Stogner, 2005) illustrate that local water-level declines (and, therefore, decreases in saturated thickness) have occurred at least seasonally in this area.

With respect to the confined aquifer, Emery and others (1973) conducted an evaluation to determine whether substantial declines in hydraulic heads or changes in vertical gradients had occurred as a result of the removal of water through flowing wells or withdrawals for public supply. No evidence of widespread, long-term changes in heads or vertical gradients was found at that time. Although long-term water-level data are available for several wells in the confined aquifer of the San Luis Valley through at least 2000 (Colorado Division of Water Resources, 2004; Brendle, 2002), no subsequent investigations are known to have focused on reevaluation of this issue.

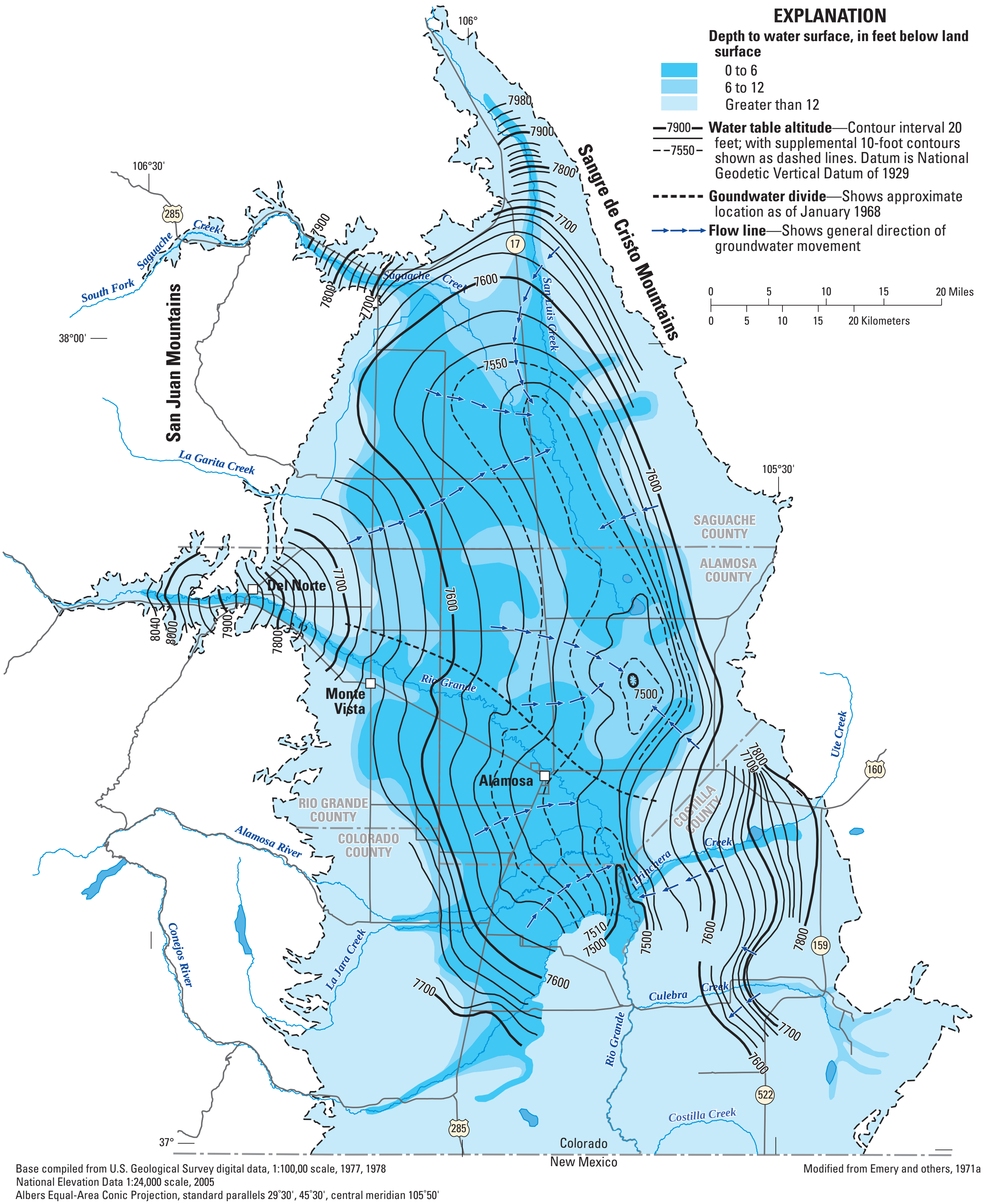


Figure 6. Groundwater levels that represent 1968 conditions in the unconfined aquifer of the San Luis Valley, Colorado.

Studies of groundwater age in the Alamosa Basin indicate that water in the unconfined aquifer typically contains a substantial fraction of young recharge (fig. 7*A*) (see Section 1 of this report for a discussion of groundwater age and environmental tracers). Mayo and others (2007) found that the tritium content of groundwater in the unconfined aquifer generally decreased from the mountain fronts toward the valley, consistent with the direction of flow inferred from water levels. They concluded that 50–100 years was a reasonable estimate of travel time from the San Juan Mountain front to the "ancestral sump" area of the San Luis closed basin, a distance of about 30 mi. Using tritium, chlorofluorocarbons, and carbon-14, Rupert and Plummer (2004) concluded that many water samples from the unconfined aquifer in the area of the Great Sand Dunes represented mixtures of young (post-1941) and old recharge, and that it took more than 60 years for the old fraction of groundwater to travel from the mountain front to the far side of the dunes, a distance of about 7 mi. Stogner (2005) used data on hydraulic gradients and aquifer properties for his study area in the closed basin to estimate a theoretical travel time of 400 years for a distance of 23 mi. Unpublished USGS data for chlorofluorocarbons in groundwater near the water table beneath agricultural areas in the San Luis Valley indicate substantial components of young water, recharged within the past 12–40 years prior to sampling.

Carbon-14 ages estimated by Mayo and others (2007) for water in the confined aquifer are generally older than 5,000 years—even relatively close to the mountain fronts—and become progressively older toward the central part of the Alamosa Basin, exceeding 27,000 years in some areas (fig. 7*B*). Carbon-14 age estimates by Rupert and Plummer (2004) of 4,300 and 30,000 years for two wells completed in the confined aquifer near the Great Sand Dunes indicated a similar age range.

Effects of Natural and Human Factors on Groundwater Quality

Groundwater quality in the San Luis Valley is determined by the source and composition of recharge and the processes occurring along a flow path, which are particularly important in the unconfined aquifer of the Alamosa Basin. Studies by Emery and others (1973), Edelman and Buckles (1984), Williams and Hammond (1989), Anderholm (1996), Stogner (1997, 2001, 2005), and Mayo and others (2007) have illustrated patterns in concentrations of dissolved solids and(or) nitrate for various parts of the valley. Anderholm (1996) and Stogner (2001) also discuss detections of organic compounds associated with human activities (volatile organic compounds [VOCs] and[or] pesticides) in groundwater of the Alamosa Basin.

General Water-Quality Characteristics and Natural Factors

The natural sources of groundwater recharge along the perimeter of the San Luis Valley tend to have low concentrations of dissolved solids, nitrate, and trace elements and tend to be oxidized. Mayo and others (2007) indicated that streams entering the valley typically have concentrations of dissolved solids less than 100 mg/L. The concentrations of dissolved solids in mountain springs, which might be indicative of groundwater underflow into the San Luis Valley, tend to be less than 200 mg/L (Mayo and others, 2007). The low concentrations of dissolved solids of stream infiltration, groundwater inflow, and precipitation recharging along the valley perimeter are reflected in both the unconfined and confined aquifers in this area, where groundwater commonly has values of specific conductance less than 250 µmhos/cm and(or) concentrations of dissolved solids less than 250 mg/L (Emery and others, 1973; Mayo and others, 2007) (fig. 8). Groundwater near the valley perimeter also tends to have concentrations of nitrate less than about 3 mg/L as nitrogen (Emery and others, 1973), which is considered the background concentration for the area (Stogner, 2001). Anderholm (1996) found generally low concentrations of trace elements (less than drinking-water standards) in water from 35 wells completed in the unconfined aquifer, even in the central part of the valley. However, arsenic (believed to be from natural sources) was elevated above the USEPA drinking-water standard of 10 µg/L in three wells toward the center of the valley and above 5 µg/L in a total of seven wells. Uranium was naturally elevated above the USEPA drinking-water standard of 30 µg/L in two wells toward the center of the valley, with a maximum concentration of 84 µg/L. Gross alpha activity exceeded the drinking-water standard in eight wells in the same area, and concentrations of radon generally exceeded 1,000 pCi/L throughout the study area (the USEPA has proposed a drinking-water standard of 300 pCi/L, along with an alternate standard of 4,000 pCi/L that would apply in states where programs are in place to reduce radon levels in indoor air [U.S. Environmental Protection Agency, 2010]). In areas of the unconfined aquifer away from the valley center, Anderholm (1996) and Rupert and Plummer (2004) found concentrations of dissolved oxygen generally were greater than 1 mg/L and pH values generally were between about 7 and 8, consistent with data in Mayo and others (2007).

Dissolved-solids concentrations, water types, and redox conditions tend to change as groundwater moves toward the center of the San Luis Valley, particularly in the unconfined aquifer of the San Luis closed basin. In the "ancestral sump" area, concentrations can exceed 20,000 mg/L (Williams and Hammond, 1989; Mayo and others, 2007) (fig. 8*A*). The groundwater tends to change from a calcium bicarbonate type near the valley perimeter to a sodium bicarbonate type down

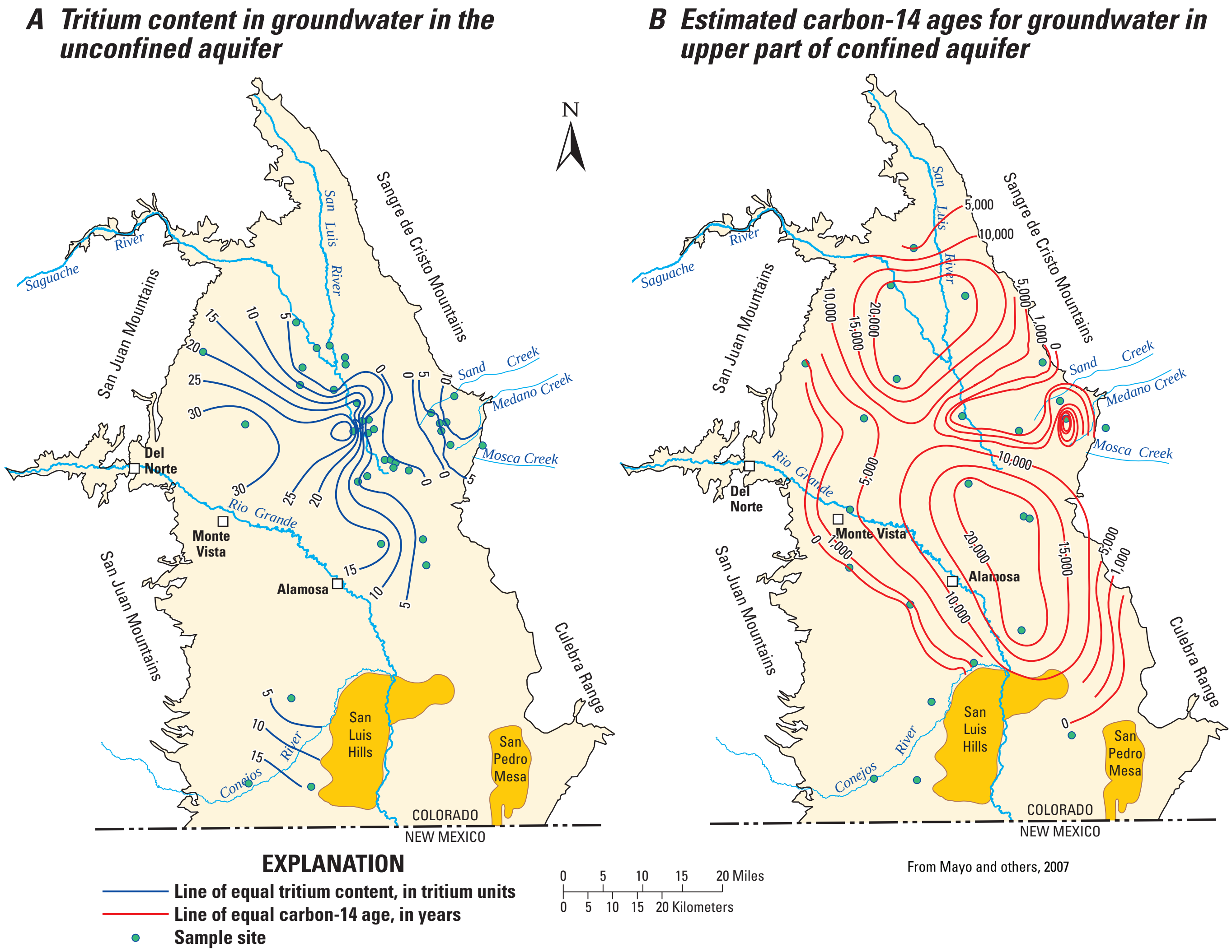


Figure 7. Distribution of (*A*) tritium content in groundwater in the unconfined aquifer and (*B*) estimated carbon-14 ages for groundwater in the upper part of the confined aquifer in the Alamosa Basin, Colorado.

gradient (Williams and Hammond, 1989), although water with elevated concentrations of sulfate and chloride also is found in the sump area (Mayo and others, 2007). Some investigators have concluded that the principal cause of the large increases in concentrations of dissolved solids in the sump area is evapotranspiration (Huntley, 1976; Williams and Hammond, 1989). Other investigators, while acknowledging that evapotranspiration is an important influence on the chemistry of groundwater in the sump area (particularly at very shallow depths), have concluded that dissolution of minerals including gypsum and halite is perhaps the most important factor in increasing concentrations of dissolved solids along flow paths in the unconfined aquifer of the valley (Emery and others, 1973; Mayo and others, 2007). Ion exchange has been cited as a major factor in the increase in the dominance of sodium in groundwater toward the sump area in the unconfined aquifer (Emery and others, 1973; Williams and Hammond, 1989; Mayo and others, 2007), although Emery and others (1973) also mention calcite precipitation as a factor. The likely effects of irrigated agriculture on dissolved-solids concentrations and elevated nitrate concentrations (particularly in the San Luis closed basin) in the unconfined aquifer will be discussed in the following section. Concentrations of dissolved oxygen in the unconfined aquifer tend to be less than 1 mg/L in parts of the study area nearest the ancestral sump (Anderholm, 1996); associated concentrations of manganese are larger here than in other parts of the study area, indicating a likely transition toward reduced conditions. Mayo and others (2007) found median pH values in the unconfined aquifer near the sump area to be 8 or above.

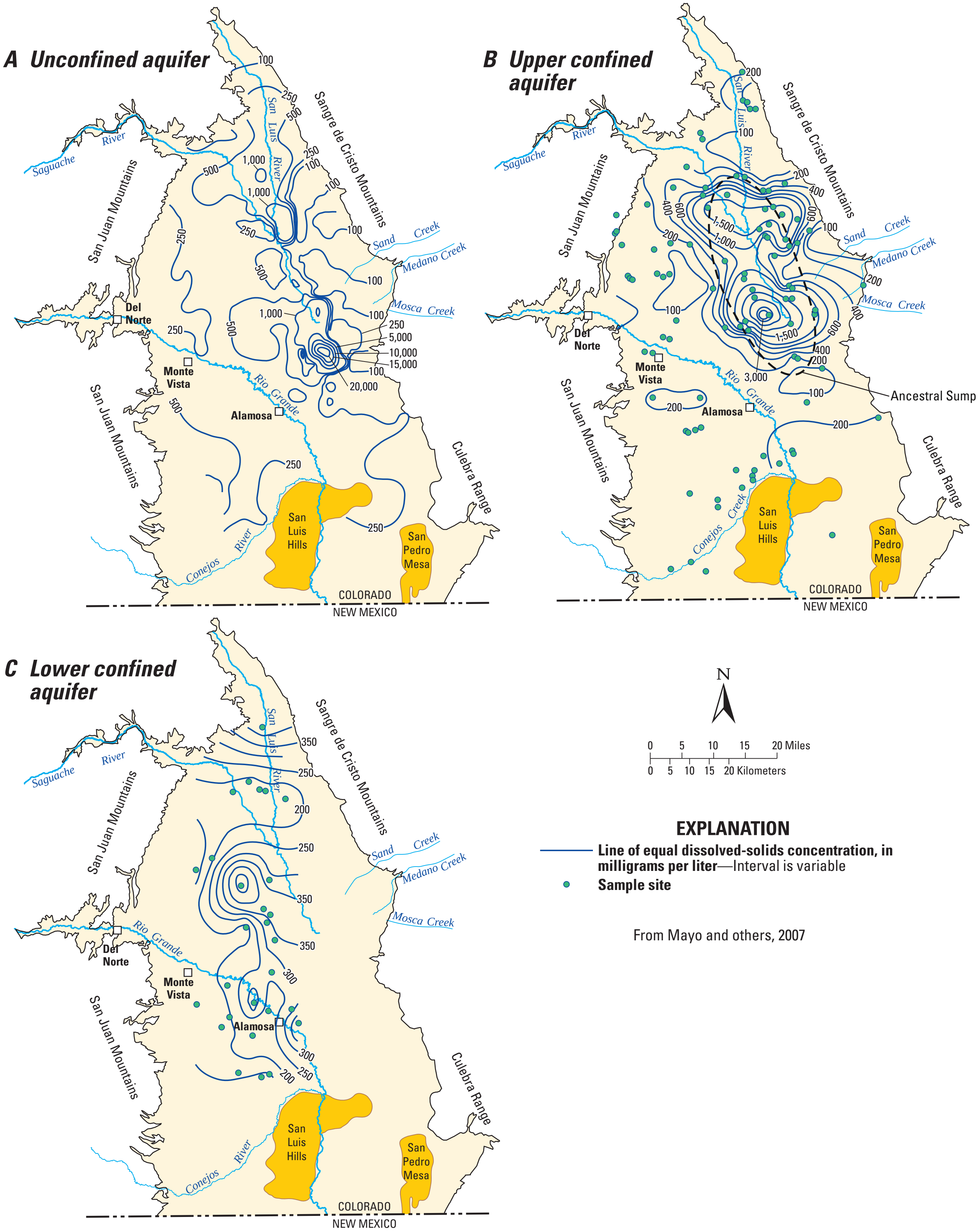


Figure 8. Distribution of dissolved-solids concentrations for the (*A*) unconfined, (*B*) upper confined, and (*C*) lower confined aquifers of the Alamosa Basin, Colorado.

Throughout most of the valley, concentrations of dissolved solids in the confined aquifer are less than those in the unconfined aquifer (fig. 8). Similar to the unconfined aquifer, changes in water chemistry also occur along flow paths in the confined aquifer of the San Luis Valley, although the changes tend to be less dramatic in the deeper confined aquifer. Increases in concentrations of dissolved solids and sodium are observed in the upper part of the confined aquifer in the "ancestral sump" area of the San Luis closed basin (fig. 8*B*). Because the confined aquifer is too deep to be affected by evapotranspiration and probably is little influenced by other near-surface processes, these changes in chemistry have been attributed to reactions with aquifer materials, including mineral dissolution and cation exchange (Emery and others, 1973; Mayo and others, 2007). Mayo and others (2007) concluded that methanogenic driven ion-exchange reactions are important in the upper confined aquifer in the sump area, where they and Emery and others (1973) detected methane and(or) hydrogen sulfide gas, which indicates reduced conditions. Even outside the ancestral sump area, Rupert and Plummer (2004) found concentrations of dissolved oxygen below 0.5 mg/L and the presence of manganese in two wells completed in the upper confined aquifer; pH values were 8.5 and 8.7. Mayo and others (2007) reported median pH values for the upper confined aquifer ranging from 8.3 in the sump area to 7.8 in other areas. For the lower confined aquifer, Mayo and others (2007) found that concentrations of dissolved solids were less than 250 mg/L throughout most of the San Luis Valley (fig. 8*C*); they reported median pH values ranging between 7.9 and 8.6 for different areas of the valley.

Potential Effects of Human Factors

As mentioned in previous parts of this section, the long history of agricultural development in the San Luis Valley has resulted in several substantial changes to the hydrologic system, including changes in the source, distribution, quantity, and chemical characteristics of recharge to the groundwater system. Groundwater levels and gradients also have been affected by the application of irrigation water to crops and by associated groundwater pumping. Observed and potential effects of these changes on groundwater quality in the San Luis Valley are discussed in this section. The discussion focuses in particular on the unconfined aquifer of the Alamosa Basin because this is the part of the groundwater system that has been most greatly affected by changes associated with human activities. In contrast, the confined aquifer is believed to have poor hydraulic connection with the land surface (Edelmann and Buckles, 1984) because of its depth, protective confining layer, and generally upward hydraulic gradients. Documented effects of human activities on groundwater quality in the unconfined aquifer of the Alamosa Basin are summarized in table 2.

Table 2. Summary of documented effects of human activities on groundwater quality in the Alamosa Basin, Colorado.

Groundwater-quality effect	Cause	General location(s)	Reference(s)
Elevated concentrations of nitrate	Agricultural fertilizer application	Unconfined aquifer beneath agricultural areas of the Alamosa Basin	Emery and others (1973); Edelmann and Buckles (1984); Anderholm (1996); Stogner (1997, 2001, and 2005)
Elevated concentrations of dissolved solids	Irrigation of agricultural fields	Unconfined aquifer beneath agricultural areas of the Alamosa Basin	Emery and others (1973); Huntley (1976); Edelmann and Buckles (1984); Williams and Hammond (1989)
Detections of agricultural pesticides	Agricultural pesticide application	Unconfined aquifer (including some domestic wells) beneath agricultural areas of the Alamosa Basin	Durnford and others (1990); Austin (1993); Anderholm (1996)
Detections of volatile organic compounds	Not determined	Only one documented detection near the water table beneath a primarily agricultural area of the San Luis closed basin	Anderholm (1996)
Detections of non-agricultural pesticides	Not determined	Only one documented detection near the water table beneath a primarily agricultural area of the San Luis closed basin	Anderholm (1996)

Irrigated agriculture and its supporting infrastructure have added to the sources and areal extent of groundwater recharge across much of the Alamosa Basin. Water from the Rio Grande was not a source of recharge under predevelopment conditions, but is now delivered by canals throughout much of the Alamosa Basin (including the San Luis closed basin) for irrigation. Water from tributaries that used to infiltrate only along the valley perimeter also is diverted for irrigation and now enters the groundwater system through infiltration from canals, fields, and recharge pits. Irrigation of crops with surface water by subirrigation has raised the water table in some areas, resulting in increased evapotranspiration. Evapotranspiration of irrigation water applied to fields can increase the dissolved-solids concentrations of the excess irrigation water that recharges the groundwater system. This water can also potentially transport to the water table the fertilizers and pesticides that were applied to fields. The advent of groundwater pumping from the unconfined aquifer to increase water supplies for irrigation in the Alamosa Basin (particularly the San Luis closed basin) has resulted in recycling of the groundwater on relatively short time scales, further increasing its exposure to evapotranspiration and agricultural chemicals. Agricultural development in the Alamosa Basin has, therefore, resulted in increased fluxes over broader areas and has introduced the means for potential transport of anthropogenic chemicals and increased dissolved solids to the unconfined aquifer throughout much of the basin.

Although the effects have not been quantified, several investigators have stated that irrigation-return flow has likely resulted in increased concentrations of dissolved solids in the unconfined aquifer of the Alamosa Basin (Emery and others, 1973; Huntley, 1976; Edelmann and Buckles, 1984; Williams and Hammond, 1989). Because applied irrigation water undergoes evapotranspiration and dissolves minerals from the soil and sediments as it recharges, irrigation-return flow contains more solutes than the applied irrigation water. Increases in concentrations of dissolved solids as a result of the irrigation cycle are likely to be most pronounced in areas of the basin where groundwater is a primary source of irrigation water and is recycled multiple times for this purpose. A study by Anderholm (1996) of shallow groundwater quality beneath areas of intense agriculture in the Alamosa Basin indicated wide local variations in concentrations of dissolved solids (ranging in value from 75 mg/L to 1,960 mg/L) superimposed on a general increase in concentrations from west to east. On the basis of ratios among various major ions, Anderholm (1996) found that compositions of several of the groundwater samples were similar to that of surface water that had been concentrated by evaporation; such samples might be indicative of irrigation water containing solutes that have been concentrated during recharge.

The effects of irrigated agriculture on concentrations of nitrate in the unconfined aquifer have been well documented, particularly in the San Luis closed basin (Emery and others, 1973; Edelmann and Buckles, 1984; Anderholm, 1996; Stogner, 1997, 2001, and 2005). Stogner (2001) noted that use of inorganic nitrogen fertilizers in the San Luis Valley began in the 1940s and increased dramatically starting in the 1960s, and that observed distributions of nitrate in shallow groundwater have been consistent with the overall pattern of fertilizer use though time. Early concentrations of nitrate reported by Scofield (1938) for 38 shallow wells in the San Luis Valley were all 0.3 mg/L or less. In subsequent samples collected during 1946–1950, Powell (1958) detected concentrations of nitrate of 3.2 mg/L or more in about 5 percent of wells.

Emery and others (1973) were among the first to map the common occurrence of concentrations of nitrate exceeding 10 mg/L as nitrogen in the unconfined aquifer of the closed basin; they attributed these elevated concentrations to heavy applications of chemical fertilizer during the previous decade. Similar patterns of nitrate concentration were observed by Edelmann and Buckles (1984), who additionally determined that concentrations of nitrate tended to be smaller toward the base of the unconfined aquifer. Anderholm (1996) detected concentrations of nitrate of 8.5 mg/L or more in several wells completed near the water table both north and south of the Rio Grande, and stated that the elevated concentrations were indicative of fertilizer leaching. Stogner (2005) used changes in the distribution of concentrations of nitrate through time (fig. 9) to estimate changes in nitrate mass in the unconfined aquifer beneath an intensively cultivated area of the closed basin. Stogner (2005) estimated that nitrate mass increased from about 6,900 tons in the 1940s to 34,000 tons in the late 1960s, and to 75,000 tons in the late 1990s.

The conclusion that agricultural practices are primarily responsible for the observed long-term increases in nitrate concentration and mass in the unconfined aquifer of the San Luis Valley is supported by the field experiments of Eddy-Miller (1993) and LeStrange (1995), which documented nitrogen leaching from irrigated fields. Stogner (1997, 2001) indicated that changes in farm-management practices (including changes in irrigation scheduling and reductions in the amount of fertilizer applied) that could reduce nitrate leaching are being encouraged in the San Luis Valley. Using study results indicating that net reductions in nitrate leaching of about 50 percent could be achieved by improved management practices (Sharkoff and others, 1996), Stogner (2005) calculated that resulting declines in the total mass of nitrate in the unconfined aquifer would be measurable within 10 to 15 years.

In addition to nitrate, pesticides have recently been studied in the unconfined aquifer of the San Luis Valley because of their potential to leach to groundwater. The pesticides Bravo, Sencor, Eptam, and(or) 2,4-D were detected at trace or low levels (7 µg/L or less) in samples from up to 10 of 34 irrigation wells sampled during the 1990 growing season by Durnford and others (1990), although the investigators indicated that sample or well-bore contamination may have affected these findings. On the basis of results from associated modeling of groundwater vulnerability in the area to pesticide contamination, Durnford and others (1990) concluded that farm-management practices and individual pesticide properties were important factors in determining contamination potential. Samples collected during the summer of 1993 from the

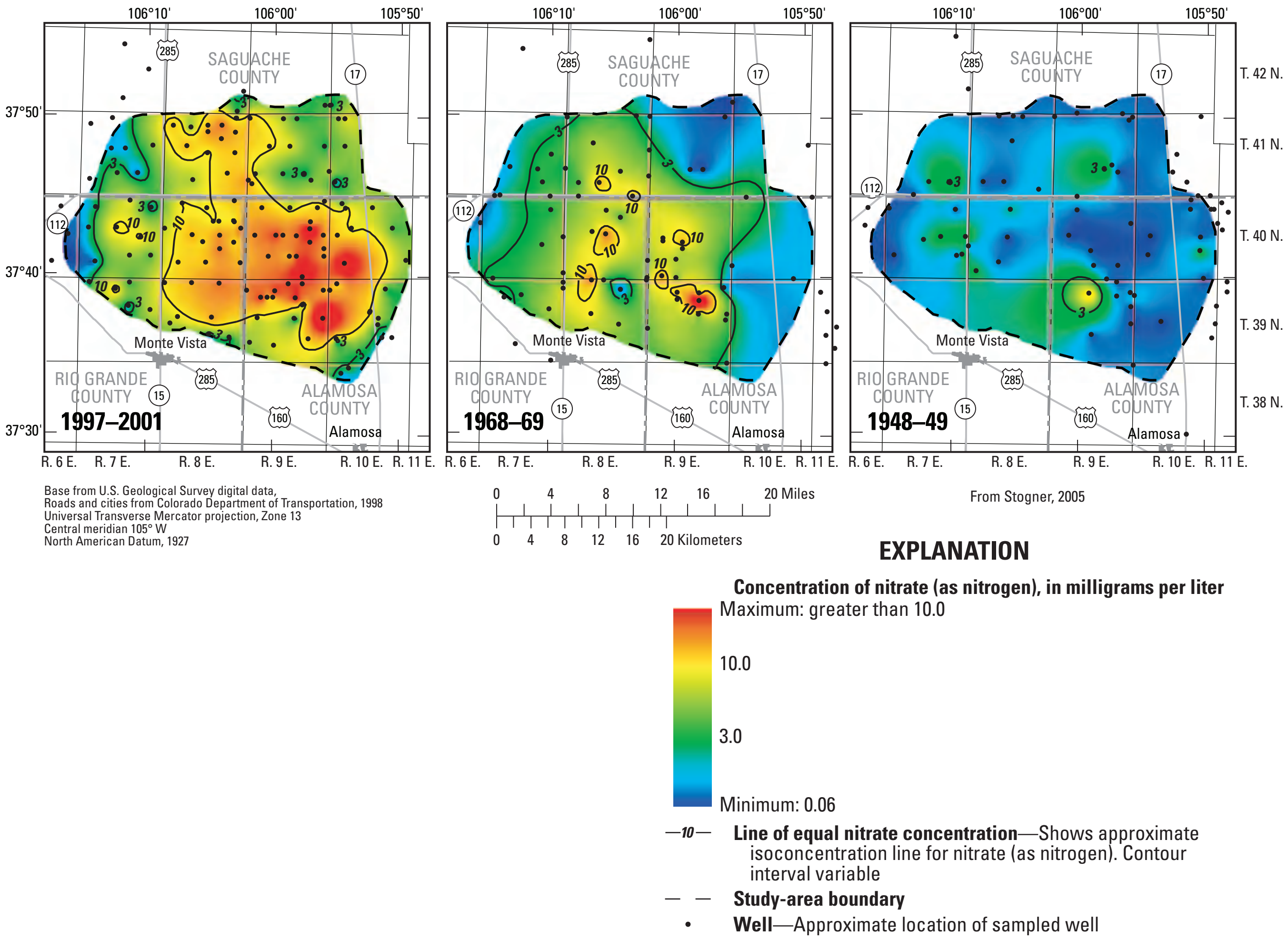


Figure 9. Estimated distribution of the concentration of nitrate (as nitrogen) in groundwater from the unconfined aquifer during 1997–2001, 1968–69, and 1948–49 for part of the Alamosa Basin, Colorado.

35 water-table wells studied by Anderholm (1996) showed only trace amounts (0.072 µg/L or less) of metribuzin, prometon (a nonagricultural herbicide), metolachlor and(or) *p,p*′-DDE in five wells, leading Anderholm (1996) to conclude that there was no widespread contamination of the unconfined aquifer by pesticide compounds. Samples collected between May and August of 1993 by the Colorado Department of Health and Environment from 93 domestic wells completed in the unconfined aquifer showed 2,4-D, hexazinone, and(or) lindane in three wells at concentrations up to 0.29 µg/L (Austin, 1993). Taken together, these studies appear to indicate that the unconfined aquifer of the San Luis Valley has been less affected by pesticide leaching than by nitrate leaching, perhaps because the pesticides used in the area are less mobile and persistent.

Potential effects of urbanization and septic tanks on water quality in the unconfined aquifer of the San Luis Valley are not known to have been specifically investigated. In one shallow well in the agricultural area studied by Anderholm (1996), however, one VOC (methyl *tert*-butyl ether) was detected at a concentration of 6 µg/L; the nonagricultural herbicide prometon was also detected in one well at a concentration of 0.01 µg/L. Given shallow depths to water and the occurrence of recent recharge throughout much of the San Luis Valley, there would appear to be potential for urban activities to affect shallow groundwater quality in the area.

Another activity with the potential to locally affect groundwater quality within the Alamosa Basin is metals mining, which is conducted in parts of the San Juan Mountains. Mine drainage has affected surface-water quality in the Alamosa River and in Little Kerber and Kerber Creeks, which enter the San Juan closed basin from the west (Emery and others, 1973). Balistrieri and others (1995) concluded that elevated concentrations of arsenic, cobalt, chromium, copper, nickel, and zinc in the Alamosa River downstream from its confluence with the Wightman Fork were likely associated with mine drainage; wetlands within the San Luis Valley that receive water from the Alamosa River also contained elevated concentrations of several of these elements. The potential effects of recharge from these sources on local groundwater quality are not known to have been studied.

Summary

The San Luis Valley in Colorado and New Mexico, which includes the Alamosa Basin, is an extensive alluvial basin with an unconfined aquifer having high intrinsic susceptibility and vulnerability to contamination as a consequence of small depths to water and widespread areal recharge, much of which now results from irrigated agriculture. The San Luis closed basin at the northern end of the valley is internally drained, whereas the groundwater system farther south is hydraulically connected to the Rio Grande, which gains water along most of its course through the area. Except near the basin margins, depths to water in the Alamosa Basin are commonly less than about 25 ft, and a thick fine-grained layer having its top at about 60 to 120 ft below land surface defines the division between the shallow, unconfined aquifer and a deeper, confined aquifer. Most wells are completed in the Alamosa Formation—consisting of discontinuous beds of clay, silt, sand, and gravel of mixed fluvial, lacustrine, and eolian origin—or in overlying deposits of similar lithology. Under natural conditions, groundwater recharges primarily along the basin margins as mountain-front recharge or groundwater underflow and discharges primarily in the central part of the basin as evapotranspiration. Because precipitation is small compared with evaporation, the direct infiltration of precipitation makes only a relatively minor contribution to aquifer recharge.

A long history of intensive agricultural land use has had a substantial effect on the groundwater-flow system in the unconfined aquifer of the Alamosa Basin. The estimated annual flux of water entering and leaving the groundwater system in the Colorado part of the San Luis Valley has more than tripled since development began. Most of this increased flux is the result of the effects of irrigation and its associated infrastructure, which has spread recharge across broad areas. Irrigation of croplands also has affected the chemical composition of recharge through evapotranspiration and recycling of shallow groundwater, which is pumped for application to crops at rates that make it the main component of discharge from the aquifer under modern conditions. Rates of evapotranspiration have also increased in some areas, primarily as the result of a rise in the water table resulting from irrigation. Even though groundwater withdrawals from both the unconfined and confined aquifers for irrigation and public supply have resulted in declines in aquifer storage, no large-scale changes in hydraulic gradients have been documented. Because the population of the basin remains small, urbanization has so far had little effect on the groundwater system.

Groundwater chemistry in the Alamosa Basin is determined by the source and composition of recharge and by processes occurring along a flow path, which are particularly important in the unconfined aquifer. Concentrations of dissolved solids are naturally high in the central part of the basin as a result of mineral dissolution and evapotranspiration, but also have increased in some areas because of irrigated agriculture. Naturally occurring concentrations of uranium and radon also might restrict the suitability of groundwater for consumption in some areas. Concentrations of nitrate, which were less than 3 mg/L throughout the basin prior to agricultural development, have increased to more than 10 mg/L over broad areas (particularly in the San Luis closed basin) as a result of the leaching of fertilizers applied to crops. Pesticides have been detected in shallow groundwater beneath agricultural areas, but not ubiquitously and generally in only trace concentrations. The occurrence of tracers of young water in shallow wells over broad areas of the Alamosa Basin is indicative of the susceptibility and vulnerability of the unconfined aquifer to contamination. In contrast, the confined aquifer is probably not substantially affected by near-surface processes, as indicated by generally upward hydraulic gradients and estimated groundwater ages on the order of thousands of years.

References Cited

Anderholm, S.K., 1996, Water-quality assessment of the Rio Grande Valley, Colorado, New Mexico, and Texas—Shallow groundwater quality of a land-use area in the San Luis Valley, south-central Colorado, 1993: U.S. Geological Survey Water-Resources Investigations Report 96–4144, 94 p.

Austin, Bradford, 1993, Groundwater monitoring activities, San Luis Valley unconfined aquifer, 1993: Report to the Commissioner of Agriculture, Colorado Department of Agriculture, 27 p.

Balistrieri, L.S., Gough, L.P., Severson, R.C., Montour, M., Briggs, P.H., Adrian, B., Curry, J., Fey, D., Hageman, P., and Papp, C., 1995, The effect of acidic, metal-enriched drainage from the Wightman Fork and Alamosa River on the composition of selected wetlands in the San Luis Valley, Colorado: U.S. Geological Survey Open-File Report 95–568, 37 p.

Brendle, D.L., 2002, Geophysical logging to determine construction, contributing zones, and appropriate use of water levels measured in confined-aquifer network wells, San Luis Valley, Colorado, 1998–2000: U.S. Geological Survey Water-Resources Investigations Report 02–4058, 58 p. Available at http://pubs.usgs.gov/wri/WRIR02-4058/.

Burroughs, R.L., 1981, A summary of the geology of the San Luis Basin, Colorado-New Mexico, with emphasis on the geothermal potential for the Monte Vista graben: Denver, Colo., Colorado Geological Survey Special Publication 17, 30 p.

Colorado Division of Water Resources, 2004, Preliminary draft—Rio Grande decision support system phase 4 groundwater model documentation: Colorado Division of Water Resources, August 2004, variously paged.

Crouch, T.M., 1985, Potentiometric surface, 1980, and water-level changes, 1969–80, in the unconfined valley-fill aquifers of the San Luis Basin, Colorado and New Mexico: U.S. Geological Survey Hydrologic Investigations Atlas HA–683, 2 plates.

Durnford, D.S., Thompson, K.R., Ellerbroek, D.A., Loftis, J.C., Davies, G.S., and Knutson, K.W., 1990, Screening methods for groundwater pollution potential from pesticide use in Colorado agriculture: Colorado State University, Colorado Water Resources Research Institute Completion Report no. 157, variously paged.

Eddy-Miller, C.A., 1993, Evaluation of shallow groundwater wells as a method of monitoring nitrate leaching in the San Luis Valley: Fort Collins, Colorado State University, unpublished M.S. thesis, 77 p.

Edelmann, P., and Buckles, D.R., 1984, Quality of groundwater in agricultural areas of the San Luis Valley, south-central Colorado: U.S. Geological Survey Water-Resources Investigations Report 83–4281, 37 p.

Emery, P.A., Boettcher, A.J., Snipes, R.J., and McIntyre, H.J., 1971a, Hydrology of the San Luis Valley, south-central Colorado: U.S. Geological Survey Hydrologic Investigations Atlas HA–381, 2 plates.

Emery, P.A., Dumeyer, J.M., and McIntyre, H.J., Jr., 1969, Irrigation and municipal wells in the San Luis Valley, Colorado: U.S. Geological Survey Open-File Report 70–119, 7 p.

Emery, P.A., Patten, E.P., Jr., and Moore, J.E., 1975, Analog model study of the hydrology of the San Luis Valley, south-central Colorado: Denver, Colo., Colorado Water Conservation Board Ground-Water Circular 29, 21 p.

Emery, P.A., Snipes, R.J., Dumeyer, J.M., 1971b, Location of wells and surface-water measurement sites, San Luis Valley, Colorado: U.S. Geological Survey Basic-Data Release Number 22, 1 plate.

Emery, P.A., Snipes, R.J., Dumeyer, J.M., and Klein, J.M., 1973, Water in the San Luis Valley, south-central Colorado: Colorado Water Resources Circular 18, 26 p.

Green, G.N., 1992, The digital geologic map of Colorado in ARC/INFO format: U.S. Geological Survey Open-File Report 92–0507. Available at http://pubs.usgs.gov/of/1992/ofr-92-0507/.

Green, G.N., and Jones, G.E., 1997, The digital geologic map of New Mexico in ARC/INFO format: U.S. Geological Survey Open-File Report 94–0052. Available at http://pubs.usgs.gov/of/1997/ofr-97-0052/.

Harmon, E.J., 2000, RGDSS groundwater, task 16—Pumping data: Lakewood, Colo., HRS Water Consultants, Inc. Appendix D—Pumping data, final memorandum, 16 p.

Hearne, G.A., and Dewey, J.D., 1988, Hydrologic analysis of the Rio Grande Basin north of Embudo, New Mexico—Colorado and New Mexico: U.S. Geological Survey Water-Resources Investigations Report 86–4113, 244 p.

Huntley, D., 1976, Groundwater recharge to the aquifers of northern San Luis Valley, Colorado: Golden, Colo., Colorado School of Mines, Remote Sensing Report 76–3, 247 p.

Huntley, D., 1979, Ground-water recharge to the aquifers of northern San Luis Valley, Colorado—Summary: Geological Society of America Bulletin, v. 90, no. 8, Part I, p. 707–709.

Leonard, G.J., and Watts, K.R., 1989, Hydrogeology and simulated effects of groundwater development on an unconfined aquifer in the Closed Basin Division, San Luis Valley, Colorado: U.S. Geological Survey Water-Resources Investigations Report 87–4284, 41 p.

LeStrange, S.L., 1995, Identification of spatial patterns in nitrate leaching: Fort Collins, Colorado State University, unpublished Ph.D. dissertation, 97 p.

Mayo, A.L., Davey, A., and Christiansen, D., 2007, Groundwater-flow patterns in the San Luis Valley, Colorado, USA revisited—An evaluation of solute and isotopic data: Hydrogeology Journal, v. 15, p. 383–408.

Mayo, A.L., and Webber, W., 1991, Preliminary evaluation of the factors contributing to spatial and temporal water-quality variations in Stage 3-Closed Basin Project salvage wells, San Luis Valley, Colorado: Orem, Utah, Mayo and Associates Final Report to the Rio Grande Water Conservancy District, variously paged.

McCalpin, J.P., 1996, General geology of the northern San Luis Valley, Colorado, *in* Thompson, R.A., Hudson, M.R., and Pillmore, C.L., eds., Geologic excursions to the Rocky Mountains and beyond, field trip guidebook for the 1996 annual meeting, GSA, Denver: Denver, Colorado Geological Survey Special Publication 44, variously paged.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

Miller, K.A., MacDonnell, L.J., and Rhodes, S.L., 1993, Groundwater rights in an uncertain environment—Theoretical perspectives on the San Luis Valley: Natural Resources Journal, v. 33, no. 3, p. 727–758.

Oak Ridge National Laboratory, 2005, LandScan Global Population Database, accessed September 2006 at http://www.ornl.gov/landscan/.

Powell, W.J., 1958, Groundwater resources of the San Luis Valley, Colorado: U.S. Geological Survey Water-Supply Paper 1379, 284 p.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM), Oregon State University, accessed May 17, 2010 at http://www.ocs.oregonstate.edu/prism/.

Robson, S.G., and Banta, E.R., 1995, Groundwater atlas of the United States—Segment 2, Arizona, Colorado, New Mexico, Utah: U.S. Geological Survey Hydrologic Atlas 730–C, 32 p. Available at http://pubs.usgs.gov/ha/ha730/ch_c/index.html.

Rupert, M.G., and Plummer, L.N., 2004, Groundwater flow direction, water quality, recharge sources, and age, Great Sand Dunes National Monument, south-central Colorado, 2000–2001: U.S. Geological Survey Scientific Investigations Report 2004–5027, 28 p. Available at http://pubs.usgs.gov/sir/2004/5027/.

Scofield, C.S., 1938, Quality of water of the Rio Grande Basin above Fort Quitman, Texas—Analytical data: U.S. Geological Survey Water-Supply Paper 839, 294 p.

Sharkoff, J.L., Delgado, A.A., Follett, R.F., and Riggenback, R.R., 1996, Nitrate leaching assessment of San Luis Valley Water Quality Demonstration Project sites, *in* Annual Potato/Grain Conference, 13th, Monte Vista, Colo., 1996, Proceedings: Colorado State University Cooperative Extension, p. 9–10.

Stogner, R.W., Sr., 1997, Variability of nitrate concentrations in the shallow groundwater in a selected area of the San Luis Valley, south-central Colorado: U.S. Geological Survey Fact Sheet FS–004–97, 4 p. Available at http://pubs.usgs.gov/fs/fs004-97/.

Stogner, R.W., Sr., 2001, Nitrate concentrations, 1936–99, and pesticide concentrations, 1990–99, in the unconfined aquifer in the San Luis Valley, Colorado: U.S. Geological Survey Water-Resources Investigations Report 01–4131, 12 p.

Stogner, R.W., Sr., 2005, Distribution and mass of nitrate in the unconfined aquifer beneath the intensively cultivated area north of the Rio Grande, San Luis Valley, Colorado, 1997 through 2001: U.S. Geological Survey Scientific Investigations Report 2004–5290, 28 p. Available at http://pubs.usgs.gov/sir/2004/5290/.

Upson, J.E., 1939, Physiographic subdivisions of the San Luis Valley, southern Colorado: Journal of Geology, v. 47, no. 7, p. 721–736.

Upson, J.E., 1941, The Vallejo Formation; new early Tertiary red-beds in southern Colorado: American Journal of Science, v. 239, p. 577–589.

U.S Census Bureau, 2001a, GCT-PH1. Population, housing units, area, and density: 2000 geography: Colorado counties, accessed July 2007 at http://factfinder.census.gov/servlet/GCTTable?_bm=y&-geo_id=04000US08&-_box_head_nbr=GCT-PH1&-ds_name=DEC_2000_SF1_U&-redoLog=false&-mt_name=PEP_2006_EST_GCTT1_ST2&-format=ST-2.

U.S Census Bureau, 2001b, GCT-PH1. Population, housing units, area, and density: 2000 geography: Colorado places, accessed July 2007 at http://factfinder.census.gov/servlet/GCTTable?_bm=y&-geo_id=04000US08&-_box_head_nbr=GCT-PH1&-ds_name=DEC_2000_SF1_U&-redoLog=false&-mt_name=DEC_2000_SF1_U_GCTP1_ST1&-format=ST-7.

U.S Census Bureau, 2001c, GCT-PH1. Population, housing units, area, and density: 2000 geography: New Mexico counties, accessed July 2007 at http://factfinder.census.gov/servlet/GCTTable?_bm=y&-context=gct&-ds_name=DEC_2000_SF1_U&-mt_name=DEC_2000_SF1_U_GCTPH1_ST2&-tree_id=4001&-redoLog=true&-_caller=geoselect&-geo_id=04000US35&-format=ST-2|ST-2S&-_lang=en.

U.S Census Bureau, 2001d, GCT-PH1. Population, housing units, area, and density: 2000 geography: New Mexico places, accessed July 2007 at http://factfinder.census.gov/servlet/GCTTable?_bm=y&-context=gct&-ds_name=DEC_2000_SF1_U&-mt_name=DEC_2000_SF1_U_GCTPH1_ST7&-tree_id=4001&-redoLog=true&-_caller=geoselect&-geo_id=04000US35&-format=ST-7|ST-7S&-_lang=en.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed June 12, 2009 at http://www.epa.gov/safewater/contaminants/.

U.S. Environmental Protection Agency, 2010, Radon in drinking water: questions and answers, accessed December 1, 2010 at http://water.epa.gov/lawsregs/rulesregs/sdwa/radon/basicinformation.cfm.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

Western Regional Climate Center, 2006a, Period of record monthly climate summary for Alamosa WSO AP, Colorado, accessed June 2007 at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?co0130.

Western Regional Climate Center, 2006b, Period of record monthly climate summary for Wolf Creek Pass 1E, Colorado accessed July 2007 at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?co9181.

Williams, R.S., Jr., and Hammond, S.E., 1989, Selected water-quality characteristics and flow of groundwater in the San Luis Basin, including the Conejos River Subbasin, Colorado and New Mexico: U.S. Geological Survey Water-Resources Investigations Report 89–4040, 43 p.

Wilson, E.M., 2000, Rio Grande consumptive uses and losses summary—Other (nonagricultural uses), Rio Grande Decision Support System—Final memorandum: Denver, Colo., Leonard Rice Consulting Water Engineers, Inc., 22 p. Available at ftp://dwrftp.state.co.us/cdss/csu/tm/RGDSScusumm.pdf.

Wilson, E.M., 2004, Historic crop consumptive use analysis, Rio Grande decision support system—Final report: Denver, Colo., Leonard Rice Consulting Water Engineers, Inc., variously paged. Available at ftp://dwrftp.state.co.us/cdss/csu/tm/RGDSScureport_6-2004.pdf.

Section 11.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the Middle Rio Grande Basin, New Mexico

By Laura M. Bexfield

Basin Overview

The Middle Rio Grande Basin is a 2,900-mi^2 alluvial basin extending along the Rio Grande in central New Mexico (fig. 1) that includes both geologic sources of natural contaminants and a long history of agricultural and urban land uses, but only local areas of substantial intrinsic groundwater susceptibility to contamination. The basin lies within the Rio Grande Rift, an area of crustal extension stretching from Colorado to Texas, and is hydraulically connected to other alluvial basins to the north and south. Despite being considered part of the Rio Grande aquifer system that extends along the Rift (Robson and Banta, 1995), the Middle Rio Grande Basin lies within the Basin and Range Physiographic Province (Fenneman, 1931) and has hydrogeologic characteristics similar to those in alluvial basins in the Basin and Range aquifer system of the southwestern United States. Altitudes range from about 4,700 ft, where the Rio Grande drains the basin at its southern end, to more than 6,200 ft in the foothills of the Jemez Mountains on the north and the Sandia and Monzano Mountains on the east. The Nacimiento Uplift, Rio Puerco fault zone, and Lucero Uplift are the primary boundary features on the west (fig. 1).

Most of the Middle Rio Grande Basin is categorized as having a semiarid climate, characterized by abundant sunshine, low humidity, and a high rate of evaporation that substantially exceeds the generally low rate of precipitation. Mean annual precipitation for 1914–2005 at Albuquerque was 8.7 in. (Western Regional Climate Center, 2006a), although mean annual precipitation for 1953–1979 at higher altitudes in the Sandia Mountains that border the basin to the east was 22.9 in. (Western Regional Climate Center, 2006b). Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 10.6 in. over the alluvial basin as a whole (McKinney and Anning, 2009). Most precipitation within the alluvial basin falls between July and October as a result of local, short duration, and high-intensity thunderstorms; winter storms of longer duration and lower intensity make a greater contribution to annual precipitation in the surrounding mountains. The mean monthly maximum temperature for 1914–2005 at Albuquerque was 47.2°F in January and 91.7°F in July (Western Regional Climate Center, 2006a).

The Middle Rio Grande Basin includes the Albuquerque metropolitan area (the most populous area in New Mexico), which grew about 20 percent between 1990 and 2000, from about 589,000 to 713,000 people (U.S. Census Bureau, 2001). Analysis of LandScan population data for 2000 (Oak Ridge National Laboratory, 2005) indicated a population of 756,000 for the alluvial basin as a whole (McKinney and Anning, 2009). Prior to substantial urbanization of the basin, land in upland areas outside of the historical Rio Grande flood plain (also referred to as the "inner valley") was almost exclusively rangeland; at 83 percent of the basin area, rangeland has remained the dominant land-use type according to the National Land Cover Database (NLCD) dataset for 2001 (U.S. Geological Survey, 2003). Irrigated agriculture is practiced throughout the Rio Grande flood plain, which is up to about 4.5 mi wide (fig. 1); irrigated cropland makes up just over 2 percent of land in the basin. Alfalfa was the most abundant crop type in 1993, followed by planted pasture (Kinkel, 1995, appendix 4). Population growth since about 1940 has led to urbanization of former agricultural land and rangeland in the Albuquerque area, resulting in urban turf grass being the second most abundant crop (in terms of planted acreage) in Bernalillo County in 1992 (Bartolino and Cole, 2002). As of 2001, the NLCD dataset classified only about 6 percent of land in the basin as urban.

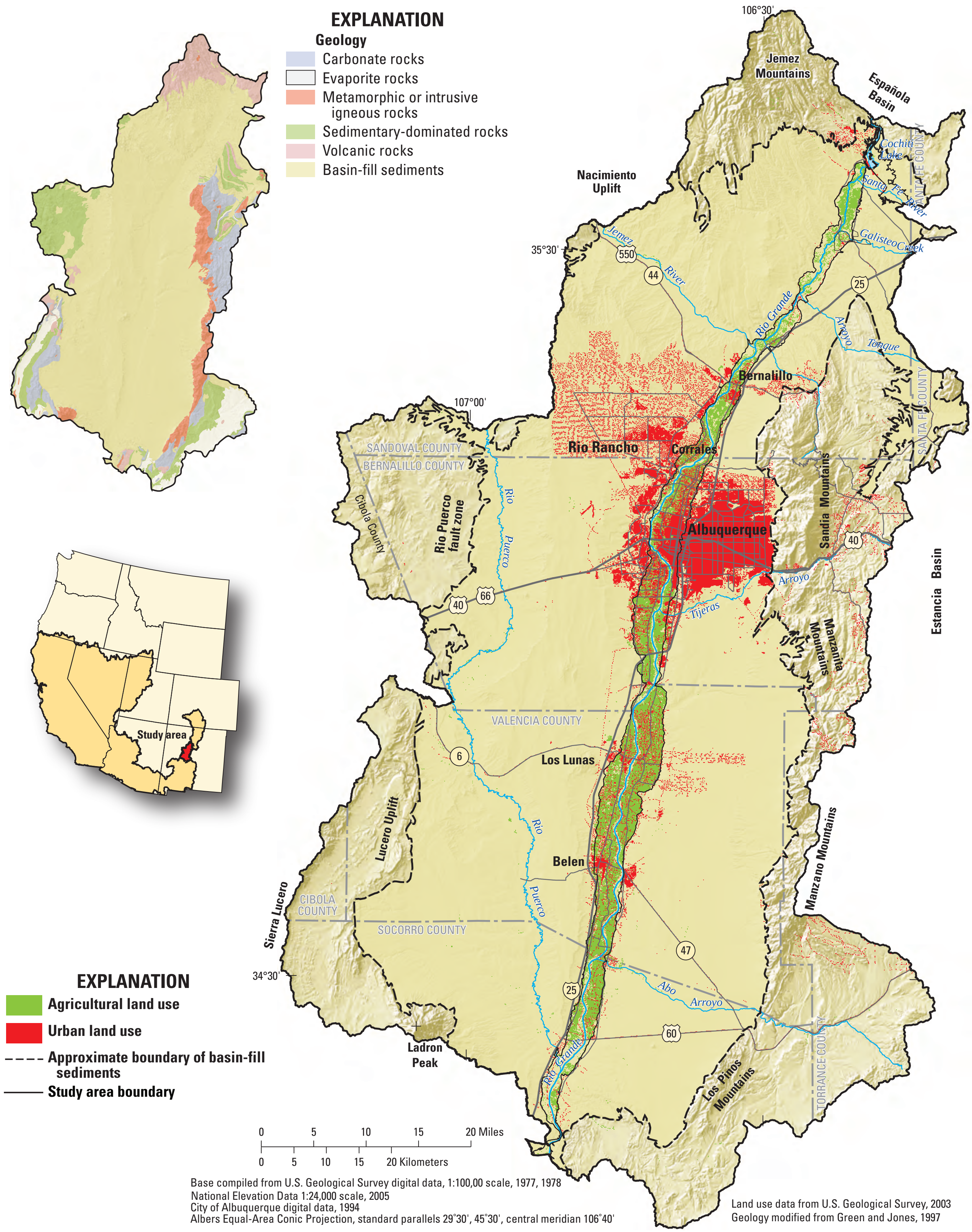


Figure 1. Physiography, land use, and generalized geology of the Middle Rio Grande Basin, New Mexico.

Despite expanding urbanization, irrigated agriculture continues to be the largest water user within the Middle Rio Grande Basin. Estimates of year-2000 water use by Wilson and others (2003) for the four counties that cover most of the basin (but also including some areas outside the basin; table 1) and by the U.S. Geological Survey (USGS) (http://water.usgs.gov/watuse/) for the area within the alluvial basin, indicate that nearly three-quarters of total combined surface-water and groundwater withdrawals were associated with irrigated agriculture; more than 90 percent of the water used for irrigated agriculture was surface water, primarily diverted from the Rio Grande and delivered to areas within the inner valley. About half of total water depletion was associated with irrigated agriculture (Wilson and others, 2003). Virtually all water demand for public supply has historically been met by groundwater withdrawals (Wilson and others, 2003; USGS water-use estimates, http://water.usgs.gov/watuse/). Combined, public supply, domestic uses, industry, and commercial uses represented about one-quarter of total withdrawals and one-third of total groundwater depletion in the major counties of the basin in 2000 (Wilson and others, 2003). Development of the water resources of the basin for agricultural and urban purposes has resulted in substantial changes to the groundwater and surface-water systems and how they interact.

Groundwater-quality issues identified in the Middle Rio Grande Basin include both naturally occurring contaminants and anthropogenic compounds. Concentrations of dissolved solids and arsenic across broad areas, particularly in the western part of the basin, exceed U.S. Environmental Protection Agency (USEPA) drinking-water standards (U.S. Environmental Protection Agency, 2009; each time a drinking-water standard or guideline is mentioned in this section, it denotes this same citation). As described later in this section, local occurrences of nitrate at concentrations greater than about 5 mg/L are believed to be natural in some areas, but to be associated with anthropogenic sources—particularly septic tanks—in others. Anthropogenic compounds that have been detected in groundwater of the basin include volatile organic compounds (VOCs) (particularly chlorinated solvents and petroleum hydrocarbons) and pesticides (particularly herbicides with urban uses). Most detections of these compounds have been in monitoring wells in or near the Rio Grande inner valley—an area that is intrinsically susceptible to groundwater contamination because of the presence of recharge and depths to groundwater generally less than about 30 ft (Anderholm, 1987)—and the detected concentrations have been below maximum concentrations specified in the USEPA's water-quality standards. In some cases, however, VOC detections near known chemical releases have resulted in the closure of private domestic wells and public-supply wells (U.S. Environmental Protection Agency, 2006).

Table 1. Water-use estimates for the Middle Rio Grande Basin, New Mexico, 2000.

[Counties included Bernalillo, Sandoval, Socorro, and Valencia. All values in acre-feet. Data from Wilson and others, 2003]

Water-use category	Surface-water withdrawal	Groundwater withdrawal	Total withdrawal	Total depletion
Public water supply	226	138,712	138,938	66,285
Domestic	0	12,576	12,576	12,576
Irrigated agriculture and livestock	429,096	47,581	476,677	146,970
Commercial, industrial, mining, and power generation	10	15,450	15,460	10,354
Reservoir evaporation	17,940	0	17,940	17,940
Total	**447,272**	**214,320**	**661,592**	**254,126**

Water Development History

By the time Spanish settlement had extended well into the Middle Rio Grande Basin in the early 1600s (Bartolino and Cole, 2002), most of the “major” pueblos of the area had already been in existence for hundreds of years (Scurlock, 1998). The pueblos had developed community irrigation ditches (or acequias) for farming in the inner valley, which the Spaniards imitated in developing their own irrigation systems (Bartolino and Cole, 2002). The intensity and extent of irrigated agriculture grew rapidly during the mid- to late-1800s with the arrival of large numbers of Anglo farmers and the introduction of improved farming practices (Scurlock, 1998). However, problems that included drought, sedimentation, salinization, and waterlogging reduced irrigated acreage after the early 1890s (Wozniak, 1996; Scurlock, 1998).

In the early- to mid-1900s, extensive efforts were undertaken throughout the Rio Grande Valley to protect and enhance the suitability of the inner valley for agriculture. A system of levees and jetty-jack works was used to confine the river to a single channel. The modern system of irrigation canals and groundwater drains also was constructed during this time (Thorn and others, 1993). The riverside and interior drains were put in place to lower the water table and allow reclamation of lands that had previously been waterlogged by canal leakage and irrigation. Reservoirs were constructed on the Rio Grande and its tributaries north of the basin as early as 1913 (Crawford and others, 1993). Cochiti Dam, at the north end of the Middle Rio Grande Basin, began operation in 1973 for flood control purposes.

Construction and operation of these man-made structures along the Rio Grande have substantially altered the configuration of the river, its seasonal discharge patterns, and its interaction with the groundwater system. The Rio Grande probably was once a perennial, braided river that migrated back and forth across the inner valley and had highly variable discharge reflecting seasonal snowmelt and storm events (Crawford and others, 1993). Currently (during 2006), the river does not deviate from its confined channel. However, surface water is diverted for irrigation through an extensive system of mostly unlined canals and applied to land throughout the inner valley between March and October of each year. The regulation of flows at dams upstream from the Middle Rio Grande Basin to sustain adequate discharge along the Rio Grande throughout the irrigation season results in a more uniform seasonal distribution of flow at Albuquerque (mean annual discharge 1,330 ft^3/s from USGS digital data for 1974–2005) than would be expected under “natural” conditions. Substantial irrigation diversions affect discharge along the river and reportedly have at times resulted in a dry stretch of channel downstream from the town of Bernalillo (Norman, 1968). The presence of the canal and drain systems within the Rio Grande inner valley has substantially increased the magnitude of fluxes between the surface-water and groundwater systems, as well as the area over which interaction between the two systems occurs. Riverside drains intercept leakage from the Rio Grande and eventually return that water to the river, along with irrigation water from the canal system and water captured by the interior drains as a result of seepage from canals and irrigated fields.

Utilization of the groundwater resources of the Middle Rio Grande Basin probably began when early settlers dug shallow wells for domestic use in unconsolidated river alluvium (Kelly, 1982). Albuquerque had about 1,307 residents in 1880, and within several years, the town had a public water-supply system that consisted of a few shallow wells located in the inner valley (Ground-Water Science, Inc., 1995). Albuquerque’s population increased steadily to about 35,500 in 1940, and then climbed rapidly to about 200,000 people in 1960 (fig. 2) (Ground-Water Science, Inc., 1995). During this period of rapid growth, Albuquerque met its increasing water demand with an expanding network of public-supply wells, including several located in upland areas, which were becoming extensively urbanized. In 1989, the city had about 90 public-supply wells and pumped 127,000 acre-ft of groundwater (City of Albuquerque files) to supply a population of almost 385,000. Although the population had increased to nearly 450,000 city residents in 2000, Albuquerque’s water demand declined—ranging from 100,000 to 114,000 acre-ft/yr during 1997–2004 (City of Albuquerque files). The decline in water use resulted primarily from conservation efforts initiated after studies indicated that groundwater resources of the basin were more limited than previously believed (City of Albuquerque, 2010). Groundwater withdrawals by domestic, commercial, and industrial users in the basin have continued to increase overall (Wilson, 1992; Wilson and Lucero, 1997; Wilson and others, 2003).

Sustained groundwater withdrawals from the Middle Rio Grande Basin have resulted in extensive water-level declines, which exceed 120 ft in eastern Albuquerque (Bexfield and Anderholm, 2002a). These declines have substantially altered groundwater flow directions (as discussed in more detail in a later section), reduced flows in the Rio Grande by inducing additional infiltration (McAda and Barroll, 2002), and decreased the amount of evapotranspiration within the Rio Grande inner valley (McAda and Barroll, 2002). Declines in water levels also have increased the cost of groundwater pumping and, if allowed to continue, have the potential to result in widespread land-surface subsidence and(or) degradation of groundwater quality in the future (Bexfield and others, 2004).

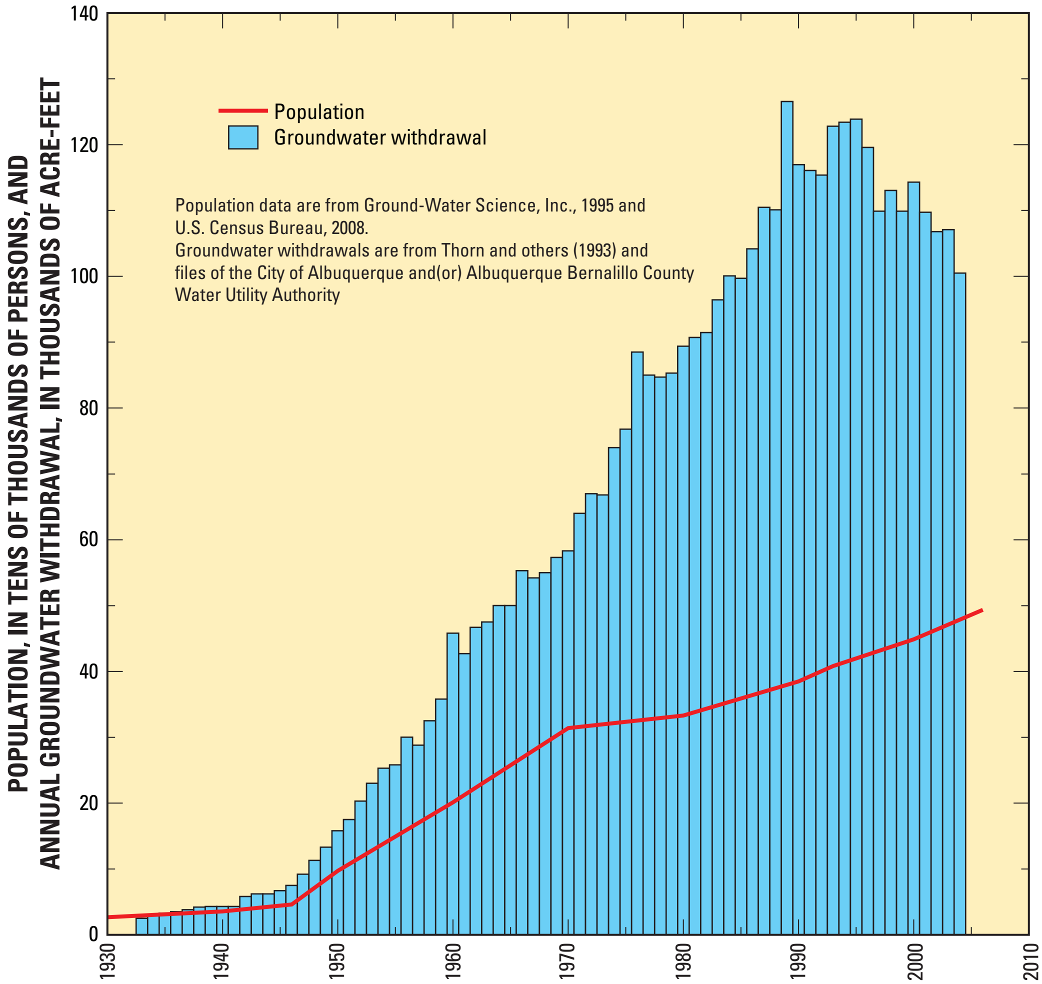


Figure 2. Albuquerque population (1930–2006) and groundwater withdrawals (1933–2005).

Recognition of existing and potential future problems resulting from continued development of groundwater resources at recent levels prompted the City of Albuquerque to adopt a new water-supply strategy in 1997 (City of Albuquerque, 2010). The City of Albuquerque owns rights to about 48,200 acre-ft/yr of surface water imported into the Rio Grande Basin from the Colorado River Basin by the San Juan Chama Transmountain Diversion Project (completed in 1971) and about 23,000 acre-ft/yr of native Rio Grande water. Under the new strategy, direct use of this surface water for public supply began in December 2008. Groundwater will still be used, but primarily to supplement supplies during periods of drought and months of high demand (typically June through September). Because the City of Albuquerque has historically been responsible for a large portion of groundwater withdrawals from the basin, as evidenced by the estimation that it was responsible for just over half of total groundwater withdrawals from the basin in 2000 (City of Albuquerque and the New Mexico Office of the State Engineer files), this change in water-supply strategy is expected to have substantial effects on the groundwater and surface-water systems, including rises in groundwater levels and decreases in the infiltration of river water (Bexfield and McAda, 2003). Additional strategies that are being implemented to reduce groundwater withdrawals include the use of treated municipal wastewater, recycled industrial wastewater, and nonpotable surface water to irrigate urban turf areas (Albuquerque Bernalillo County Water Utility Authority, 2010).

Hydrogeology

The Middle Rio Grande Basin lies along the Rio Grande Rift, which is a generally north-south trending area of Cenozoic crustal extension. Successive episodes of extension starting about 32 million years ago (Russell and Snelson, 1990) caused large blocks of crust to drop down relative to adjacent areas, forming a series of structural and physiographic basins, many of which are hydraulically connected. The Middle Rio Grande Basin includes three subbasins that are separated by bedrock structural highs and contain alluvial fill as much as about 15,000 ft thick (Grauch and others, 1999). Bedrock benches on the east and west bound the deeper parts of the basin. In addition to major faults that juxtapose alluvium and bedrock along uplifts and benches near the basin margins, numerous other primarily north-south trending faults have caused offsets within the alluvial fill (Grauch and others, 2001; Connell, 2006). The Sandia, Manzanita, Manzano, and Los Pinos Mountains on the east, the Ladron Mountains on the southwest, and the Nacimiento Uplift on the northwest are composed of Precambrian plutonic and metamorphic rocks, generally overlain by Paleozoic and(or) Mesozoic sedimentary rocks (Hawley and Haase, 1992; Hawley and others, 1995) (fig. 1). The Jemez Mountains on the north are a major Cenozoic volcanic center. Primarily Paleozoic and Mesozoic sedimentary rocks border the basin on the west.

The alluvial fill of the Middle Rio Grande Basin is composed primarily of the unconsolidated to moderately consolidated Santa Fe Group deposits of late Oligocene to middle Pleistocene age, which overlie lower and middle Tertiary rocks in the central part of the basin, and Mesozoic, Paleozoic, and Precambrian rocks near the basin margins (McAda and Barroll, 2002). Post-Santa Fe Group valley- and basin-fill deposits of Pleistocene to Holocene age typically are in hydraulic connection with the Santa Fe Group deposits; in combination, these deposits form the Santa Fe Group aquifer system (Thorn and others, 1993). The sediments in the basin consist generally of sand, gravel, silt, and clay that were deposited in fluvial, lacustrine, or piedmont-slope environments.

Hawley and Haase (1992) defined broad lower, middle, and upper parts of the Santa Fe Group on the basis of both the timing and the environment of deposition. Sediments of the lower Santa Fe Group, which may be as much as 3,500-ft thick in places, include extensive basin-floor playa deposits with low hydraulic conductivity. The middle Santa Fe Group ranges from about 250 to 9,000-ft thick and consists largely of basin-floor fluvial deposits in the north and fine-grained playa deposits in the south. The upper unit generally is less than about 1,000-ft thick, except in some areas near Albuquerque, and was deposited during development of the ancestral Rio Grande system (about 1 to 5 million years ago). The axial-channel deposits of this high-energy, fluvial system include thick zones of well-sorted sand and gravel that constitute the most productive aquifer materials in the basin. Most public-supply wells in the study area are completed in the upper and(or) middle units east of the Rio Grande and in the middle and(or) lower units west of the river. Post-Santa Fe Group valley-fill sediments generally are less than about 130-ft thick. These sediments, in which the estimates for hydraulic conductivity vary widely, provide a connection between the surface-water system and the underlying Santa Fe Group deposits.

Conceptual Understanding of the Groundwater System

Groundwater within the Santa Fe Group aquifer system of the Middle Rio Grande Basin generally is unconfined, but is semiconfined at depth. Depths to water range from a few feet near the Rio Grande to more than 700 ft beneath upland areas both east and west of the river (and at least 900 ft beneath parts of Rio Rancho). Transmissivity estimates for the aquifer system have ranged widely because of variations in both aquifer thickness and hydraulic conductivity across the basin, but estimates from aquifer tests (mostly in Albuquerque public-supply wells) generally fall between about 3,000 and 70,000 ft^2/d (Thorn and others, 1993). These values were used by Thorn and others (1993) to estimate horizontal hydraulic conductivities as ranging from about 4 to 150 ft/d; in their groundwater-flow model of the basin, McAda and Barroll (2002) used hydraulic conductivity values of 0.05 to 60 ft/d. The basin-wide occurrence of interbedded fine- and coarse-grained sediments suggests a relatively high degree of anisotropy, that is, a large ratio of horizontal to vertical hydraulic conductivity. Through calibration, McAda and Barroll (2002) selected a ratio of 150:1 for their model (compared with ratios of 80:1 to 1,000:1 used in previous models).

Water Budget

Water budgets have been developed for the Middle Rio Grande Basin in association with groundwater-flow models. The McAda and Barroll (2002) model incorporated estimates of various budget components resulting from the most recent multiagency study of hydrogeology in the basin, during 1995–2001; the water budget from this model (table 2) provides the basis for most of the discussion in this section. Individual components of recharge and discharge are illustrated in the conceptual diagrams of regional groundwater flow in figures 3*A* and 3*B*.

Table 2. Water-budget components for the Middle Rio Grande Basin, New Mexico, under predevelopment and modern conditions, as simulated by the McAda and Barroll (2002) groundwater-flow model.

[All values are in acre-feet per year and are rounded to the nearest thousand. Small differences in total recharge and total discharge that are not accounted for by change in aquifer storage are the result of rounding and(or) model error. Percentages of water-budget components are illustrated on figure 3]

	Predevelopment conditions (steady state)	Modern conditions (year ending October 1999)	Change from predevelopment to modern conditions
Budget component		**Net recharge**	
Mountain-front recharge[1]	12,000	12,000	0
Tributary recharge	9,000	9,000	0
Subsurface recharge[2]	31,000	31,000	0
Canal seepage	0	90,000	90,000
Crop-irrigation seepage	0	35,000	35,000
Rio Grande and Cochiti Lake	63,000	316,000	253,000
Jemez River and Jemez Canyon Reservoir	15,000	17,000	2,000
Septic-field seepage	0	4,000	4,000
Total recharge	**130,000**	**514,000**	**384,000**
Budget component		**Net discharge**	
Riverside drains	0	208,000	208,000
Interior drains	0	133,000	133,000
Groundwater withdrawal	0	150,000	150,000
Riparian evapotranspiration	129,000	84,000	-45,000
Total discharge	**129,000**	**575,000**	**446,000**
Change in aquifer storage	0	-60,000	-60,000

[1]As defined for the McAda and Barroll (2002) model, the mountain-front recharge budget component does not include mountain-front recharge along the Jemez Mountains on the north because mountain-front recharge could not be distinquished from subsurface recharge through the mountain block in this area.

[2]As defined for the McAda and Barroll (2002) model, the subsurface recharge budget component includes groundwater inflow from adjacent basins on the west and north, groundwater inflow from mountain blocks on the east, and combined subsurface and mountain-front recharge along the Jemez Mountains on the north.

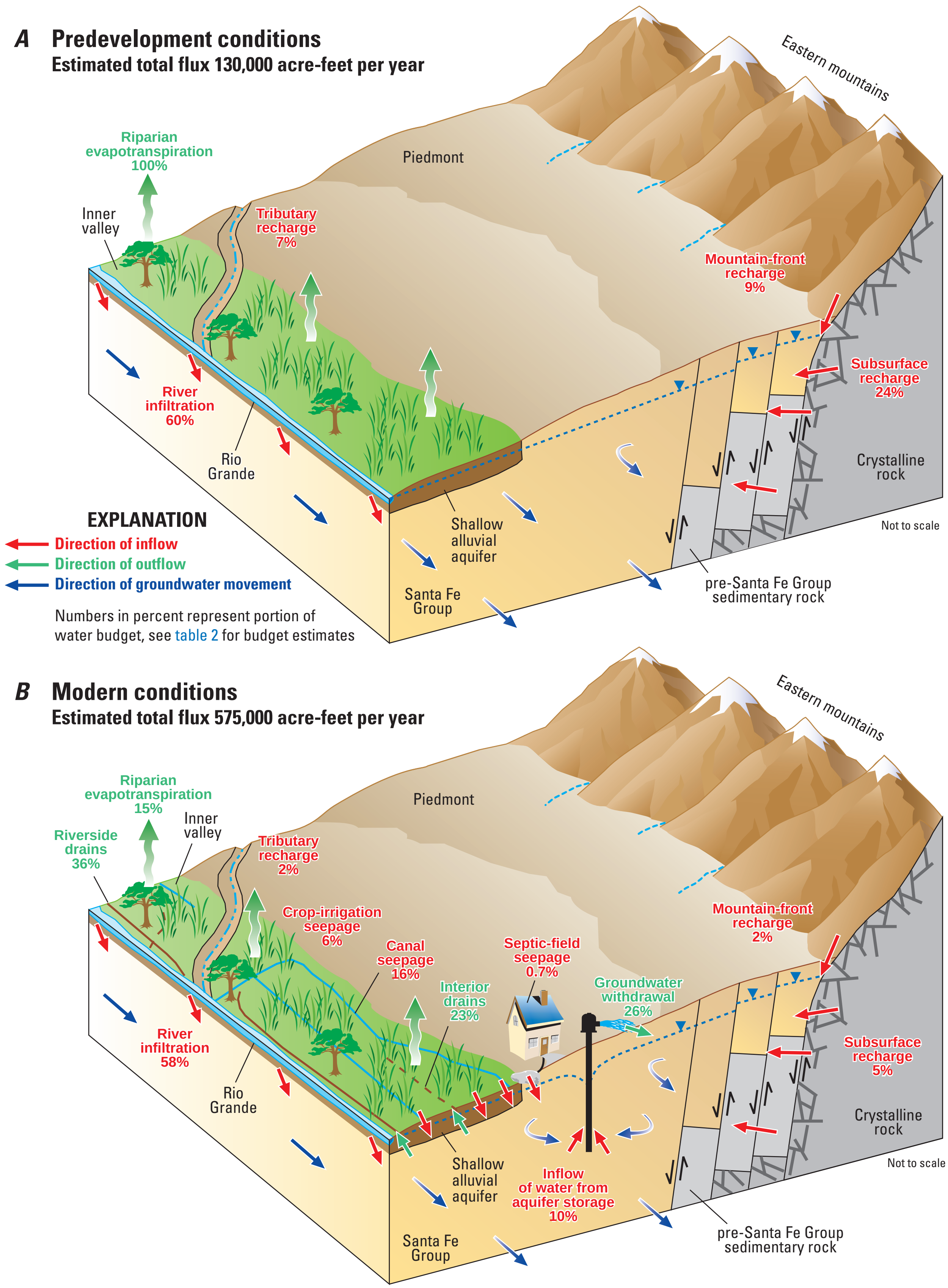


Figure 3. Generalized diagrams for the Middle Rio Grande Basin, New Mexico, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

As a result of low precipitation rates combined with high evaporation rates and generally large depths to groundwater, areal recharge to the Santa Fe Group aquifer system of the Middle Rio Grande Basin from precipitation is believed to be minor (Anderholm, 1987; 1988). Instead, groundwater recharge occurs primarily along surface-water features and at the basin boundaries (fig. 3). Using the chloride-balance method, Anderholm (2001) calculated mountain-front recharge along the entire eastern margin of the basin to total about 11,000 acre-ft/yr. The McAda and Barroll (2002) model uses a value of about 12,000 acre-ft/yr for the basin (table 2), excluding areas along the Jemez Mountains to the north; this value is about 9 percent of the total simulated recharge of 130,000 acre-ft/yr under steady-state (that is, predevelopment) conditions. Subsurface recharge occurring as groundwater inflow from adjacent basins to the west and north (through sedimentary rocks and alluvial fill), subsurface recharge occurring as groundwater inflow from mountain blocks to the east, and combined subsurface and mountain-front recharge occurring along the Jemez Mountains to the north (fig. 3) has been estimated through groundwater-flow modeling, using supporting evidence from studies of hydrogeology (Smith and Kuhle, 1998; Grant, 1999) and groundwater ages (Sanford and others, 2004a, 2004b). McAda and Barroll (2002) use a total of about 31,000 acre-ft/yr of subsurface recharge for the basin (including combined subsurface and mountain-front recharge along the Jemez Mountains), or about 24 percent of total simulated recharge under steady-state conditions.

Within the Middle Rio Grande Basin, most recharge to the aquifer system results from infiltration of surface water (shown by the red arrows in fig. 3; table 2), which occurs along the Rio Grande and its main tributary, the Jemez River. By comparison, tributary recharge along the Rio Puerco in the west, the Rio Salado in the south, and streams and arroyos entering the basin from the east (which generally do not contain persistent flow more than a few hundred feet from the mountain front) is small. Tributary recharge assigned in the McAda and Barroll (2002) model totals about 9,000 acre-ft/yr, or about 7 percent of total simulated recharge under steady-state conditions. The Rio Grande, which is in hydraulic connection with the Santa Fe Group aquifer system, is believed to lose water along most of its length within the basin. The McAda and Barroll (2002) model simulated infiltration of Rio Grande streamflow to the aquifer system under steady-state conditions to be about 63,000 acre-ft/yr, or about 48 percent of total steady-state recharge. Along the Jemez River (which is in hydraulic connection with the aquifer system through most of its length within the basin), these losses are simulated to be about 15,000 acre-ft/yr under steady-state conditions, or about 12 percent of total steady-state recharge.

Since urbanization and the development of large-scale irrigation systems in the Middle Rio Grande Basin, fluxes of water through the aquifer system have increased substantially, as illustrated by the simulated water budget of McAda and Barroll (2002) for the year ending in October 1999 (table 2). Infiltration of water to the aquifer system in the Rio Grande inner valley is about seven times larger than it was under predevelopment conditions and is spread over a much larger area of the inner valley. The model simulates seepage from irrigation canals (including some along the Jemez River) as contributing about 90,000 acre-ft/yr of water to the aquifer system. By applying an estimated average recharge rate of about 0.5 acre-ft/acre to all irrigated cropland in the model, recharge through crop-irrigation seepage is estimated to total about 35,000 acre-ft/yr. Given the declines in groundwater levels as a result of withdrawals for public supply, along with filling of the Cochiti Reservoir starting in 1973, infiltration along the Rio Grande is simulated to be 316,000 acre-ft/yr, or about five times the infiltration simulated under steady-state conditions. An additional source of recharge resulting from urbanization is septic-field seepage, which occurs both within and outside the Rio Grande inner valley and is estimated by McAda and Barroll (2002) to total about 4,000 acre-ft/yr for the year ending in October 1999, based on census data and an estimated seepage rate of 60 gallons per day per person. Additional sources of recharge outside the inner valley that have likely resulted from urbanization, but that would be expected to occur only locally and are not represented in the McAda and Barroll (2002) model include seepage from sewer and water-distribution lines and from turf irrigation.

Under predevelopment conditions, water was discharged from the aquifer system primarily through evapotranspiration from riparian vegetation and wetlands in the Rio Grande inner valley (Kernodle and others, 1995). Groundwater withdrawals for public supply and construction of an extensive groundwater drainage system in the inner valley have lowered the water table and resulted in reduced evapotranspiration from native riparian vegetation and wetlands (about 84,000 acre-ft for the year ending in October 1999 in comparison with about 129,000 acre-ft/yr under steady-state conditions, as simulated by McAda and Barroll [2002]). The largest component of outflow from the aquifer system currently is discharge to the groundwater drain system ("Riverside drains" and "Interior drains" in table 2), which McAda and Barroll (2002) simulated to total about 341,000 acre-ft/yr (table 2). Slightly more than 60 percent of this discharge was to the riverside drains, and the remainder discharged to interior drains located farther from the Rio Grande. Most of the groundwater discharging to the drain system is water that has moved through the shallow system after infiltrating from the Rio Grande or seeping from irrigation canals and irrigated fields (McAda and Barroll, 2002), although groundwater from the deep regional system also discharges to the drains.

Some groundwater discharges from the aquifer system by means of subsurface flow through alluvial fill to the Socorro Basin on the south, but this discharge is considered negligible relative to other budget components (Sanford and others, 2004a). Some groundwater also may discharge directly to the Rio Grande in individual reaches, particularly in the northern part of the basin (Trainer and others, 2000). Groundwater withdrawals currently are a major component of the water budget, discharging an estimated 150,000 acre-ft from the aquifer system during the year ending in October 1999 (table 2) and resulting in the removal of water from aquifer storage.

Groundwater Flow

Maps of predevelopment (generally, pre-1960) groundwater levels in the study area (Meeks, 1949; Bjorklund and Maxwell, 1961; Titus, 1960; Bexfield and Anderholm, 2000) indicate that the principal direction of groundwater flow was from north to south through the center of basin, with greater components of east-to-west flow near the basin margins (fig. 4). This general flow pattern reflects the areal distribution of groundwater recharge and discharge (fig. 3). Predevelopment water-level maps indicate the presence of depressions—or “troughs”—in the water-level surface both east and west of the Rio Grande. The origin of these troughs has not been conclusively determined, but McAda and Barroll (2002) suggest the presence of high-permeability pathways, horizontal anisotropy, and(or) faults acting as flow barriers as possible explanations for their presence. Plummer and others (2004a, 2004b, 2004c) and Sanford and others (2004a, 2004b) hypothesized that the trough west of the Rio Grande may be a transient feature that reflects changes in the quantity and spatial distribution of recharge through time.

Large and extensive water-level declines caused by sustained groundwater withdrawals for public supply have substantially altered the direction of groundwater flow in the Albuquerque metropolitan area (Bexfield and Anderholm, 2002a) (fig. 5). Water-level declines since predevelopment in the production zone (the range of aquifer depths from which most withdrawals by public-supply wells occur—typically from about 200 to 900 ft or more below the water table) have exceeded 100 ft across more than 15 mi^2 east of the Rio Grande and 80 ft across smaller areas west of the Rio Grande. Consequently, groundwater currently flows toward the major pumping centers from all directions (fig. 5), and the magnitudes of horizontal hydraulic gradients in the Albuquerque area have increased (figs. 4 and 5). Water-level declines in the aquifer also have induced additional inflow from the surface-water system compared to that under predevelopment conditions. In most areas where water-level declines have occurred, the saturated thickness of the aquifer has not been substantially affected because of the large thickness of Santa Fe Group sediments.

Water-level data from deep piezometer nests across the Albuquerque area indicate that vertical hydraulic gradients generally are downward in the Rio Grande inner valley and areas to the west, and upward in areas east of the inner valley, except in proximity to the mountain front (Bexfield and Anderholm, 2002b). These deep nests typically include three piezometers that are screened across relatively short depth intervals near the water table (shallow), near the middle of the production zone (middle), and near the bottom of the production zone (deep). Using data from continuous water-level monitors for 1997–99, Bexfield and Anderholm (2002b) illustrated that water levels in the middle and deep zones tended to show fairly substantial changes (exceeding 20 ft in places) in response to seasonal variations in groundwater withdrawals, whereas water levels at the water table (where the storage coefficient is largest) generally showed much smaller seasonal changes. Similar patterns can be seen in water-level data for 2001–04 (fig. 6). Groundwater withdrawals, therefore, tend to increase the magnitude of—and, in some cases, change the direction of—vertical hydraulic gradients. The magnitudes of typical vertical gradients also vary among locations, probably reflecting local variations in the degree of vertical hydraulic connection and in the intensity of groundwater withdrawals. In one piezometer nest (at Garfield Park in the Rio Grande inner valley; fig. 6), water-level changes at the water table appear to be affected by land use—in particular, seasonal operation of the irrigation system (Bexfield and Anderholm, 2002b).

The age of most groundwater in the Santa Fe Group aquifer system of the Middle Rio Grande Basin is on the order of thousands of years (fig. 7), as estimated using carbon-14 dating methods (Plummer and others, 2004a, 2004b, 2004c) for water from wells generally screened within about the upper 1,000 ft of the aquifer. (See Section 1 of this report for a discussion of groundwater age and environmental tracers.) Groundwater less than 2,000 years in age typically is found only near known areas of recharge—primarily, basin margins and surface-water features. Chlorofluorocarbons and tritium—indicators of the presence of at least a small fraction of young (post-1950s) recharge—were relatively common at shallow depths within the Rio Grande inner valley, along mountain fronts, and near arroyos (Plummer and others, 2004a). Chlorofluorocarbons and tritium were detected in some samples collected from near the water table beneath upland areas, indicating the potential presence of recharge sources in these areas that have not been well characterized. Overall spatial patterns in groundwater ages indicate that the residence time of most of the groundwater in the basin exceeds 10,000 years (fig. 7), thereby illustrating that the flux of water through the basin is relatively small given the volume of aquifer.

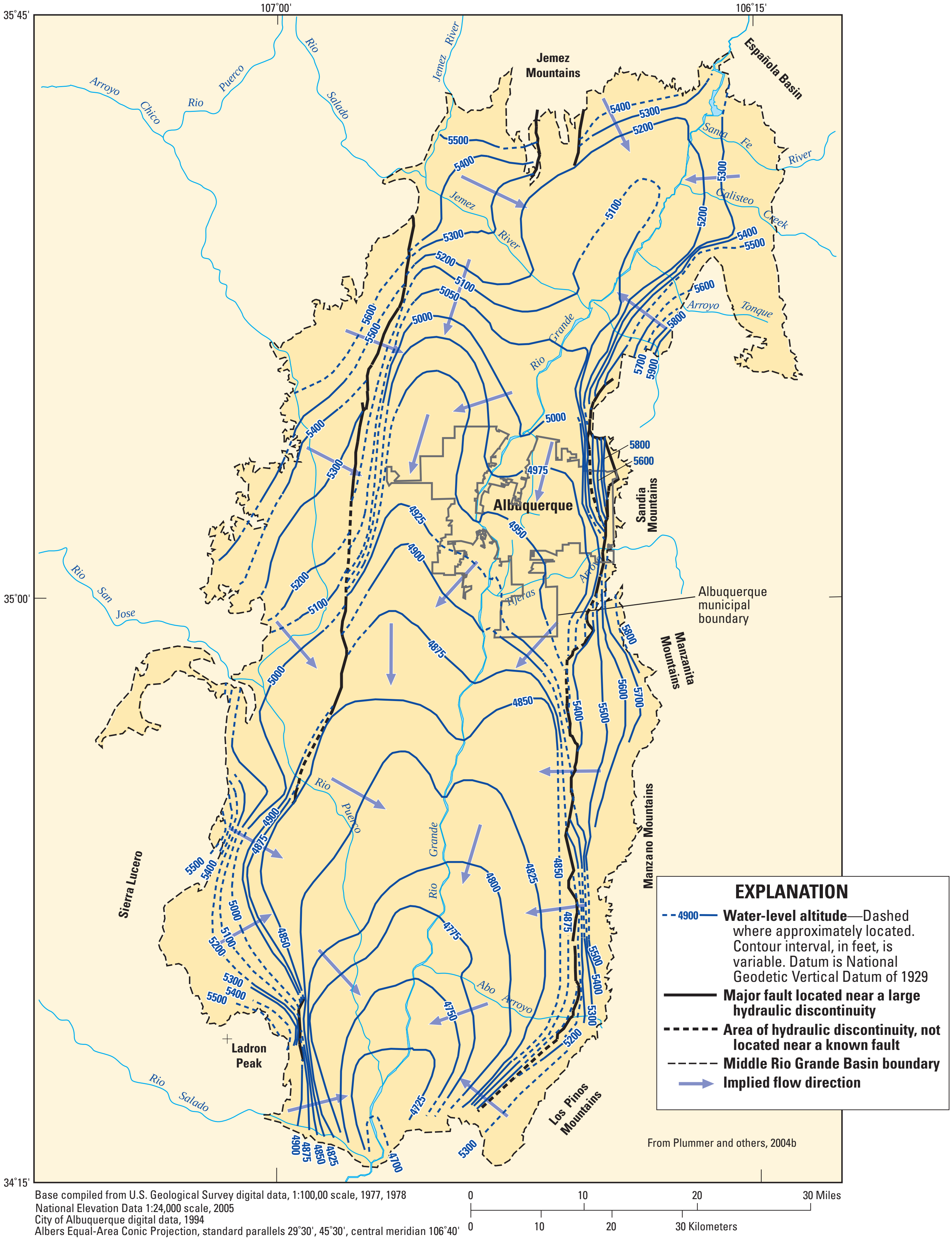


Figure 4. Groundwater levels representing predevelopment conditions in the Middle Rio Grande Basin, New Mexico.

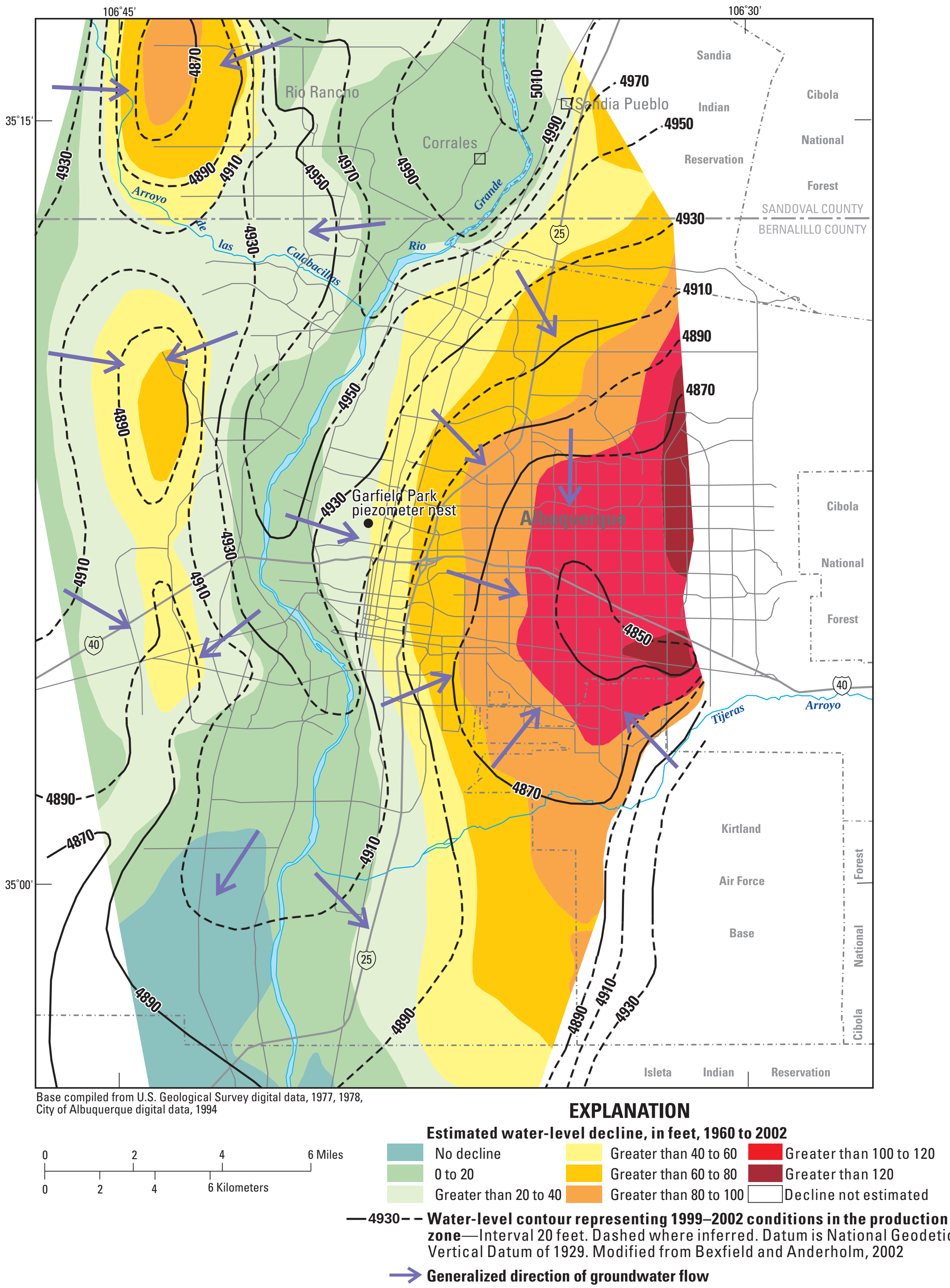


Figure 5. Water levels representing 1999–2002 conditions in the production zone, and estimated water-level declines, 1960 to 2002 , in the Albuquerque area, New Mexico.

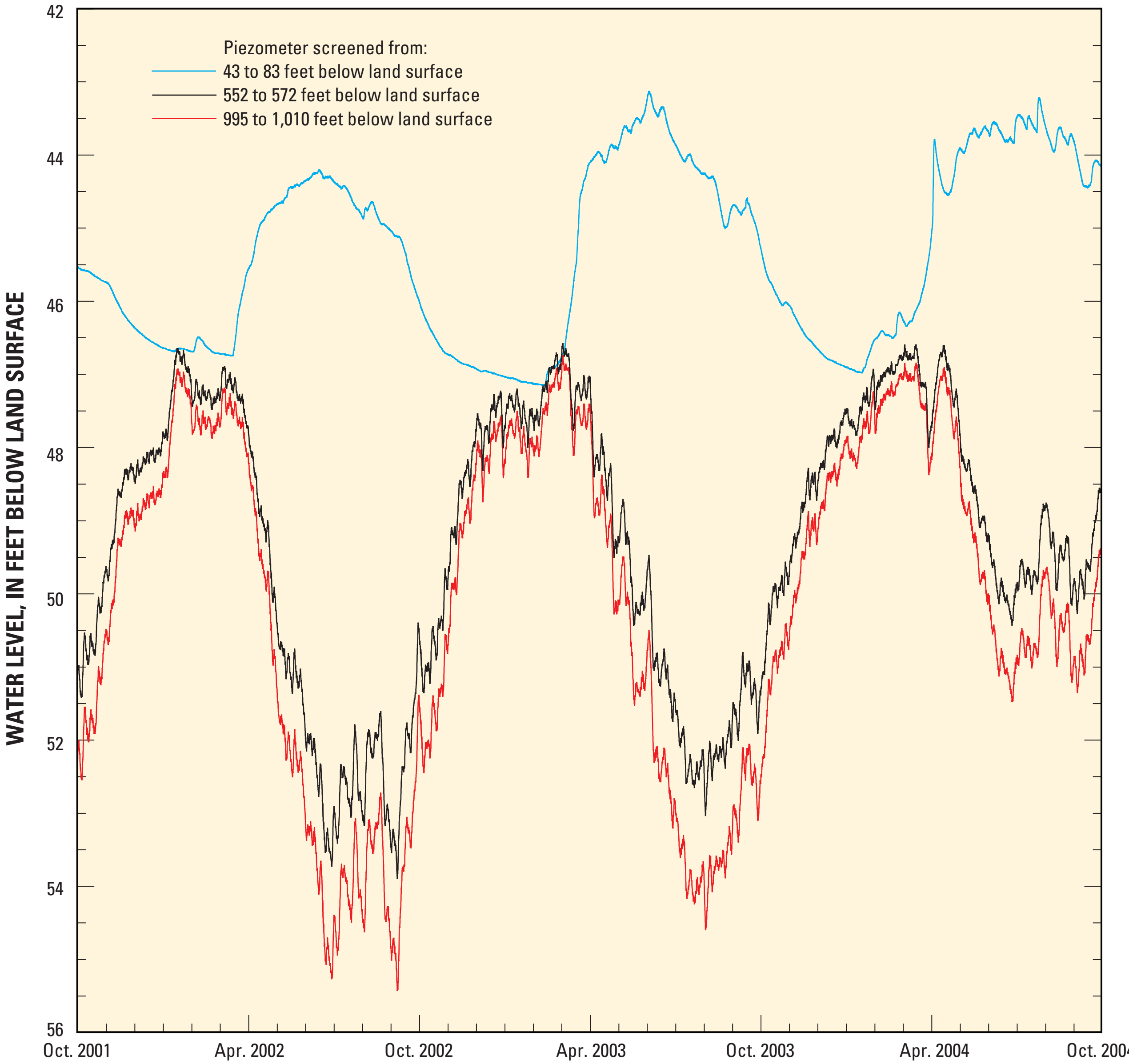


Figure 6. Water levels in the Garfield Park piezometer nest in the Rio Grande inner valley, Albuquerque, New Mexico. Location is shown on figure 5.

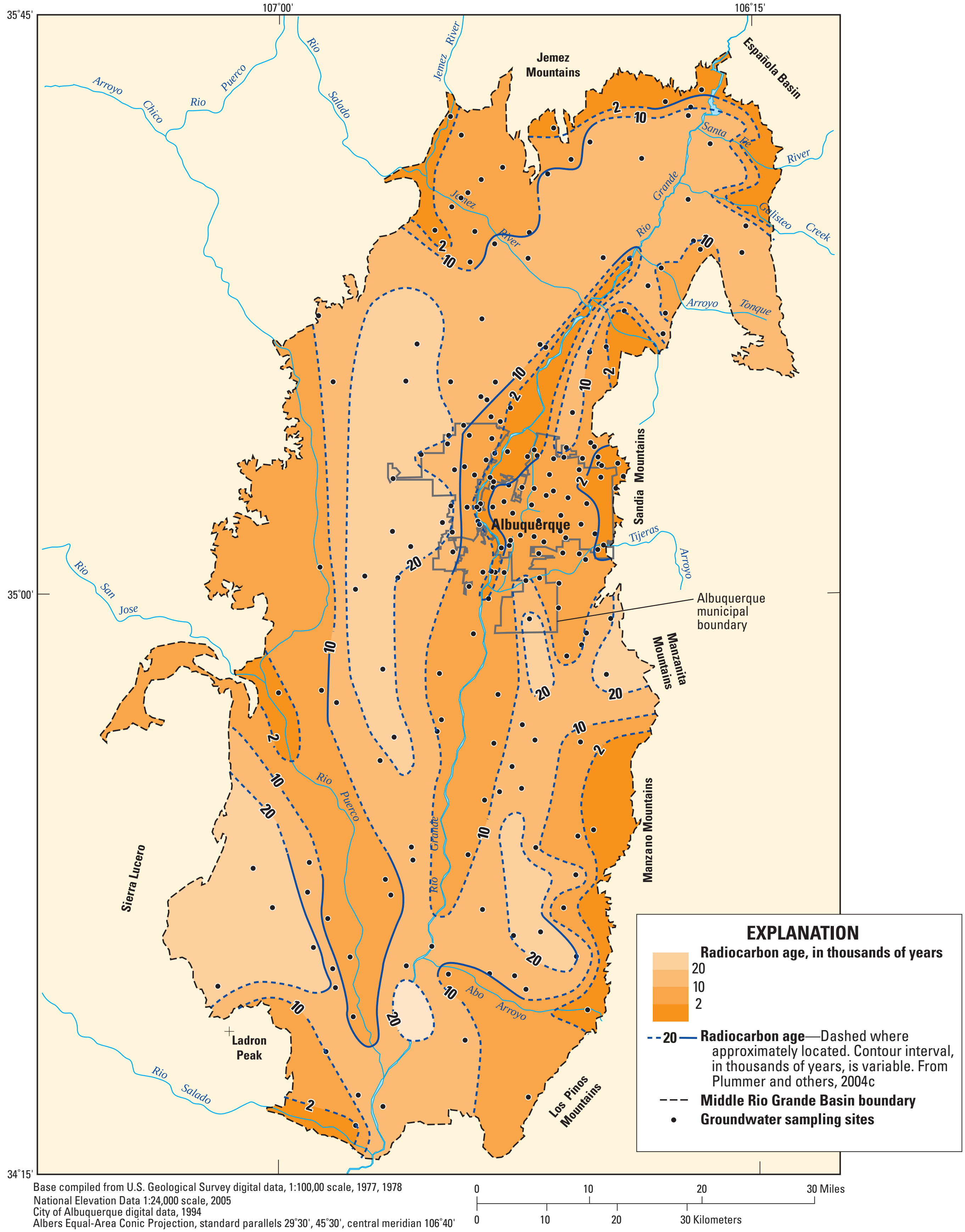


Figure 7. Estimated ages of groundwater in the Santa Fe Group aquifer system of the Middle Rio Grande Basin, New Mexico.

Effects of Natural and Human Factors on Groundwater Quality

Because sediments of the Santa Fe Group aquifer system are relatively unreactive, groundwater quality in the Middle Rio Grande Basin is determined primarily by the source and composition of recharge rather than by geochemical reactions and other processes occurring within the aquifer (Plummer and others, 2004a). Studies by Anderholm (1988), Logan (1990), Bexfield and Anderholm (2002b), and Plummer and others (2004a, 2004b) have illustrated spatial patterns in water chemistry across the Albuquerque area and(or) the basin. Based on primarily patterns in the hydrochemical data from hundreds of wells of various types (public-supply, monitoring, domestic, and other) that were generally screened within about the upper 1,000 ft of the aquifer, Plummer and others (2004a, 2004b) delineated 13 individual hydrochemical zones throughout the basin (fig. 8 and table 3), each with relatively homogeneous groundwater chemistry that is distinct from that in the other zones. Twelve zones represent individual sources of recharge to the basin and are used to facilitate this discussion of water chemistry within the basin. The other zone represents the area in which groundwater from upgradient zones and from depth within the aquifer system converges before discharging to the inner valley or the Socorro Basin to the south.

General Water-Quality Characteristics and Natural Factors

The Northern Mountain Front and Eastern Mountain Front zones of Plummer and others (2004a, 2004b) delineate areas of the basin where groundwater recharges primarily through relatively high-elevation mountain-front recharge processes (shallow subsurface inflow and infiltration through mountain stream channels). Groundwater in these zones has among the smallest dissolved-solids concentrations found in the basin, as indicated by specific-conductance values that typically are less than 400 µS/cm (fig. 9 and table 3). The groundwater chemistry is generally reflective of the chemistry of local precipitation that has undergone some evapotranspiration during recharge (Plummer and others, 2004a). Dissolved-oxygen concentrations indicate that the groundwater is well oxidized; nitrate also is present in most wells, but generally at concentrations less than 1 mg/L. In most areas of the Middle Rio Grande Basin, groundwater continues to be well oxidized even far from sources of recharge and at depths of several hundred feet (figs. 10*A* and 10*B*), probably because of a general paucity of organic carbon in aquifer materials (Plummer and others, 2004a). Arsenic concentrations in the Northern and Eastern Mountain Front zones generally are 3 µg/L or less (fig. 11 and table 3), but locally approach or exceed the drinking-water standard of 10 µg/L. In the Northern Mountain Front zone, most elevated concentrations of arsenic probably are associated with volcanic sources in the Jemez Mountains. In the Eastern Mountain Front zone (and some other areas of the basin), elevated concentrations of arsenic typically are associated with old, deep, mineralized water that upwells along major structural features (Bexfield and Plummer, 2003; Plummer and others, 2004a).

In the Northwestern zone, which delineates groundwater believed to have recharged at relatively low elevations along the Jemez Mountain Front (Plummer and others, 2004a), concentrations of dissolved solids are slightly larger than those found in the Northern Mountain Front zone (table 3). Concentrations of nitrate also are larger, and commonly approach or exceed 5 mg/L. Because there is relatively little human activity in the area, and the age of the groundwater is generally greater than 7,000 years, these concentrations of nitrate likely result from natural sources in precipitation and(or) soils (Plummer and others, 2004a). Concentrations of arsenic commonly approach or exceed 10 µg/L (fig. 11) and probably are primarily associated with volcanism in the Jemez Mountains. Groundwater chemistry in the small Southwestern Mountain Front zone also represents recharge by relatively low-elevation mountain-front processes.

The West Central zone extends southward from the area of the Jemez Mountains through much of the western half of the Middle Rio Grande Basin (fig. 8) and extends at depth beneath adjacent hydrochemical zones to the east. The West Central zone represents relatively old groundwater inflow that entered the basin at depth along the northern margin. Despite the long residence times of the groundwater, concentrations of dissolved solids are moderate throughout much of this zone (specific-conductance values generally are less than 600 µS/cm) (fig. 9 and table 3) and exceed the USEPA's non-enforceable guideline of 500 mg/L in only some wells. Values of pH exceed 8 across broad areas of the West Central zone. The groundwater is generally well oxidized (fig. 10) and contains nitrate at concentrations below 2 mg/L; however, dissolved oxygen and nitrate are below detection in some wells (Plummer and others, 2004a). Groundwater of the West Central zone commonly has concentrations of arsenic greater than the USEPA drinking-water standard (fig. 11); these large concentrations generally are associated with volcanism in the Jemez Mountains and with desorption from metal oxides, especially in areas where pH exceeds about 8.5 (Bexfield and Plummer, 2003; Plummer and others, 2004a). In one well sampled by Plummer and others (2004a), the standard of 30 µg/L for uranium was exceeded.

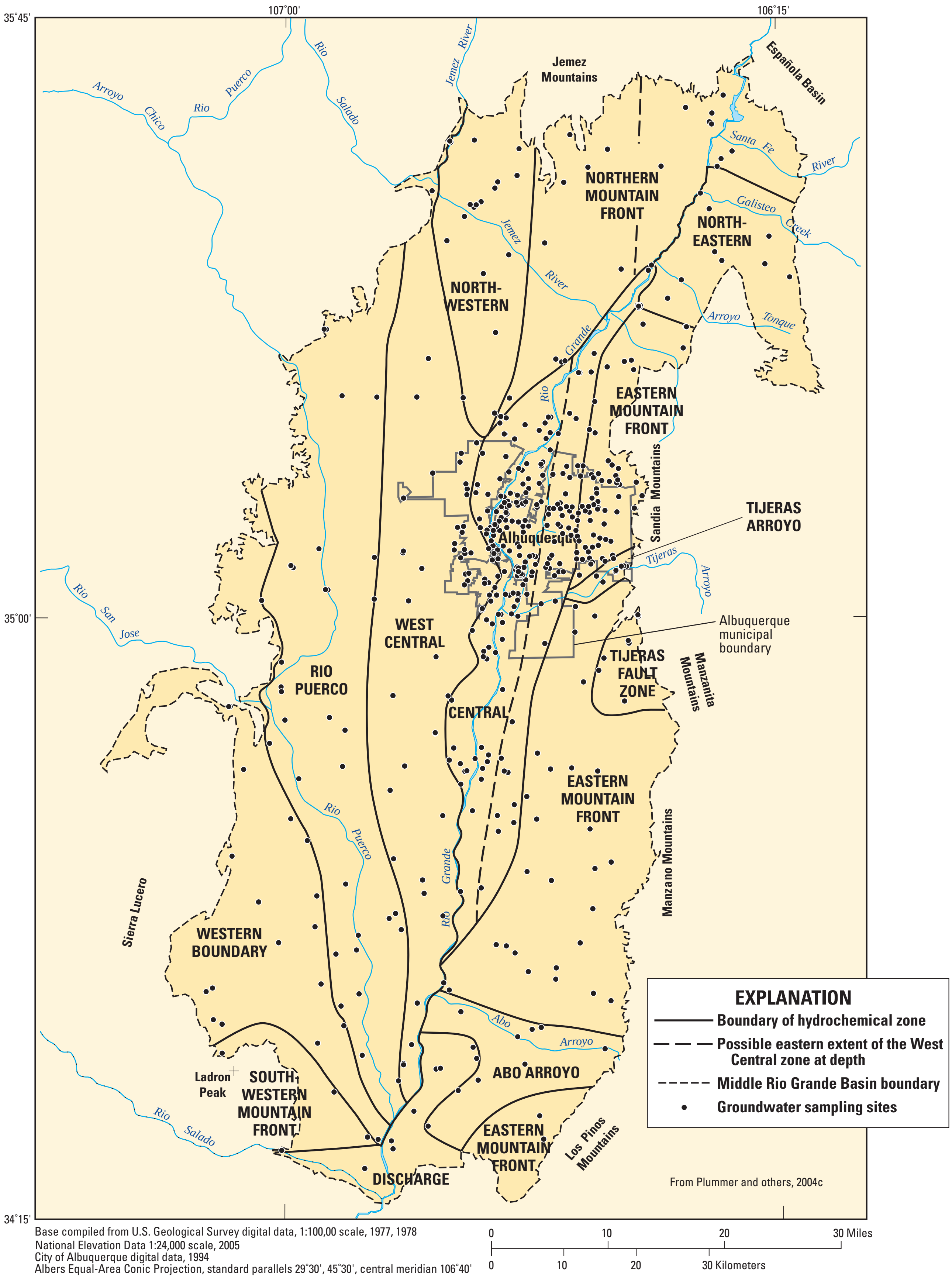


Figure 8. Hydrochemical zones and well sites in the Middle Rio Grande Basin, New Mexico.

Table 3. Median values of selected water-quality parameters by hydrochemical zone for the Middle Rio Grande Basin, New Mexico.

[Specifice conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C). **Abbreviations**: µS/cm, microsiemens per centimeter; mg/L, milligrams per liter; µg/L, micrograms per liter; pmC, percent modern carbon]

Hydrochemical zone	Specific conduct-ance (µS/cm)	Field pH	Dissolved oxygen (mg/L)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Alkalinity (mg/L as HCO_3)	SO_4 (mg/L)	Cl (mg/L)	F (mg/L)	SiO_2 (mg/L)
Northern Mountain Front	340	7.49	5.12	38.5	6.1	20.0	4.9	137	19.5	5.6	0.35	53.3
Northwestern	400	7.84	6.68	33.9	4.2	49.9	5.7	160	44.8	8.5	0.61	30.1
West Central	535	8.22	3.00	12.0	2.5	103	4.2	174	92.0	13.4	0.99	34.5
Western Boundary	4,572	7.70	4.09	135	56.4	589	15.2	300	793.	820	1.64	22.5
Rio Puerco	2,731	7.50	3.73	135	42.7	290	10.4	190	1,080	185.8	0.63	21.8
Southwestern Mountain Front	462	8.11	4.43	52.6	13.5	27.8	2.5	202	53.0	15.0	1.02	17.6
Abo Arroyo	1,055	7.45	6.23	92.5	34.4	49.2	3.1	148	346	25.9	0.90	24.0
Eastern Mountain Front	382	7.67	5.16	45.0	5.1	29.2	2.2	157	31.0	10.5	0.60	28.4
Tijeras Fault Zone	1,406	7.42	4.66	171	36.0	95.0	6.1	599	100	139	1.27	18.9
Tijeras Arroyo	677	7.39	6.97	89.4	24.5	29.3	3.8	240	115	56.6	0.60	19.5
Northeastern	1,221	7.50	6.44	141	29.5	81.8	4.8	208	390	22.7	0.51	38.5
Central	436	7.74	0.12	42.9	8.0	31.0	6.4	158	66.0	16.6	0.44	47.0
Discharge	1,771	7.70	0.08	93.0	31.0	190	10.5	157	290	280	1.40	39.0

Hydrochemical zone	Nitrate (mg/L as N)	As (µg/L)	Fe (mg/L)	Mn (mg/L)	Mo (µg/L)	Sr (mg/L)	U (µg/L)	V (µg/L)	δD (per mil)	$\delta^{18}O$ (per mil)	$\delta^{13}C$ (per mil)	^{14}C (pmC)
Northern Mountain Front	0.56	3.2	0.060	0.005	1.7	0.31	1.0	6.4	-77.7	-10.9	-8.50	33.4
Northwestern	2.44	9.8	0.030	0.002	3.4	0.57	2.7	15.6	-64.7	- 8.73	-6.93	29.6
West Central	1.24	23.2	0.028	0.002	8.2	0.20	3.7	27.9	-96.7	-12.7	-7.18	8.80
Western Boundary	0.86	1.8	0.213	0.041	9.9	2.09	4.4	5.7	-64.4	- 9.12	-4.70	6.19
Rio Puerco	0.88	1.0	0.130	0.015	7.0	3.92	6.0	3.4	-61.6	- 8.51	-7.65	36.4
Southwestern Mountain Front	1.12	0.2	0.030	0.007	3.0	0.86	0.9	1.0	-53.5	- 7.74	-5.76	40.0
Abo Arroyo	1.40	5.2	0.105	0.004	3.4	1.48	5.4	9.5	-65.2	- 9.05	-6.72	24.1
Eastern Mountain Front	0.31	2.0	0.031	0.003	2.0	0.32	3.6	7.5	-81.0	-11.4	-8.70	47.2
Tijeras Fault Zone	1.09	2.2	0.111	0.023	3.7	1.11	7.3	6.3	-74.2	-10.3	-0.98	9.70
Tijeras Arroyo	3.79	1.0	0.050	0.005	1.9	0.47	3.7	3.0	-75.7	-10.3	-6.80	72.8
Northeastern	0.64	2.7	0.170	0.004	6.7	1.72	8.5	3.8	-68.6	- 9.72	-6.40	28.5
Central	0.08	5.4	0.041	0.015	5.0	0.40	3.6	9.3	-95.4	-12.8	-8.87	61.0
Discharge	0.42	9.9	0.080	0.010	10.3	3.02	3.9	7.1	-90.8	-12.1	-7.00	10.8

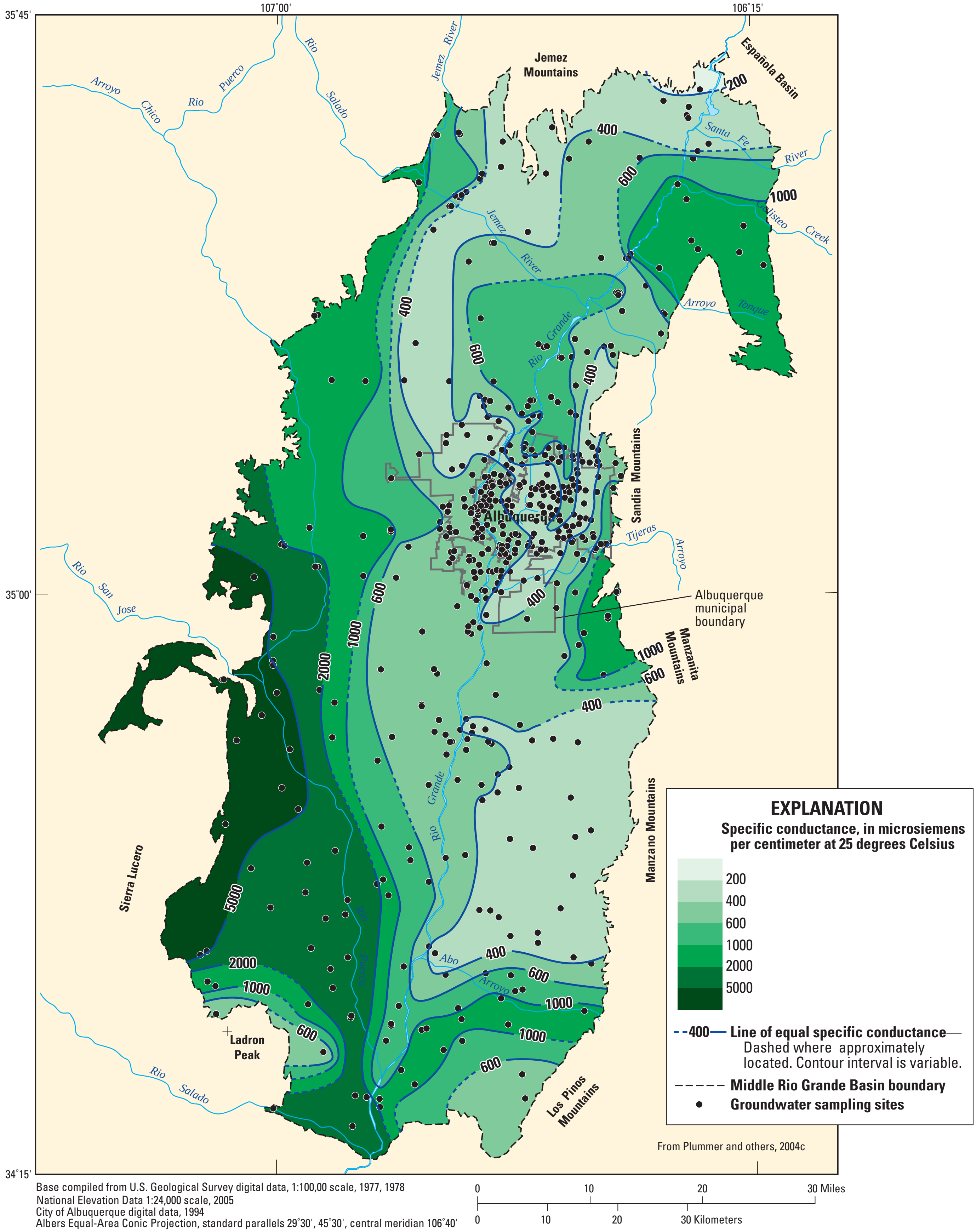


Figure 9. Specific conductance of groundwater in the Middle Rio Grande Basin, New Mexico.

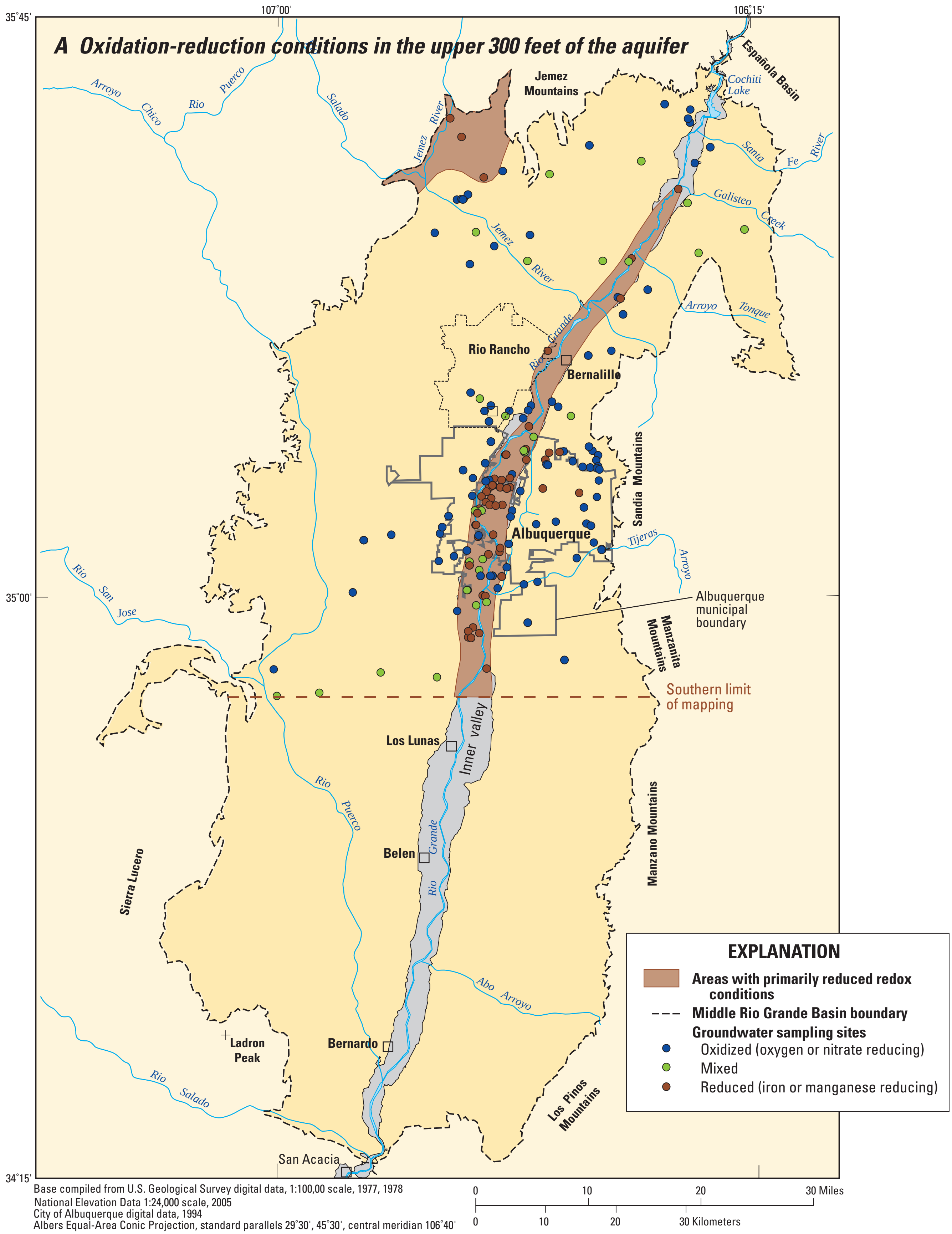


Figure 10. Oxidation-reduction conditions in the Middle Rio Grande Basin, New Mexico. (*A*) Conditions in the upper 300 feet of the aquifer, and (*B*) conditions in the deeper aquifer.

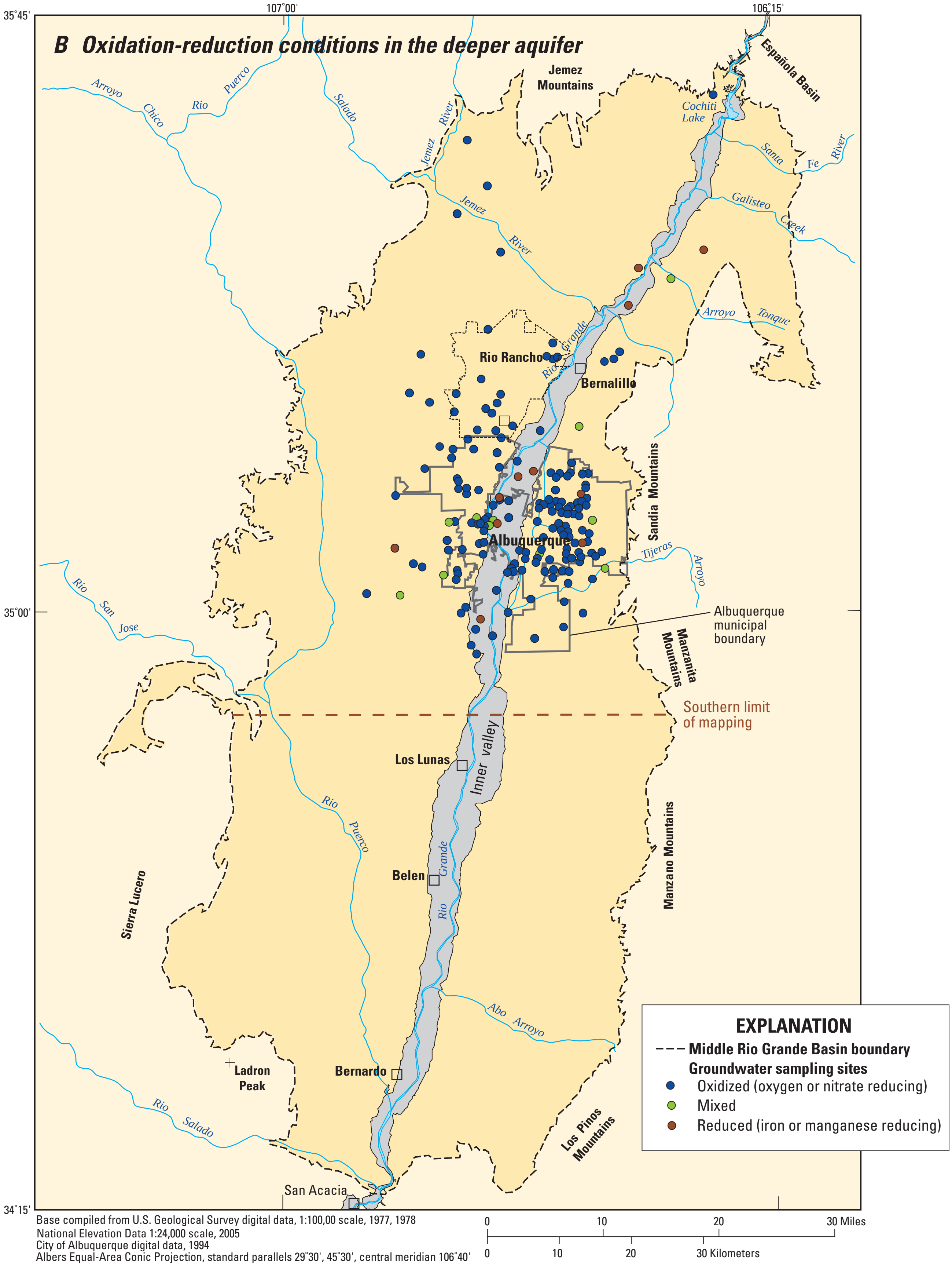


Figure 10. Oxidation-reduction conditions in the Middle Rio Grande Basin, New Mexico. (*A*) Conditions in the upper 300 feet of the aquifer, and (*B*) conditions in the deeper aquifer—Continued.

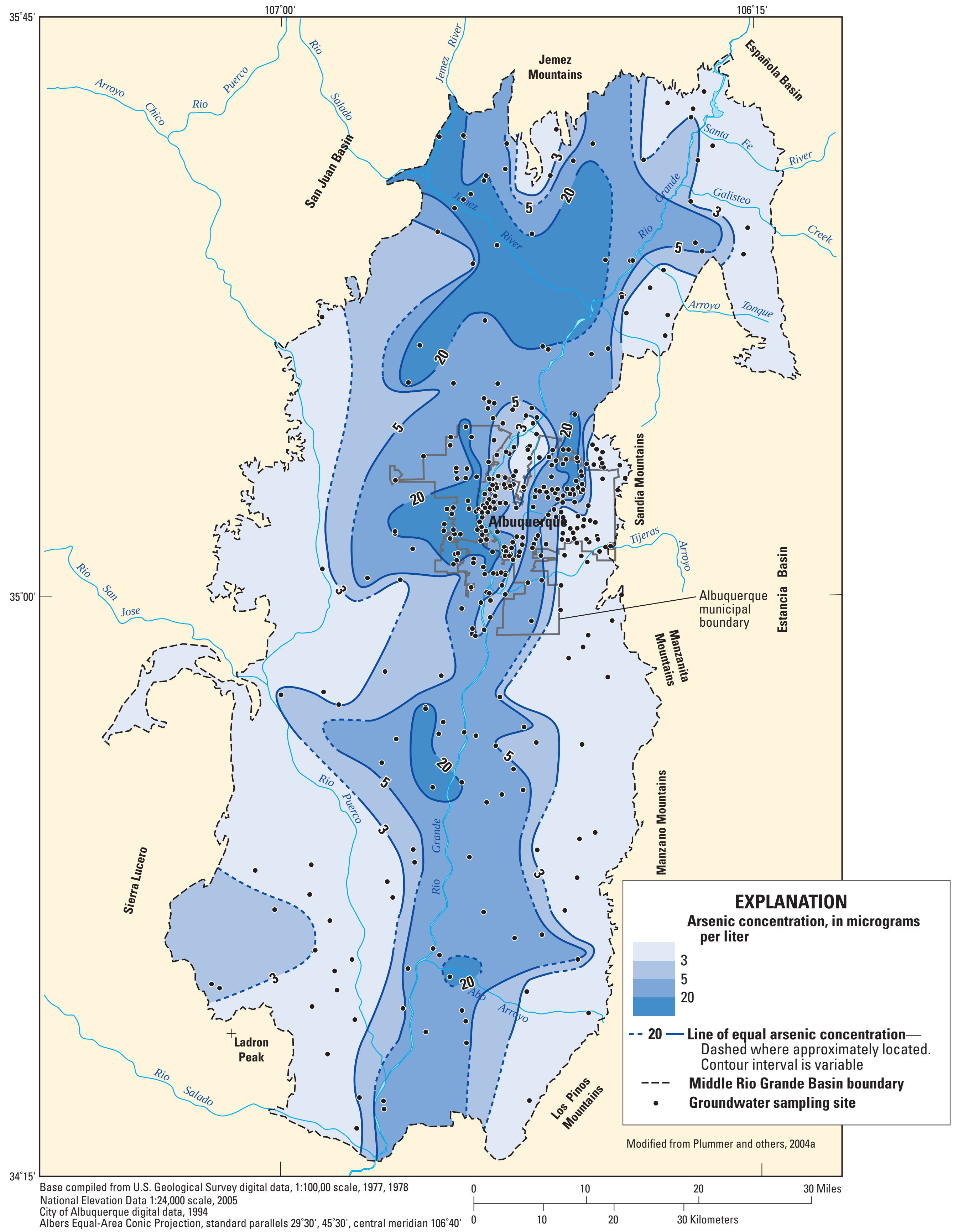


Figure 11. Concentrations of arsenic in groundwater in the Middle Rio Grande Basin, New Mexico.

Groundwater in the Central zone (fig. 8), representing recharge from the Rio Grande and its associated irrigation system, has mostly relatively small to moderate concentrations of dissolved solids (values of specific conductance generally less than 500 µS/cm) (fig. 9 and table 3) that reflect the local surface-water chemistry. Unlike groundwater throughout most of the basin, the water at shallow depths within the Central zone tends to have concentrations of dissolved oxygen and nitrate near or below detection limits (fig. 10*A*), which probably reflects a greater organic-carbon content for sediments within the Rio Grande inner valley and, therefore, greater oxygen and nitrate reduction. Plummer and others (2004a) did, however, detect nitrate at concentrations up to about 5 mg/L in some wells of the Central zone; also, some shallow groundwater had elevated concentrations of dissolved solids (values of specific conductance exceeding 800 µS/cm). Groundwater in the Central zone generally has small to moderate concentrations of arsenic (fig. 11), but concentrations exceed 10 µg/L in some areas, which is probably the result of local upwelling of deep, mineralized water along major faults or over structural highs (Bexfield and Plummer, 2003; Plummer and others, 2004a).

As defined by Plummer and others (2004a, 2004b), six hydrochemical zones are dominated by recharge through groundwater inflow along basin margins or major fault systems and(or) by arroyo infiltration: the Western Boundary, Rio Puerco, Northeastern, Tijeras Fault Zone, Tijeras Arroyo, and Abo Arroyo zones (fig. 8). Concentrations of arsenic tend to be small in all six of these hydrochemical zones (fig. 11 and table 3). With the exception of the Tijeras Arroyo zone, groundwater in these zones generally is not used for public supply because of relatively large concentrations of dissolved solids (values of specific conductance generally exceeding 1,000 µS/cm) (fig. 9 and table 3), probably as a result of either long residence times in more reactive pre-Santa Fe Group rocks (groundwater inflow) or high rates of evapotranspiration (arroyo infiltration). The relatively small area of groundwater that is noticeably influenced by infiltration from Tijeras Arroyo is generally suitable for use in public supplies, although relatively high concentrations of dissolved solids (larger than 500 mg/L) and nitrate (larger than 4 mg/L) occur in some wells in the zone (Plummer and others, 2004a). The larger concentrations of nitrate in the area might result from natural geologic sources (McQuillan and Space, 1995), septic-tank effluent from urbanization of the watershed (Blanchard, 2003), or both. The concentration of uranium in one well sampled by Plummer and others (2004a) in the Rio Puerco zone exceeded the drinking-water standard of 30 µg/L.

Potential Effects of Human Factors

As mentioned in previous sections, the long history of agricultural and urban development in the Middle Rio Grande Basin has resulted in several substantial changes to the hydrologic system, including the following: changes in the source, distribution, and chemical characteristics of recharge to the groundwater system (particularly within the Rio Grande inner valley); changes in the degree of groundwater/surface-water interaction and the magnitudes of associated fluxes of water entering and leaving the groundwater system (again, particularly in the inner valley); and changes in direction and magnitude of hydraulic gradients (particularly in and near Albuquerque). Observed and potential effects of these changes on groundwater quality in the basin are discussed in this section; previously documented effects of human activities on groundwater quality are summarized in table 4.

Irrigated agriculture and its supporting infrastructure have added to the sources and areal extent of groundwater recharge in the Rio Grande inner valley. During predevelopment, recharge in the inner valley occurred only along the wetted Rio Grande channel (although the position of the channel probably shifted frequently). Under modern conditions, recharge occurs not only along the now fixed channel of the river, but also along the unlined irrigation canals criss-crossing the inner valley and across the wider expanse of irrigated fields. Evapotranspiration of irrigation water applied to fields can increase the concentrations of dissolved solids in the excess irrigation water that recharges the groundwater system. This water can also potentially transport to the water table fertilizers and pesticides that were applied to fields. Substantial quantities of the excess irrigation water that reaches the groundwater system (or that runs off fields) can subsequently be captured by the groundwater drain system and transported back to the Rio Grande, along with increased dissolved solids and any agricultural chemicals. This water is then re-diverted into irrigation canals downstream. Agricultural development in the Middle Rio Grande Basin has, therefore, resulted in increased interaction between the groundwater and surface-water systems—in particular, increased fluxes occurring over broader areas—and introduced the means for potential transport of anthropogenic chemicals and increased dissolved solids to shallow groundwater in the inner valley.

One study is known to have been conducted to determine the effects of agricultural practices on shallow groundwater quality in the inner valley of the Middle Rio Grande Basin in particular. Bowman and Hendrickx (1998) found increases in specific conductance and concentrations of nitrate, along with low-level pesticide detections (1 µg/L or less), during the growing season directly beneath the agricultural field being studied in the southern part of the basin (table 4).

Table 4. Summary of documented effects of human activities on groundwater quality in the Middle Rio Grande Basin, New Mexico.

[<, less than; ft, feet; MRGB, Middle Rio Grande Basin; U.S. EPA, United States Evironmental Protections Agency.]

Groundwater-quality effect	Cause	General location(s)	Reference(s)
Elevated concentrations of nitrate	Agricultural fertilizer application	Shallow groundwater (depths < 100 ft) in current or former agricultural areas within the Rio Grande inner valley, including in Bernalillo, Socorro, and Valencia Counties [also found in an agricultural area of the Rio Grande inner valley in Dona Ana County to the south of the MRGB]	Nuttall (1997); Bowman and Hendrickx (1998); McQuillan and Parker (2000); Anderholm (2002)
Elevated concentrations of dissolved solids	Irrigation of agricultural fields	Shallow groundwater (depths < 100 ft) in current or former agricultural areas within the Rio Grande inner valley, including in Bernalillo and Socorro Counties [also found in an agricultural area of the Rio Grande inner valley in Dona Ana County to the south of the MRGB]	Bowman and Hendrickx (1998); McQuillan and Parker (2000); Anderholm (2002)
Detections of agricultural pesticides	Agricultural pesticide application	Shallow groundwater (depths < 100 ft) beneath an irrigated agricultural field in the Rio Grande inner valley in Socorro County [also found in an agricultural area of the Rio Grande inner valley in Dona Ana County to the south of the MRGB]	Bowman and Hendrickx (1998); Anderholm, 2002
Elevated concentrations of nitrate, dissolved solids, and chloride	Septic-tank effluent	Shallow groundwater (depths < 100 ft) in urbanized but unsewered areas, particularly in the Rio Grande inner valley near Albuquerque	McQuillan and Keller (1988); McQuillan and others (1989); Anderholm (1997)
Detections of detergent additives	Septic-tank effluent	Shallow groundwater (depths generally < 100 ft) in urbanized but unsewered areas, particularly in the Rio Grande inner valley near Albuquerque	Kues and Garcia (1995)
Detections of volatile organic compounds	Point sources, including mainly leaky underground storage tanks and industrial sites	Primarily in shallow groundwater (depths < 100 ft) in the Rio Grande inner valley in and near Albuquerque, but also locally at greater depths and (or) outside the inner valley	McQuillan and Keller (1988); Earp (1991); Anderholm (1997); U.S. Department of Energy (1999); McQuillan and Parker (2000); U.S. EPA (2005); U.S. EPA (2006b)
Detections of urban pesticides	Urban pesticide application	Shallow groundwater (depths < 100 ft) in the Rio Grande inner valley in and near Albuquerque	Anderholm (1997)

Bowman and Hendrickx (1998) concluded that these effects were rapidly mitigated by dilution with ambient groundwater and, therefore, that agricultural management practices did not pose a broad threat to the quality of shallow groundwater in the valley. A review of available nitrate data for groundwater beneath a variety of land uses in the Albuquerque area appears to support this conclusion by indicating that concentrations of nitrate were smaller in agricultural land-use settings than in urban or rangeland land-use settings (Anderholm and others, 1995). A plume of nitrate contamination in the South Valley area of southern Albuquerque is, however, believed to be associated with a former vegetable farm (Nuttall, 1997). Also, agricultural use of nutrients has reportedly caused nitrate pollution of groundwater in Bernalillo, Socorro, and Valencia Counties (McQuillan and Parker, 2000).

Results from a National Water-Quality Assessment (NAWQA) Program agricultural land-use study along about a 38-mi reach of the Rio Grande in the Rincon Valley (about 175 mi south of Albuquerque) showed low-level pesticide detections and elevated concentrations of nitrate (up to 33 mg/L) in shallow groundwater that were indicative of likely leaching of agricultural chemicals (Anderholm, 2002). Although this study did not address implications of these results for shallow groundwater quality in other areas of the Rio Grande Valley, the results indicate the potential for similar impacts in other areas with similar hydrogeology and agricultural practices, such as the Middle Rio Grande Basin. Surface-water data from the Anderholm (2002) study showed similar findings of elevated concentrations of nitrate (up to about 3 mg/L in groundwater drains) and low-level pesticide detections, indicating that agricultural practices also have had an effect on surface-water quality in the valley. A recent study of sources of salinity to the Rio Grande from its headwaters to Fort Quitman, Texas, found that a large contributor of river salinization is seepage of deep, sedimentary-origin brines to the river and drains under structural controls (the primary mechanism being movement along faults near the southern ends of structural basins) and that agriculture plays only a secondary role in salinization of the river (Phillips and others, 2003).

One effect of urbanization on the groundwater-flow system of the basin has been to alter flow directions and travel times, primarily in and around Albuquerque. Groundwater withdrawals for public supply and associated declines in hydraulic head have resulted in dominating east-west components of flow away from the Rio Grande inner valley near Albuquerque, in contrast to the primarily north-south flow through the valley area under predevelopment conditions. In some areas, declines in hydraulic head in the production zone have resulted in changes in vertical flow directions (at least during summer months), causing flow into the production zone from both shallower and deeper parts of the aquifer (Bexfield and Anderholm, 2002). Changes in head also have increased the magnitudes of both horizontal and vertical hydraulic gradients over broad areas, thereby decreasing groundwater travel times.

The changes in hydraulic gradients caused by groundwater withdrawals have the potential to affect groundwater quality. For example, changes in hydraulic gradients might exacerbate any existing groundwater-quality problems associated with land use by causing contaminants to spread more quickly across larger areas and to be drawn to greater depths in the aquifer. In particular, contaminants reaching the water table in the inner valley (where the aquifer is most susceptible) could be drawn toward major pumping centers to the east or west. Also, declining heads in the production zone could potentially cause deeper, more mineralized groundwater to move upward and degrade the quality of water used for public supply. A study of changes in 10 chemical parameters in groundwater from 93 City of Albuquerque public-supply wells over a 10-year period (1988–97) found that five parameters had a greater number of upward rather than downward trends among the wells; the opposite was true for the other five. For the five parameters—concentrations of dissolved solids, chloride, sulfate, sodium, and silica—that had more increasing than decreasing trends, the magnitudes of those trends were small (generally less than 1 mg/L), indicating no substantial regional changes in water quality during the time period of study (Bexfield and Anderholm, 2002). Concentrations of arsenic, which are believed to be elevated in deep, mineralized waters of the basin, had more decreasing than increasing trends.

Additional effects of urbanization have been to add potentially substantial new sources of recharge to the groundwater system—seepage from septic tanks, sewer and distribution lines, and turf irrigation, for example—as well as to change the chemical characteristics of some important sources of recharge, such as arroyo infiltration. In addition, urban land uses can result in local water-quality issues where (for example) contaminants produced or released at landfills, industrial operations, military operations, or underground storage tanks are transported to the water table. In the Middle Rio Grande Basin, seepage from various urban water sources would be expected to recharge the aquifer and affect groundwater quality almost exclusively in and near the inner valley of the Albuquerque area, where depths to water are generally within about 30 ft of land surface (Anderholm, 1987) and urban development is extensive. Indicators that groundwater quality has been affected by one or more urban activities would include elevated concentrations of nutrients and(or) dissolved solids and detections of pesticides and VOCs.

McQuillan and Keller (1988) report that septic-tank effluent has resulted in groundwater being contaminated with nitrate and(or) anaerobic respiration byproducts in Albuquerque, Belen, Bernalillo, Corrales, and Los Lunas (table 4). McQuillan and others (1989) concluded that elevated concentrations of dissolved solids, nitrate, and chloride in shallow groundwater in an area of the inner valley of southern Albuquerque were the result of residential development utilizing septic systems. In a study conducted in unincorporated areas of Bernalillo County, Kues and Garcia (1995) detected detergent additives—indicating the likely presence of domestic sewage—in 4 of 15 domestic wells sampled in the inner valley; detections were generally in wells with shallower known depths and were accompanied by relatively high concentrations of ammonia. Anderholm (1997) studied shallow groundwater quality in 30 wells in a NAWQA urban land-use study area in the inner valley near Albuquerque and concluded that infiltration of septic-system effluent had affected the groundwater quality in some areas (based on small concentrations of dissolved oxygen, large concentrations of dissolved organic carbon, and elevated concentrations of chloride).

Anderholm (1997) did not address the effects of specific land uses or of urban recharge sources besides septic-tank effluent on shallow groundwater quality in the inner valley. However, pesticides of primarily urban use were detected in several wells (all in areas of nonagricultural land use), which might reflect infiltration of turf irrigation water and(or) urban runoff from precipitation events (table 4). Elevated concentrations of nitrate and dissolved solids reported by Plummer and others (2004a) in some samples of young, shallow groundwater in the inner valley might be indicative of recent recharge of irrigation water, septic-tank effluent, or other urban recharge sources. Also, elevated concentrations of nitrate have been found in both perched and regional groundwater on Kirtland Air Force Base, southeast of Albuquerque (U.S. Department of Energy, 1999). The sources of elevated nitrate have not been conclusively determined, but suspected sources have included septic tanks and leach fields, waste storage and disposal sites, and landfills (U.S. Department of Energy, 1999).

VOCs indicative of urban recharge sources also have been detected in the basin, primarily in shallow groundwater of the inner valley (table 4). In the South Valley area of Albuquerque, McQuillan and Keller (1988) make reference to about 10 sites at which groundwater was contaminated by VOCs—particularly chlorinated solvents—that are associated with industrial development (which began in the area in the 1950s) and to 20 or more sites of groundwater contamination with petroleum products. McQuillan and Keller (1988) indicate that groundwater contamination in the South Valley area was once limited to depths of about 100 ft or less, but that pumping has drawn contamination to increasingly greater depths. All three sites of groundwater contamination with VOCs that are on the USEPA Superfund list in the Middle Rio Grande Basin are within the inner valley near Albuquerque (U.S. Environmental Protection Agency, 2006), although VOCs also have been detected in groundwater beneath upland areas, including on Kirtland Air Force Base (U.S. Department of Energy, 1999). A network consisting mostly of shallow wells within the inner valley that is monitored by the City of Albuquerque Environmental Health Department (Earp, 1991) has yielded detections of chlorinated solvents and(or) petroleum products in several wells. Although point sources—particularly leaky underground storage tanks—appear to account for most of the cases of groundwater contamination with VOCs in New Mexico (McQuillan and Parker, 2000) and the South Valley area of Albuquerque in particular (McQuillan and Keller, 1988), urban runoff has the potential to contribute VOCs to the aquifer. In the Albuquerque area, stormwater that does not infiltrate locally runs off into a storm-drain system that typically carries the water to concrete-lined drainage channels and(or) natural arroyo channels (Kelly and Romero, 2003; City of Albuquerque, 2007); these channels carry the untreated stormwater to the Rio Grande when flow is sufficient.

The NAWQA urban land-use study by Anderholm (1997) detected low levels of chlorinated solvents and(or) petroleum products or additives in five shallow monitoring wells in the inner valley (table 4). A separate NAWQA study of the quality of deeper groundwater from domestic wells in the Rio Grande inner valley of the Middle Rio Grande Basin and basins to the south detected no VOCs (Bexfield and Anderholm, 1997). However, a NAWQA study of the vulnerability of public-supply wells in the Albuquerque metropolitan area to contamination found very small (subparts per billion) concentrations of VOCs in some supply wells both inside and outside of the inner valley (Carter and others, 2007). Also, concentrations of VOCs have approached or exceeded drinking-water standards in some deep public-supply wells near known chemical releases, resulting in well closures (U.S. Environmental Protection Agency, 2010). As McQuillan and Keller (1988) suggest, the substantial water-level declines that are common in the vicinity of active public-supply wells in the Albuquerque area likely contribute to movement of contaminants beyond the shallow zone of the aquifer. McQuillan and Parker (2000) also state that an increasing number of contamination cases are being discovered in New Mexico in areas where the depth to groundwater is more than 200 ft.

Summary

The Middle Rio Grande Basin is an extensive alluvial basin with a large thickness of relatively unconsolidated aquifer sediments, generally long groundwater travel times, and only local areas of substantial intrinsic groundwater susceptibility to contamination. The groundwater system is hydraulically connected to the through-flowing Rio Grande, which is within an inner valley where depths to water are generally less than 30 ft. Groundwater conditions in the basin-fill aquifer generally are unconfined, although they are semiconfined at depth. Under natural conditions, the aquifer is recharged primarily through infiltration of surface water along the Rio Grande and its major tributaries, mountain-front processes, and subsurface inflow along the basin margins. Because of low precipitation rates relative to evapotranspiration and generally large depths to groundwater, there is little or no direct areal recharge from precipitation across most of the basin. The estimated rate of natural recharge (130,000 acre-ft/yr) is small relative to the volume of the aquifer in the basin, resulting in groundwater travel times that commonly exceed 10,000 years.

A long history of agricultural and urban land uses has had a substantial effect on the groundwater-flow system of the Middle Rio Grande Basin. The estimated annual flux of water entering and leaving the groundwater system has more than quadrupled since predevelopment. Most of this increased flux occurs in the inner valley as a result of the effects of irrigated agriculture and its associated infrastructure, which has also spread recharge across broader areas and affected its chemical composition. Changing hydraulic gradients that have resulted from large groundwater withdrawals for public supplies in and around Albuquerque have induced greater infiltration from the Rio Grande, in addition to changing horizontal and vertical groundwater-flow directions and locally increasing groundwater-flow velocities. Urbanization also has resulted in new sources of recharge (such as septic tanks) and affected the chemical composition of existing sources of recharge.

Groundwater chemistry in the Middle Rio Grande Basin is determined primarily by the source and composition of recharge. Evapotranspiration, geology, and other natural factors in recharge areas have resulted in relatively large concentrations of some contaminants (particularly dissolved solids and arsenic). Human activities have affected the quality of groundwater in some areas, although the general lack of areal recharge results in low susceptibility of the aquifer to anthropogenic contamination across much of the basin. Groundwater susceptibility and vulnerability is highest in the inner valley, where the occurrence of recharge combines with shallow depths to water and intense agricultural and urban activity. Within the inner valley, detections of elevated concentrations of dissolved solids, nitrate, pesticides, and VOCs have been associated with urban and(or) agricultural sources, including septic tanks, industrial activities, and fertilizer use. In most cases, anthropogenic contaminants have migrated only relatively short distances and have been detected only at relatively shallow depths, probably because generally low groundwater fluxes and high horizontal to vertical anisotropy tend to result in slow horizontal and vertical migration, respectively. In some areas, however, increased horizontal and vertical gradients resulting from urban groundwater withdrawals have caused more extensive migration of contaminants, which has affected the quality of water in a small number of public-supply wells. Also, detections of tracers of young groundwater and(or) anthropogenic contaminants in some areas that are located at substantial distances from primary recharge sources and that have relatively large depths to groundwater could imply the existence of local sources of recharge that have not been well characterized. Such detections also could imply that these areas are more susceptible to contamination than most historical studies would appear to indicate.

References Cited

Albuquerque Bernalillo County Water Utility Authority, 2010, San Juan-Chama drinking water project, accessed April 2010 at http://www.abcwua.org/content/view/31/24/.

Anderholm, S.K., 1987, Reconnaissance of hydrology, land use, ground-water chemistry, and effects of land use on ground-water chemistry in the Albuquerque-Belen Basin, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 86–4174, 37 p.

Anderholm, S.K., 1988, Ground-water geochemistry of the Albuquerque-Belen Basin, central New Mexico: U.S. Geological Survey Water-Resources Investigations Report 86–4094, 110 p.

Anderholm, S.K., 1997, Water-quality assessment of the Rio Grande Valley, Colorado, New Mexico, and Texas—Shallow ground-water quality and land use in the Albuquerque area, central New Mexico, 1993: U.S. Geological Survey Water-Resources Investigations Report 97–4067, 73 p.

Anderholm, S.K., 2001, Mountain-front recharge along the east side of the Albuquerque Basin, central New Mexico (revised): U.S. Geological Survey Water-Resources Investigations Report 00–4010, 36 p.

Anderholm, S.K., 2002, Water-quality assessment of the Rio Grande Valley, Colorado, New Mexico, and Texas—Surface-water quality, shallow ground-water quality, and factors affecting water quality in the Rincon Valley, south-central New Mexico, 1994–95: U.S. Geological Survey Water-Resources Investigations Report 02–4188, 117 p. Available at http://pubs.usgs.gov/wri/wri02-4188/.

Anderholm, S.K., Radell, M.J., and Richey, S.F., 1995, Water-quality assessment of the Rio Grande Valley study unit, Colorado, New Mexico, and Texas—Analysis of selected nutrient, suspended-sediment, and pesticide data: U.S. Geological Survey Water-Resources Investigations Report 94–4061, 203 p.

Bartolino, J.R., and Cole, J.C., 2002, Ground-water resources of the Middle Rio Grande Basin, New Mexico: U.S. Geological Survey Circular 1222, 132 p. Available at http://pubs.usgs.gov/circ/2002/circ1222/.

Bexfield, L.M., and Anderholm, S.K., 1997, Water-quality assessment of the Rio Grande Valley, Colorado, New Mexico, and Texas—Ground-water quality in the Rio Grande flood plain, Cochiti Lake, New Mexico, to El Paso, Texas, 1995: U.S. Geological Survey Water-Resources Investigations Report 96–4249, 93 p.

Bexfield, L.M., and Anderholm, S.K., 2000, Predevelopment water-level map of the Santa Fe Group aquifer system in the Middle Rio Grande Basin between Cochiti Lake and San Acacia, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 00–4249, 1 sheet. Available at http://pubs.er.usgs.gov/publication/wri004249.

Bexfield, L.M., and Anderholm, S.K., 2002a, Estimated water-level declines in the Santa Fe Group aquifer system in the Albuquerque area, central New Mexico, predevelopment to 2002: U.S. Geological Survey Water-Resources Investigations Report 02–4233, 1 sheet. Available at http://pubs.er.usgs.gov/usgspubs/wri/wri024233.

Bexfield, L.M., and Anderholm, S.K., 2002b, Spatial patterns and temporal variability in water quality from City of Albuquerque drinking-water supply wells and piezometer nests, with implications for the ground-water flow system: U.S. Geological Survey Water-Resources Investigations Report 01–4244, 101 p.

Bexfield, L.M., Danskin, W.R., and McAda, D.P., 2004, Simulation-optimization approach to management of ground-water resources in the Albuquerque area, New Mexico, through 2040: U.S. Geological Survey Scientific Investigations Report 2004–5140, 82 p.

Bexfield, L.M., and McAda, D.P., 2003, Simulated effects of ground-water management scenarios on the Santa Fe Group aquifer system, Middle Rio Grande Basin, New Mexico, 2001–2040: U.S. Geological Survey Water-Resources Investigations Report 03–4040, 39 p. Available at http://pubs.usgs.gov/wri/wri034040/.

Bexfield, L.M., and Plummer, L.N., 2003, Chapter 11—Occurrence of arsenic in ground water of the Middle Rio Grande Basin, central New Mexico, *in* Welch, A.H., and Stollenwerk, K.G., eds., Arsenic in ground water—geochemistry and occurrence: Boston, Kluwer Academic Publishers, p. 295–327.

Bjorklund, L.J., and Maxwell, B.W., 1961, Availability of ground water in the Albuquerque area, Bernalillo and Sandoval counties, New Mexico: Santa Fe, N. Mex., New Mexico State Engineer Technical Report 21, 117 p.

Blanchard, P.J., 2003, Water quality and depth to water, 2001–02, and graphs of selected constituents and depths to water, period of record through 2002, in selected wells, eastern Bernalillo County, New Mexico: U.S. Geological Survey Open-File Report 03–81, 43 p.

Bowman, R.S., and Hendrickx, J.M.H., 1998, Determination of agricultural chemical impacts on shallow groundwater quality in the Rio Grande Valley—Las Nutrias groundwater project: Las Cruces, N. Mex., New Mexico Water Resources Research Institute Technical Completion Report no. 308, 45 p.

Carter, J.M., Delzer, G.C., Kingsbury, J.A., and Hopple, J.A., 2007, Concentration data for anthropogenic organic compounds in ground water, surface water, and finished water of selected community water systems in the United States, 2002–05: U.S. Geological Survey Data Series 268, 30 p. Available at http://pubs.usgs.gov/ds/2007/268/.

City of Albuquerque, 2007, Storm water pollution prevention, accessed January 2007 at http://www.cabq.gov/flood/swpp.html.

City of Albuquerque, 2010, Water resources management strategy, accessed April 2010 at http://www.abcwua.org/content/view/190/332/.

Connell, S.D., 2006, Preliminary geologic map of the Albuquerque-Rio Rancho metropolitan area and vicinity, Bernalillo and Sandoval counties, New Mexico: Socorro, N. Mex., New Mexico Bureau of Geology and Mineral Resources Open-File Report 496, 2 plates.

Crawford, C.S., Cully, A.C., Leutheuser, R., Sifuentes, M.S., White, L.H., and Wilber, J.P., 1993, middle Rio Grande ecosystem—Bosque biological management plan: Albuquerque, N. Mex., Middle Rio Grande Bosque Biological Interagency Team, 291 p.

Earp, D., 1991, Summary of locations, construction details and monitoring results for ground-water quality monitoring wells through 1990: Albuquerque, N. Mex., City of Albuquerque Environmental Health Department report, 95 p.

Fenneman, N.M., 1931, Physiography of the western United States: New York, McGraw-Hill, 534 p.

Grant, P.R., Jr., 1999, Subsurface geology and related hydrologic conditions, Santa Fe embayment and contiguous areas, New Mexico, *in* Pazzaglia, F.J., and Lucas S.G., eds., Albuquerque geology, New Mexico Geological Society Fiftieth Annual Field Conference, September 22–25, 1999: Socorro, N. Mex., New Mexico Geological Society, p. 425–435.

Grauch, V.J.S., Gillespie, C.L., and Keller, G.R., 1999, Discussion of new gravity maps for the Albuquerque Basin area, *in* Pazzaglia, F.J., and Lucas, S.G., eds., Albuquerque geology, New Mexico Geological Society Fiftieth Annual Field Conference, September 22–25, 1999: Socorro, N. Mex., New Mexico Geological Society, p. 119–124.

Grauch, V.J.S., Hudson, M.R., and Minor, S.A., 2001, Aeromagnetic expression of faults that offset basin fill, Albuquerque Basin, New Mexico: Geophysics, v. 66, no. 3, p. 707–720.

Green, G.N., and Jones, G.E., 1997, The digital geologic map of New Mexico in ARC/INFO format: U.S. Geological Survey Open-File Report 94–0052. Available at http://pubs.usgs.gov/of/1997/ofr-97-0052/.

Ground-Water Science, Inc., 1995, Land-use trends and their effect on water use and the hydrologic budget in the Albuquerque Basin, New Mexico: Albuquerque, N. Mex., Ground-Water Science, Inc., for the Bureau of Reclamation Middle Rio Grande Assessment, 72 p.

Hawley, J.W., and Haase, C.S., 1992, Hydrogeologic framework of the northern Albuquerque Basin: Socorro, N. Mex., New Mexico Bureau of Mines and Mineral Resources Open-File Report 387, variously paged.

Hawley, J.W., Haase, C.S., and Lozinsky, R.P., 1995, An underground view of the Albuquerque Basin, *in* Ortega-Klett, C.T., ed., The water future of Albuquerque and the Middle Rio Grande Basin: Proceedings of the 39th Annual New Mexico Water Conference, November 3–4, 1994, New Mexico Water Resources Research Institute WRRI Report No. 290, p. 37–55.

Kelly, T., and Romero, O., 2003, Rainfall, runoff, and water-quality data for the urban storm-water program in the Albuquerque, New Mexico, metropolitan area, water year 2001: U.S. Geological Survey Open-File Report 03–169, 153 p.

Kelly, T.E., 1982, History of water use in the greater Albuquerque area, *in* Grambling, J.A., and Wells, S.G., eds., Albuquerque Country II: Socorro, N. Mex., New Mexico Geological Society Guidebook, 33d Annual Field Conference, p. 351–355.

Kernodle, J.M., McAda, D.P., and Thorn, C.R., 1995, Simulation of ground-water flow in the Albuquerque Basin, central New Mexico, 1901–1994, with projections to 2020: U.S. Geological Survey Water-Resources Investigations Report 94–4251, 114 p.

Kinkel, B., 1995, Estimates of consumptive use requirements for irrigated agriculture and riparian vegetation, vol. I: Bureau of Reclamation Albuquerque Area Office, Middle Rio Grande Water Assessment Supporting Document no. 6, 31 p.

Kues, G.E., and Garcia, B.M., 1995, Ground-water-quality and ground-water-level data, Bernalillo County, central New Mexico, 1990–1993: U.S. Geological Survey Open-File Report 95–385, 76 p.

Logan, L.M., 1990, Geochemistry of the Albuquerque municipal area, Albuquerque, New Mexico: Socorro, N. Mex., New Mexico Institute of Mining and Technology, Independent study, 234 p.

McAda, D.P., and Barroll, Peggy, 2002, Simulation of ground-water flow in the Middle Rio Grande Basin between Cochiti and San Acacia, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 02–4200, 81 p. Available at http://pubs.usgs.gov/wri/wri02-4200/.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

McQuillan, D.M., Jasper, M.J., and Swanson, B.H., 1989, Ground-water contamination by septic-tank use—A field study in the Albuquerque south valley-west mesa region, Bernalillo County, New Mexico: New Mexico Health & Environment Department report EID/GWB–89/2, 37 p.

McQuillan, D.M., and Keller, N.S., 1988, Ground-water contamination in New Mexico—1927–1986, *in* Proceedings of the 32nd Annual New Mexico Water Conference, November 5–6, 1987, Albuquerque, N. Mex., New Mexico Water Resources Research Institute Report No. 229, p. 12–33.

McQuillan, D., and Parker, J., 2000, Ground-water contamination and remediation in New Mexico—1927–2000: Santa Fe, N. Mex., New Mexico Environment Department Open-File Report.

McQuillan, D., and Space, M., 1995, High ground-water nitrate in Tijeras Arroyo, Hell Canyon, and Abo Arroyo—Evidence for a natural geologic origin: New Mexico Geology, v. 17, no. 2.

Meeks, T.O., 1949, The occurrence of ground water in the Tijeras soil conservation district, Bernalillo County, New Mexico: U.S. Department of Agriculture Regional Bulletin 109, Geological Series I, Soil Conservation Service Region 6, variously paged.

Norman, V.W., 1968, Trends of suspended sediments in the upper Rio Grande Basin in New Mexico: Albuquerque, N. Mex., University of New Mexico, Master's thesis, 108 p.

Nuttall, H.E., 1997, A technology for in-situ denitrification of ground water, *in* Proceedings, 42nd Annual New Mexico Water Conference—Water Issues of Eastern New Mexico: Las Cruces, N. Mex., Water Resources Research Institute Report No. 304.

Oak Ridge National Laboratory, 2005, LandScan global population database, accessed September 2006 at http://www.ornl.gov/landscan/.

Phillips, F.M., Hogan, J., Mills, S., and Hendrickx, J.M.M., 2003, Environmental tracers applied to quantifying causes of salinity in arid-region rivers—Preliminary results from the Rio Grande, southwestern USA, *in* Alsharhan, A.S., and Wood, W.W., eds., Water resource perspectives—evaluation, management, and policy: Amsterdam, Elsevier Science, p. 327–334.

Plummer, L.N., Bexfield, L.M., Anderholm, S.K., Sanford, W.E., and Busenberg, Eurybiades, 2004a, Geochemical characterization of ground-water flow in the Santa Fe Group aquifer system, Middle Rio Grande Basin, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 03–4131, 395 p. Available at http://pubs.usgs.gov/wri/wri034131/.

Plummer, L.N., Bexfield, L.M., Anderholm, S.K., Sanford, W.E., and Busenberg, Eurybiades, 2004b, Hydrochemical tracers in the Middle Rio Grande Basin, USA—1. conceptualization of groundwater flow: Hydrogeology Journal, v. 12, no. 4, p.359–388.

Plummer, L.N., Sanford, W.E., Bexfield, L.M., Anderholm, S.K., and Busenberg, Eurybiades, 2004c, Using chemical data and aquifer simulation to characterize recharge and ground-water flow in the Middle Rio Grande Basin, New Mexico, *in* Hogan, J.F., Phillips, F.M., and Scanlon, B.R., eds., Groundwater recharge in a desert environment—The southwestern United States, Water Science and Applications Series, vol. 9: Washington, D.C., American Geophysical Union, p. 185–216.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM), Oregon State University, accessed April 2005 at http://www.ocs.oregonstate.edu/prism/.

Robson, S.G., and Banta, E.R.,1995, Ground water atlas of the United States—Segment 2, Arizona, Colorado, New Mexico, Utah: U.S. Geological Survey Hydrologic Atlas 730–C, 32 p. Available at http://pubs.usgs.gov/ha/ha730/ch_c/index.html.

Russell, L.R., and Snelson, S., 1990, Structural style and tectonic evolution of the Albuquerque Basin segment of the Rio Grande Rift, *in* Pinet, B., and Bois, C., eds., The potential of deep seismic profiling for hydrocarbon exploration: Editions Technip, Paris, French Petroleum Institute Research Conference Proceedings, p. 175–207.

Sanford, W.E., Plummer, L.N., McAda, D.P., Bexfield, L.M., and Anderholm, S.K., 2004a, Use of environmental tracers to estimate parameters for a predevelopment ground-water-flow model of the Middle Rio Grande Basin, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 03–4286, 102 p.

Sanford, W.E., Plummer, L.N., McAda, D.P., Bexfield, L.M., and Anderholm, S.K., 2004b, Hydrochemical tracers in the Middle Rio Grande Basin, USA—2. calibration of a groundwater-flow model: Hydrogeology Journal, v. 12, no. 4, p. 389–407.

Scurlock, D., 1998, An environmental history of the Middle Rio Grande Basin: Fort Collins, Colo., U.S. Department of Agriculture Forest Service General Technical Report RMRS–GTR–5, 440 p.

Smith, G.A., and Kuhle, A.J., 1998, Hydrostratigraphic implications of new geological mapping in the Santo Domingo basin, New Mexico: New Mexico Geology, v. 20, no. 1, p. 21–27.

Thorn, C.R., McAda, D.P., and Kernodle, J.M., 1993, Geohydrologic framework and hydrologic conditions in the Albuquerque Basin, central New Mexico: U.S. Geological Survey Water-Resources Investigations Report 93–4149, 106 p.

Titus, F.B., 1961, Ground-water geology of the Rio Grande trough in north-central New Mexico, with sections on the Jemez Caldera and Lucero uplift, *in* Northrop, S.A., ed., Guidebook of the Albuquerque country: New Mexico Geological Society, 12th Field Conference, p. 186–192.

Trainer, F.W., Rogers, R.J., and Sorey, M.L., 2000, Geothermal hydrology of Valles caldera and the southwestern Jemez Mountains, New Mexico: U.S. Geological Survey Water-Resources Investigations Report 00–4067, 115 p.

U.S. Census Bureau, 2001, Ranking tables in metropolitan areas—Population in 2000 and population change from 1990 to 2000 (PHC–T–3), accessed October 2006 at http://www.census.gov/population/www/cen2000/phc-t3.html.

U.S. Department of Energy, 1999, Sandia National Laboratories/New Mexico—Final site-wide environmental impact statement: Department of Energy EIS–0281, v. 1, chapters 1–15, October 1999, variously paged.

U.S. Environmental Protection Agency, 2006, Site status summaries—New Mexico, accessed December 2006 at http://www.epa.gov/earth1r6/6sf/6sf-nm.htm.

U.S. Environmental Protection Agency, 2009, Drinking water contaminants, accessed June 12, 2009 at http://www.epa.gov/safewater/contaminants/.

U.S. Environmental Protection Agency, 2010, South Valley (Bernalillo County), Albuquerque, New Mexico, EPA ID# NMD 980745558, accessed April 2010 at http://www.epa.gov/region6/6sf/pdffiles/0600881.pdf.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed February 2007 at http://www.mrlc.gov/.

Western Regional Climate Center, 2006a, Period of record monthly climate summary for Albuquerque WSFO Airport, New Mexico, accessed October 2006 at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nm0234.

Western Regional Climate Center, 2006b, Period of record monthly climate summary for Sandia Crest, New Mexico, accessed October 2006 at http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nm8011.

Wilson, B.C., 1992, Water use by categories in New Mexico counties and river basins, and irrigated acreage in 1990: Santa Fe, N. Mex., New Mexico State Engineer Office Technical Report 47, 141 p.

Wilson, B.C., and Lucero, A.A., 1997, Water use by categories in New Mexico counties and river basins, and irrigated acreage in 1995: Santa Fe, N. Mex., New Mexico State Engineer Office Technical Report 49, 149 p.

Wilson, B.C., Lucero, A.A., Romero, J.T., and Romero, P.J., 2003, Water use by categories in New Mexico counties and river basins, and irrigated acreage in 2000: Santa Fe, N. Mex., New Mexico State Engineer Office Technical Report 51, 164 p.

Wozniak, F.E., 1996, Human impacts on riparian ecosystems of the middle Rio Grande valley during historic times, *in* Shaw, D.W., and Finch, D.M., eds., Desired future conditions for southwestern riparian ecosystems—Bringing interests and concerns together: U.S. Department of Agriculture, Forest Service General Technical Report RM–GTR–272, p. 33–43.

Section 12.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifers in the Santa Ana Basin, California

By Susan A. Thiros

Basin Overview

The hydrologic cycles of the groundwater basins in the Santa Ana Basin are greatly affected by human activities as a result of the semiarid climate and the water demands of the large urban population (Belitz and others, 2004). Pumping from the basin-fill aquifers and changes in the sources and distribution of recharge that have accompanied development have accelerated the rate of groundwater flow and the transport of dissolved constituents through the aquifers. The quality of groundwater in parts of these aquifers reflects the quality of the surface water used for recharge during the past 50 years. Groundwater recharged before any substantial human effects on water quality or the flow system occurred has been partly replaced by human-affected water that has entered the aquifers since the early 1950s. Similarly, the future quality of groundwater will be affected by the quality of surface water currently being used for recharge in the basins.

The 2,700-mi^2 Santa Ana Basin watershed is within the Coastal Range Physiographic Province in southern California, which is characterized by prominent mountains that rise steeply from the relatively flat-lying coastal plain and inland valleys (fig. 1). The tallest peaks in the San Gabriel, San Bernardino, and San Jacinto Mountains rise to altitudes greater than 10,000 ft. The Santa Ana Basin comprises three distinct groundwater basins that were studied by the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program—the San Jacinto Basin, the Inland Basin, and the Coastal Basin. These sediment-filled basins are hydraulically separated from each other by relatively impervious rocks of intervening hills and mountains. The groundwater basins of the Santa Ana Basin are three of many such basins along the length of the State of California (Planert and Williams, 1995) and are part of the California Coastal Basin aquifers, a principal aquifer of the United States (U.S. Geological Survey, 2003a).

The climate of the Santa Ana Basin is Mediterranean, with hot, dry sum¬mers and cool, wet winters. Average annual precipitation ranges from 10 to 24 in. in the coastal plain and inland valleys and from 24 to 48 in. in the San Gabriel and San Bernardino Mountains (Belitz and others, 2004, p. 3). Most of the precipitation occurs between November and March in the form of rain, but with variable amounts of snow in the higher elevations. This seasonal precipitation pattern can result in high streamflow in the spring, followed by low flow during the dry season.

The Santa Ana Basin is drained by the Santa Ana River, which has the largest drainage area of any stream in southern California. The Santa Ana River begins in the San Bernardino Mountains and flows westward more than 100 mi to the Pacific Ocean near Huntington Beach. Streamflow during the summer months is maintained by discharge from wastewater treatment plants, urban runoff, mountain runoff, and groundwater forced to the surface by shallow bedrock (Belitz and others, 2004, p. 1). Currently, the lower part of the Santa Ana River is a concrete channel from the city of Santa Ana to the City of Huntington Beach that usually does not contain water during dry periods (Santa Ana Watershed Project Authority, 2005, p. 28).

The Santa Ana Basin has one of the fastest growing populations in California and includes parts of Orange, San Bernardino, Riverside, and Los Angeles Counties. The watershed was home to about 5.1 million people in 2000 (Santa Ana Watershed Project Authority, 2002, table 11.1). Land use in the Santa Ana Basin is about 35 percent urban, 10 percent agricultural, and 55 percent open spaces that are primarily on steep mountain slopes. Population density for the entire study area is 1,500 people/mi^2; excluding the land area that is steep, the population density is about 3,000 people/mi^2 (Belitz and others, 2004, p. 3). The most densely populated part of the basin is in the city of Santa Ana, where there are as many as 20,000 people/mi^2.

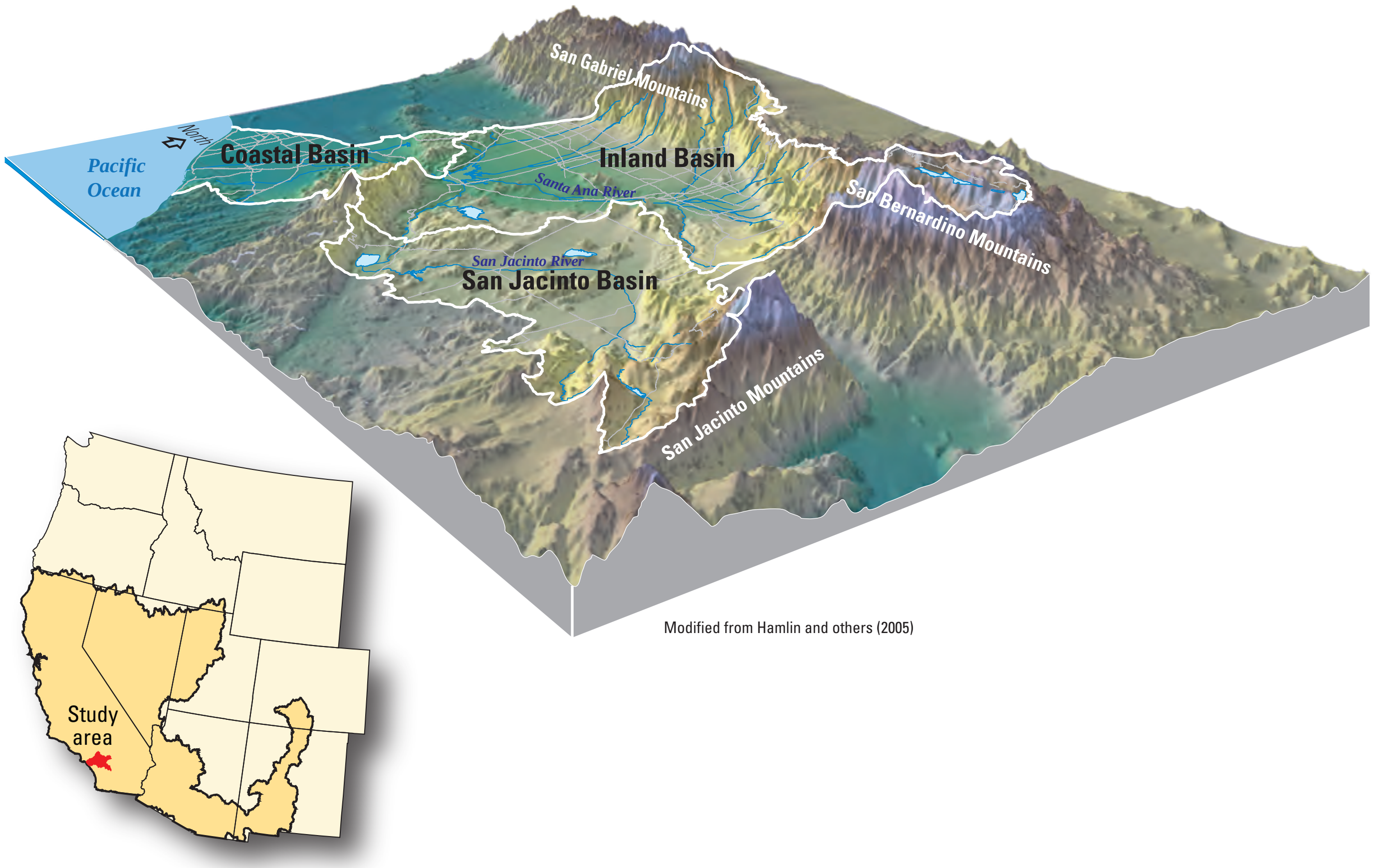


Figure 1. View to the northwest of the San Jacinto, Inland, and Coastal Basins in the Santa Ana drainage basin, California.

About 1.4 million acre-ft of water (467 billion gallons) was required to meet demand in the Santa Ana River watershed in 2000 (Santa Ana Watershed Project Authority, 2002, table 2.1). Estimated urban water use (70 percent of the estimated total use) exceeded estimated agricultural water use in the groundwater basins in 2000 based on county water-use data disaggregated to irrigated agricultural land and urban areas in the basins (McKinney and Anning, 2009, table 1). Groundwater pumped from the basins is the major water supply in the watershed, providing about two-thirds of the total water used (Belitz and others, 2004, p. 3). Water imported from northern California and the Colorado River also are important sources of water supply, accounting for 27 percent of the consumptive demand. Imported water is treated and delivered to consumers and is or has been used to recharge the aquifers. Projections are that the demand for water will increase by about 48 percent from 2000 to 2050, so that in 2050 the total water demand within the watershed will be about 2.1 million acre-ft (Santa Ana Watershed Project Authority, 2002, table 2.1).

Water Development History

Modifications to the natural surface-water system began in the early 1800s to supply water for irrigation in the San Bernardino area (Scott, 1977). San Bernardino is in the upper part of the Santa Ana River drainage basin, within the Inland Basin. Widespread irrigation began in 1848 (Scott, 1977) and by the 1880s, large tracts of land were dedicated to citrus and other crops, and diversions from the Santa Ana River and other streams were common. Groundwater in the Coastal Basin was used for irrigation beginning in the late 1800s. Around 1940, the urban population began to steadily increase along with water use for municipal purposes, while water use for irrigation began to decrease due to the urbanization of agricultural land (Scott, 1977).

Much of the runoff from the San Bernardino Mountains is diverted into storm-detention basins in or adjacent to stream channels along the mountain front. These facilities have been in operation since the early 1900s, and others have been constructed in other parts of the Inland, Coastal, and San Jacinto Basins to recharge the heavily used basin-fill aquifers. The groundwater recharge facilities near the San Bernardino Mountains began receiving imported water from the Colorado River via the Colorado Aqueduct in 1948 and from northern California through the State Water Project in the 1970s (Hardt and Freckleton, 1987; Reichard and others, 2003, p. 24). Imported Colorado River water is not currently used for artificial recharge and its use as a public supply in many areas of the Santa Ana Basin is limited because of its high concentration of dissolved solids—an average of 700 mg/L—and the effect of this level of salinity on treated wastewater discharge (Santa Ana Watershed Project Authority, 2002, p. 3-11). Pumping from the aquifer and additional sources of recharge have accelerated groundwater flow and the transport of dissolved constituents through the basin-fill aquifers in the San Jacinto, Inland, and Coastal Basins.

Hydrogeology

The dominant structural features in the Santa Ana Basin are its major fault zones. Motion along the San Andreas Fault Zone, which trends southeast-northwest along the western base of the San Bernardino Mountains and the eastern base of the San Gabriel Mountains, has caused the uplift of these generally east-west trending (transverse) mountain ranges. The San Jacinto Mountains are the result of uplift along both the San Andreas (eastern base) and San Jacinto (western base) Fault Zones. The Elsinore, Chino, and Whittier Fault Zones merge south of the Santa Ana River and bound the Santa Ana Mountains and Chino Hills (Morton and Miller, 2006, fig. 3). The Perris Block is an area between the Santa Ana Mountains and the San Jacinto Fault Zone of lower relief than the surrounding mountains where mainly Quaternary sediments discontinuously overlie bedrock. The adjacent basins have been filled with sediments eroded from these uplifted areas. The northwest-trending Newport-Inglewood Fault Zone extends into the Coastal Basin from offshore near Newport Beach. Faulting along the zone has formed the Newport-Inglewood Uplift, a series of folds visible as hills or mesas along the coast (Reichard and others, 2003, p. 5).

Groundwater flow in the basins is highly controlled by the geology, including the configuration of the surrounding and underlying bedrock and the extensive faulting that can create barriers to flow within the aquifer system. The basin-fill aquifers in the Santa Ana Basin consist primarily of Quaternary-age unconsolidated alluvium with interbedded marine sediments in the Coastal Basin (Dawson and others, 2003, p. 4). Unconfined conditions exist in most of the aquifer area; however, layers of fine-grained material, variable depth to bedrock, and the presence of faults can cause pressure zones where water flows toward (or to) the ground surface. Groundwater flow generally follows the topography and surface flow. Exceptions include areas where groundwater pumping has produced depressions in the water table and areas where faults act as barriers to flow.

Conceptual Understanding of the Groundwater Flow System

The three groundwater basins described in this section illustrate a wide range in groundwater and land-use conditions within the Santa Ana Basin. The groundwater system in the San Jacinto Basin is largely unconfined and land use is still primarily agricultural. The groundwater system in the Inland Basin also is predominantly unconfined and the major land use is now urban. Groundwater flow in the San Bernardino area of the Inland Basin, known as the Bunker Hill groundwater subbasin, is characterized by flow paths that originate along the mountain front and converge to a focused discharge area (Dawson and others, 2003, p. 58; Wildermuth Environmental, Inc., 2000, p. 3-4). The groundwater system in the mostly urban Coastal Basin consists of a relatively small unconfined recharge area and a relatively large confined area in which pumping is now the predominant form of groundwater discharge. Groundwater flow is generally characterized by areas of focused recharge and distributed discharge.

Some of the recharge to the groundwater basins occurs at facilities that receive and temporarily hold local stormwater and urban runoff, tertiary-treated municipal wastewater, or imported surface water. Such recharge facilities are more numerous in the Inland and San Jacinto Basins than in the Coastal Basin (fig. 2). Currently, flow in the Santa Ana River to the Coastal Basin consists predominantly of perennial base flow that is mostly treated wastewater discharged from municipal treatment plants in the Inland Basin (Mendez and Belitz, 2002) and intermittent stormflow that includes runoff from urban and agricultural land. Almost all of the flow in the Santa Ana River is diverted after it enters the Coastal Basin for recharge at engineered recharge facilities designed to replenish the basin-fill aquifer used for public supply. Treated wastewater from Coastal Basin communities is injected into the aquifer along the coast as a barrier to seawater intrusion, and starting recently (2008), is recharged at spreading basins near the Santa Ana River after advanced treatment. The remainder is discharged to the ocean. Groundwater discharge in the Santa Ana Basin is primarily by pumping, but also occurs as base flow to the Santa Ana River and its tributaries in some areas of the Inland Basin.

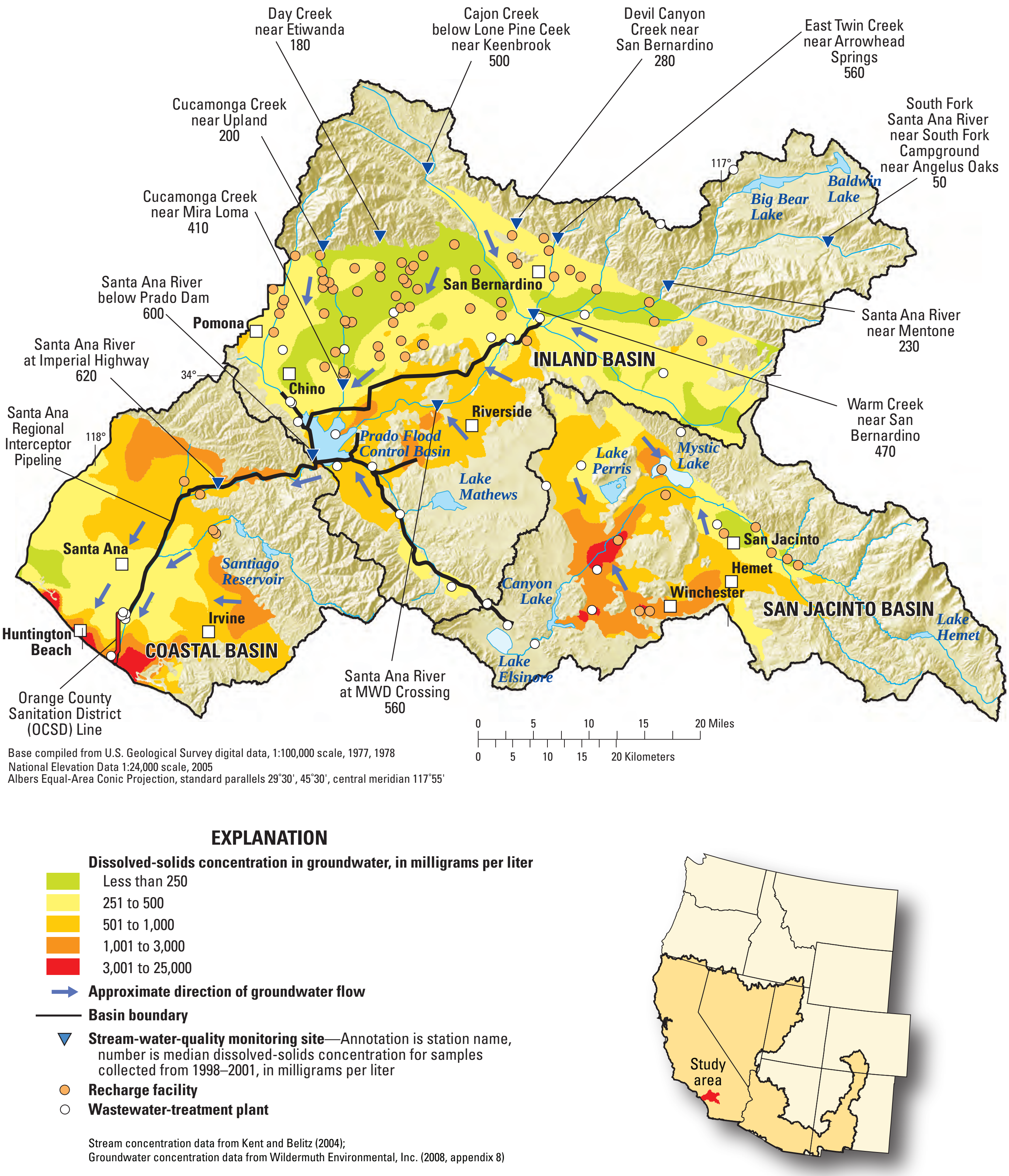


Figure 2. Dissolved-solids concentrations in groundwater in the San Jacinto, Inland, and Coastal Basins and in surface water in the Santa Ana Basin, California.

Groundwater Quality

In general, the quality of surface and groundwater in the Santa Ana Basin becomes progressively poorer— its content of dissolved mineral, chemical, and organic constituents increase as the water moves along flow paths. On the basis of this definition, the best quality water in the watershed is the flow in streams that drain the surrounding mountains and the parts of the groundwater system recharged by these flows. As the water flows away from the mountains, either on the surface or in the subsurface, its chemical composition is affected or changed by mineral dissolution, urban runoff, discharge of treated wastewater, agricultural operations, landscape irrigation, the use of surface water imported from the Colorado River and from northern California, and by enhanced recharge at engineered facilities. Groundwater quality can also be affected by accidents or activities at the land surface, such as spills and leaks of industrial solvents and by agricultural practices. Major water-quality issues in the Santa Ana Basin are elevated (above background) concentrations of dissolved solids, nitrate, perchlorate, and volatile organic compounds (VOCs) in groundwater.

The distribution of concentrations of dissolved solids and nitrate in groundwater within the Santa Ana Basin is monitored by local water suppliers (Wildermuth Environmental, Inc., 2008, p. 1-1). Concentrations of dissolved solids and nitrate in well water sampled from 1987 to 2006 were used to compute point statistics (Wildermuth Environmental, Inc., 2008, appendix B) that were then contoured to provide maps of the distribution of these constituents within the basin (figure 2 shows dissolved-solids concentrations). The distribution of dissolved solids in the Coastal Basin and parts of the Inland Basin (the confined parts of the Bunker Hill and Chino subbasins) was estimated from the results of analyses of water samples collected from intermediate depths, the zone from which water is generally withdrawn for public supply, as well as analyses of water from shallower depths.

Concentrations of dissolved solids in water in the basin-fill aquifers within the Santa Ana Basin are generally lowest (less than 250 mg/L) in areas recharged by surface runoff originating in the surrounding higher altitude mountain drainages (fig. 2). Concentrations can increase as groundwater moves away from the mountains because of urban and agricultural activities, alteration of the hydrologic cycle—including the importation of surface water to the basin—and from contact with natural sources of dissolved solids, such as salts released from geologic materials (Anning and others, 2007, p. 102). Desalting plants are used to reduce concentration of dissolved solids in groundwater in parts of the San Jacinto, Inland, and Coastal Basins. Brine generated at these facilities is typically transported through the Santa Ana Regional Interceptor pipeline to the Pacific Ocean for disposal.

Water samples were collected from 207 wells in the Santa Ana Basin from 1999 to 2001 as part of eight studies by the NAWQA Program to assess the occurrence and distribution of dissolved constituents in groundwater (Hamlin and others, 2002). These studies were designed to gain a better understanding of the used groundwater resource at different scales: (1) three studies were done to characterize water quality at a basin scale; (2) two studies focused on spatial and temporal variations in the chemical characteristics of water along selected flow paths; (3) two studies assessed aquifer susceptibility to VOC contamination; and (4) one study focused on an evaluation of the quality of shallow groundwater in an urban area. The aquifer susceptibility studies were done in collaboration with the California State Water Resources Control Board as part of the California Aquifer Susceptibility Program (Hamlin and others, 2002, p. 13).

Most of the samples collected for the NAWQA studies were analyzed for the field parameters temperature, specific conductance, dissolved oxygen content, and pH as well as for a wide suite of constituents, including the major ions, trace elements, radon, nutrients, dissolved organic carbon, pesticides, VOCs, and isotopes (oxygen-18, deuterium, and tritium) (Hamlin and others, 2002, appendixes). The samples collected for the aquifer susceptibility studies were analyzed for selected VOCs and isotopes (Shelton and others, 2001; Dawson and others, 2003; Hamlin and others, 2005). A summary of the physical properties and chemical characteristics of the water in wells sampled by NAWQA in the Santa Ana Basin is presented in table 1. The wells are divided into classes based on groundwater basin, aquifer confinement, and (or) depth. Information from local entities and studies and the findings of the several NAWQA studies are used to describe in this section of the report the general groundwater flow system, water-quality characteristics, and the potential effects of natural and human factors on groundwater quality in the San Jacinto, Inland, and Coastal groundwater basins within the larger Santa Ana Basin.

Table 1. Summary of physical and water-quality characteristics of wells in the Santa Ana Basin, California, sampled by the NAWQA Program, 1999–2001.

[ps, public-supply well; irr, irrigation well; mon, monitoring well; per mil, parts per thousand; pCi/L, picocuries per liter; mg/L, milligrams per liter; µg/L, micrograms per liter; <, less than; pesticide and volatile organic compound (VOC) detections include estimated values below the laboratory reporting level]

Well class	A	B	C	D	E	F	G	H	I
Number of wells	[1] 10 / [2] 13 depth to top of well screen less than 270 feet	[1] 10 / [2] 18 depth to top of well screen greater than 270 feet	9	17	[3] 15 depth to top of well screen less than 240 feet	[3] 14 depth to top of well screen greater than 240 feet	[4] 16 / [5] 35	[4] 26 / [5] 58	26
Predominant well type sampled	ps, irr	ps, irr	ps, mon	ps, mon	ps, irr	ps, irr	ps	ps	mon (shallow)
Ground-water basin	San Jacinto Basin	San Jacinto Basin	Bunker Hill subbasin	Bunker Hill subbasin	Inland Basin	Inland Basin	Coastal Basin	Coastal Basin	Coastal Basin
General aquifer confinement	Unconfined	Unconfined	Unconfined	Confined	Unconfined	Unconfined	Unconfined	Confined	Unconfined
General location	Basin wide	Basin wide	Closer to recharge facilities	Near San Jacinto Fault	Basin wide	Basin wide	Forebay area	Pressure area	Mostly pressure area
Land use	Agricultural and urban	Agricultural and urban	Mostly urban areas	Mostly urban areas	Mostly urban areas	Mostly urban areas	Urban	Urban	Urban
Physical characteristics									
Median well depth, feet	[1] 569 / [2] 625	[1] 960 / [2] 1,238	575	400	415	843	[4] 554 / [5] 714	[4] 876 / [5] 966	24
Median depth to top of well screen, feet	[1] 173 / [2] 170	[1] 420 / [2] 402	500	300	126	385	[4] 272 / [5] 342	[4] 338 / [5] 372	14
Median deuterium concentration, per mil	[2] -56.8	[2] -58.2	-68.9	-55.4	-56.5	-57.6	[5] -54.6	[5] -56.5	-48.3
Median tritium concentration, pCi/L	4.3	1.0	12.2	6.0	9.0	2.6	19.2	4.5	21.6
Water-quality characteristics									
Median dissolved-solids concentration, mg/L	504	460	226	354	338	276	583	436	2,390
Median nitrate concentration, mg/L	5.6	1.7	3.4	0.99	6.0	6.1	4.3	1.4	<0.05
Median dissolved-oxygen concentration, mg/L	5.8	0.4	7.2	0.7	7.6	8.0	1.9	1.6	0.9
Median arsenic concentration, µg/L	1.2	1.6	<0.9	1.8	1.1	0.85	1.2	1.4	3.0
Number of different pesticides detected	14	7	10	7	17	13	17	6	18
Number of pesticide detections	38	10	18	26	64	57	64	13	45
Percentage of wells where pesticides were detected	100%	30%	67%	47%	87%	79%	56%	23%	69%
Number of different VOCs detected	[2] 10	[2] 8	11	36	22	18	[5] 19	[5] 31	22
Number of VOC detections	[2] 31	[2] 21	16	94	82	49	[5] 109	[5] 114	68
Percentage of wells where VOCs were detected	[2] 84%	[2] 67%	56%	94%	87%	79%	[5] 80%	[5] 64%	88%

[1] The median depth to the top of the well screen in 20 wells sampled as part of study to characterize water quality in the San Jacinto Basin is 270 feet (Hamlin and others, 2005, p. 6).

[2] Includes wells sampled as part of the San Jacinto Aquifer Susceptibility Study (Hamlin and others, 2002, p. 13).

[3] The median depth to the top of the well screen in 29 wells sampled as part of study to characterize water quality in the Inland Basin is 240 feet (Hamlin and others, 2005, p. 6).

[4] Wells sampled as part of studies to characterize water quality in the Coastal Basin on basin and flow path scales (Hamlin and others, 2002, p. 12).

[5] Includes wells sampled as part of the Orange County Aquifer Susceptibility Study (Hamlin and others, 2002, p. 13).

San Jacinto Basin of the Santa Ana Basin

The San Jacinto Basin covers about 350 mi^2 in the Santa Ana drainage basin and contains Perris, Moreno, San Jacinto, and Menifee Valleys (fig. 3). Granitic and metamorphic rock "islands," the largest of which are the Lakeview Mountains, protrude through and underlie the unconsolidated sediment in the valleys (Wildermuth Environmental, Inc., 2000, p. 3-10). Excluding the consolidated rock protrusions, altitudes within the sediment-filled part of the basin range from about 1,400 to 2,000 ft, and reach 10,751 ft at the crest of the San Jacinto Mountains in the drainage area to the east. The San Jacinto Basin is bounded by fault zones on the east and west and by consolidated rock on the north and south. The San Jacinto Fault Zone separates the basin-fill deposits of the San Jacinto Valley from the San Jacinto Mountains (fig. 3), which are composed mostly of igneous and metamorphic rocks, and the San Timoteo Badlands, composed chiefly of Tertiary-age sedimentary rocks to the east (Schlehuber and others, 1989, p. 81).

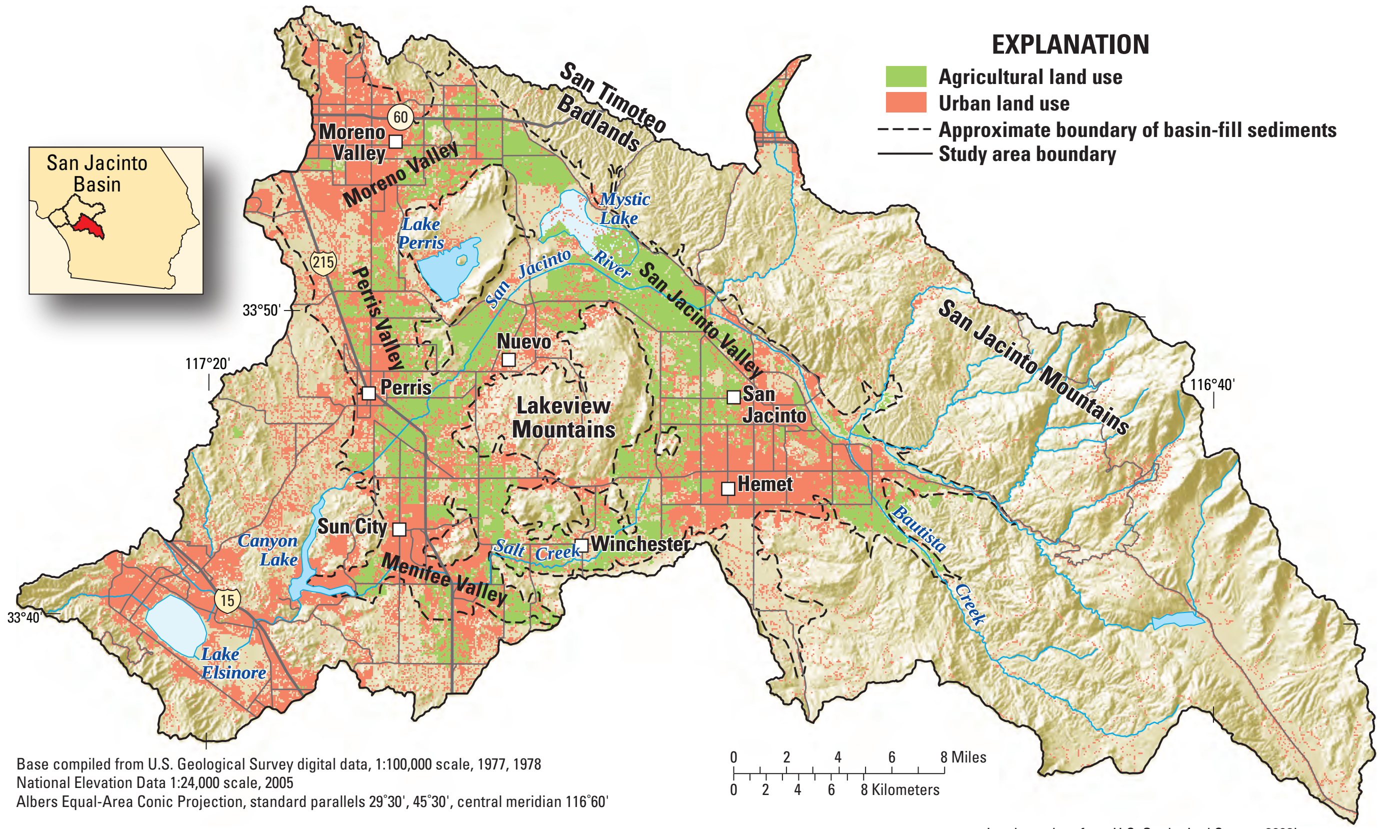
The San Jacinto Basin has a semiarid climate, with hot dry summers and cooler, wetter winters. Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 13.7 in. over the groundwater basin as a whole (McKinney and Anning, 2009, table 1). The San Jacinto Mountains receive up to 47 in. of precipitation annually. Most precipitation falls from October to March and most runoff in the basin results from winter storms. Drainage from the 800-mi^2 watershed is mostly to the San Jacinto River and its tributaries, which become ephemeral streams after entering the groundwater basin. Runoff from the watershed flows out of the San Jacinto Basin to Lake Elsinore and the Santa Ana River via Temescal Wash only during extremely wet periods. Water imported from northern California for public supply in the San Jacinto Basin and other parts of southern California is stored in Lake Perris.

Analysis of LandScan population data for 2005 (Oak Ridge National Laboratory, 2005) indicated a population of about 385,000 in the San Jacinto Basin (McKinney and Anning, 2009, table 1) and a population density of about 1,600 people/mi^2. About 34 percent of the basin was classified as urban and 42 percent as irrigated agricultural land in 2001 (U.S. Geological Survey, 2003b) (fig. 3). County-level water-use data for 2000 (U.S. Geological Survey, 2004) was disaggregated to a finer scale based on spatially distributed agricultural land use and population data in order to distribute water use on a basin scale (McKinney and Anning, 2009, p. 9). This method of determining water use in a basin may have a large uncertainty in the San Jacinto Basin because it is a relatively small part of Riverside County, which extends to the California/Arizona stateline and includes other large areas of agricultural land use. On the basis of this method of determining water use, the largest use of water in the San Jacinto Basin is the irrigation of crops. Groundwater pumped from wells is estimated to provide about 74 percent of public supply, the other major used of water in the basin.

Conceptual Understanding of the Groundwater System in the San Jacinto Basin

Geologically, the San Jacinto Basin can be characterized as a series of interconnected alluvium-filled valleys that are bounded by bedrock mountains and hills and cut by faults and bedrock highs (figs. 3 and 4). As part of a study to estimate the concentrations of dissolved solids and nitrate in groundwater in the Santa Ana watershed, the basin was subdivided into groundwater management zones that correspond to groundwater subbasins (fig. 3) on the basis of relatively impermeable boundaries such as bedrock and faults, bedrock constrictions, groundwater divides, and internal flow systems (Wildermuth Environmental, Inc., 2000, p. 3-12). The Canyon, San Jacinto-Upper Pressure, Hemet North, and Hemet South groundwater management zones were grouped together as the eastern subbasins, and the San Jacinto-Lower Pressure, Lakeview, Perris North, Perris South, and Menifee groundwater management zones were grouped together as the western subbasins. These groupings follow those of the groundwater management plans for the San Jacinto Basin (Eastern Municipal Water District, 2007a and 2007b) and are not based solely on similarities in the groundwater flow systems.

The Canyon, San Jacinto-Upper Pressure, and San Jacinto-Lower Pressure subbasins are west of the San Jacinto Mountains and between faults in the San Jacinto Fault Zone. Coincident with the San Jacinto Valley, a graben, this area consists of a forebay area in the southeast, where surface water recharges the groundwater basin, and a pressure area in the northwest, where groundwater occurs under confined conditions. The thickness of unconsolidated deposits in the graben is not known, but may exceed 5,000 ft (California Department of Water Resources, 2003a). A branch of the fault zone separates the Canyon and San Jacinto-Upper Pressure subbasins where it cuts through the basin fill and crosses the San Jacinto River. The low permeability of this fault zone causes groundwater to back up behind it, with water levels about 200 ft higher on the up gradient side than on the down gradient side. Water levels on the Canyon subbasin side of the fault zone in the early 1900s were high enough that groundwater discharged to the river channel (MacRostie and Dolcini, 1959).



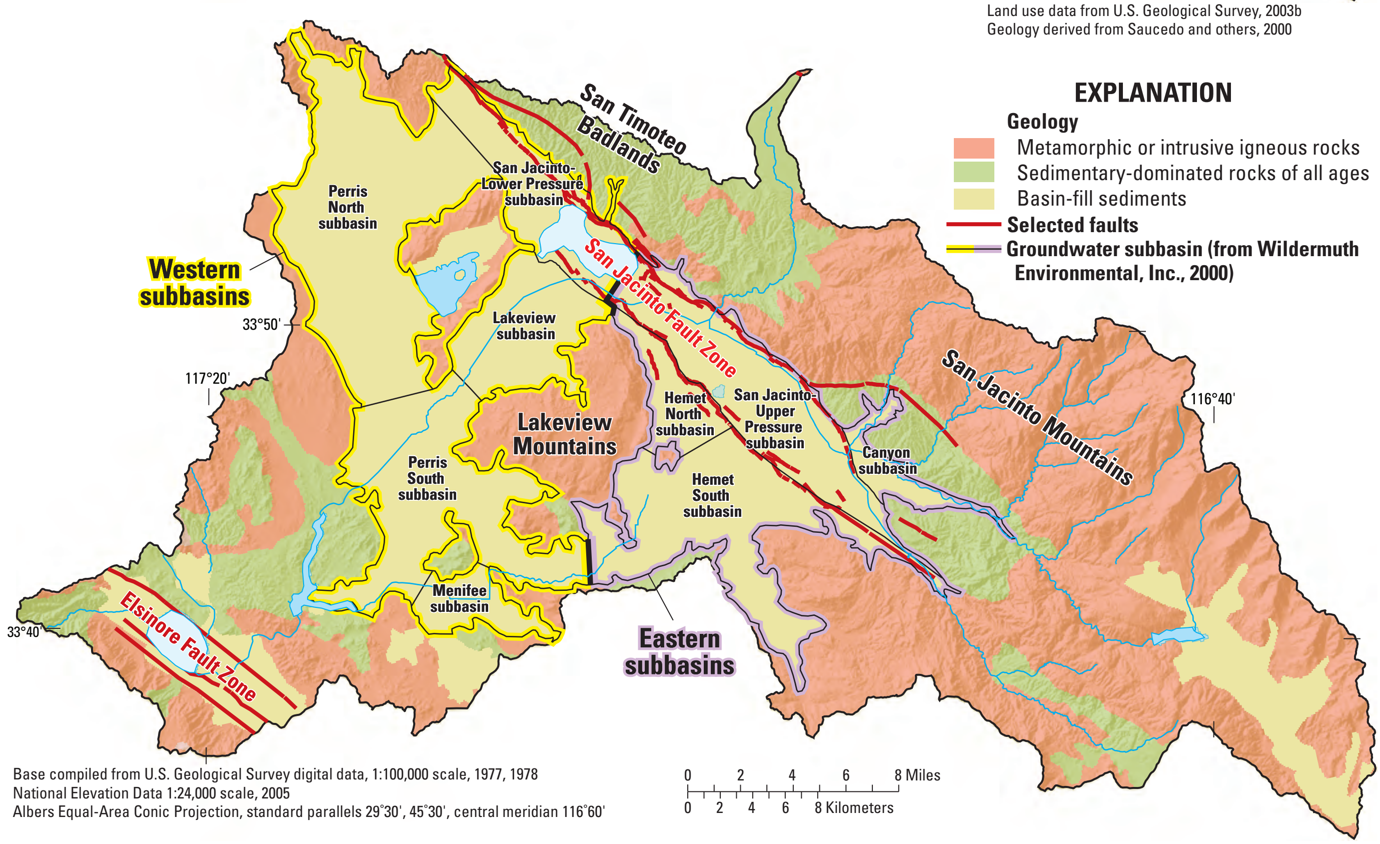


Figure 3. Physiography, land use, and generalized geology of the San Jacinto Basin, California.

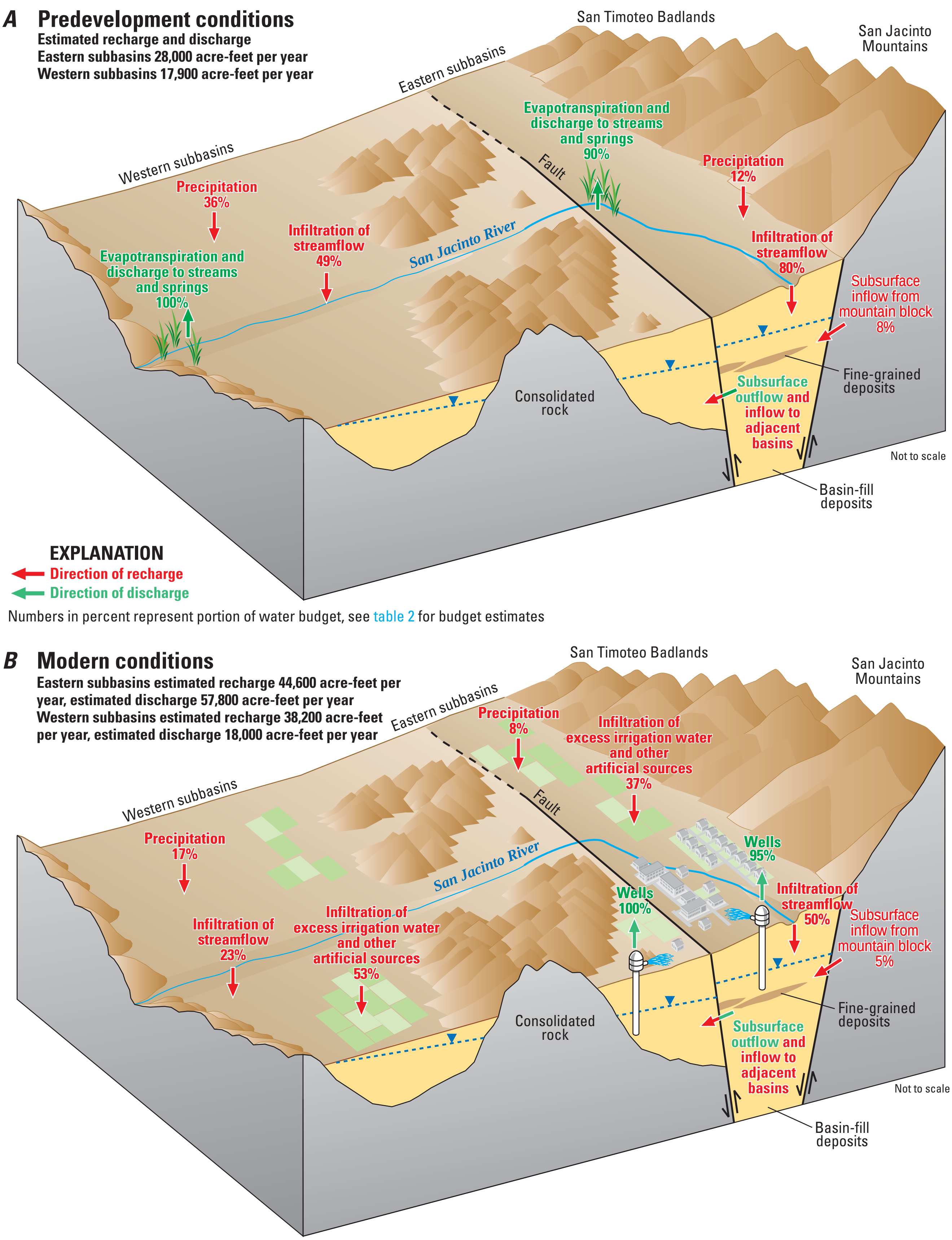


Figure 4. Generalized diagrams for the San Jacinto Basin, California, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Confined conditions caused by layers of fine-grained material and faults occur in much of the San Jacinto-Upper and Lower Pressure subbasins. A large area around the town of San Jacinto within which groundwater was under artesian pressure was noted between 1904 and 1915 where flowing wells were generally open to sand and gravel layers 100–200 ft below land surface (Waring, 1919, plate 3 and p. 27). Discharge from flowing wells and presumably by evapotranspiration occurred in this area before water levels were lowered by pumping. Silt and clay deposition in the tectonically subsiding basin contributes to the formation of the ephemeral Mystic Lake. This topographically low area with low permeability soils receives overflow from the San Jacinto River and from shallow perched groundwater, and probably some discharge from the confined groundwater system. Subsurface flow from the San Jacinto-Upper and Lower Pressure subbasins to the west is impeded by the western branch of the fault zone, so that under natural conditions, artesian pressure exists along the east side of the fault. Some groundwater was thought to move west to the Hemet North, Hemet South, and Lakeview subbasins under natural conditions (Wildermuth Environmental, Inc., 2000, p. 3-11).

The San Jacinto River enters the Canyon and San Jacinto-Upper Pressure subbasins from the mountains at the south end of the San Jacinto Valley and flows northwesterly. In most years, the river becomes ephemeral within these subbasins, mainly as a result of infiltration of the river's flow into the coarse-grained basin-fill deposits. The loss of water from the river channel is the source of most of the groundwater recharge to the Canyon and San Jacinto-Upper Pressure subbasins.

The thickness of unconsolidated deposits in the subbasins west of the San Jacinto Fault Zone (Hemet North, Hemet South, Lakeview, Perris North, Perris South, and Menifee subbasins) typically ranges from 200 to 1,000 ft. These basins are basically erosional depressions back-filled with alluvial sediments. The basin fill is thinnest adjacent to bedrock outcrops and is thickest along probable paleochannels incised in the underlying bedrock (Wildermuth Environmental, Inc., 2000, fig. 3-5). Groundwater in these subbasins generally occurs under unconfined conditions in the more permeable deposits. Depths to water, flow directions, recharge sources, and forms of discharge have changed in these subbasins due to water development in the area (Wildermuth Environmental, Inc., 2000, p. 3-11).

Water Budget and Groundwater Flow

Prior to development, most recharge to basin-fill aquifers in the San Jacinto Basin took place by infiltration of mountain streamflow, primarily the San Jacinto River near where it enters the basin, and runoff from precipitation on consolidated rocks within the basin and on the basin fill. Some groundwater moved through the subsurface across faults to recharge adjacent subbasins. Little information is available about groundwater conditions prior to development in the basin and estimates of recharge to and discharge from the aquifers presented in this report are intended only to provide a basis for comparison of change with development. Recharge to the eastern subbasins under predevelopment conditions is estimated to be about 28,000 acre-ft/yr: 3,400 acre-ft/yr from infiltration of precipitation on the basin; 22,500 acre-ft/yr from infiltration of streamflow in the San Jacinto River and its tributaries near the mountain front; and 2,100 acre-ft/yr from subsurface inflow from the mountain block (table 2). These estimates of natural recharge are based on average values determined for the area for the period 1958–2001 (Water Resources and Information Management Engineering Inc., 2003). In reality, recharge from these sources likely varies with extremes in annual precipitation. Groundwater recharge to the western subbasins under predevelopment conditions was estimated from long-term averages to be about 17,900 acre-ft/yr: 6,400 acre-ft/yr from infiltration of precipitation on the basin, 8,700 acre-ft/yr from infiltration of streamflow, and 2,800 acre-ft/yr from subsurface inflow from adjacent subbasins (Eastern Municipal Water District, 2005, appendix B, table 4-2).

The San Jacinto Basin is virtually closed to subsurface outflow because of low permeability consolidated rock surrounding the basin-fill deposits. Discharge of groundwater from the basin prior to development was primarily by evapotranspiration and by seepage to streams along the lower reaches of the San Jacinto River and Salt Creek in the western part of the basin (Wildermuth Environmental, Inc., 2000, p. 3-11). Subsurface flow between subbasins under predevelopment conditions occurred as a result of the larger volumes of natural recharge to the eastern subbasins spilling across faults or through bedrock constrictions into the western subbasins.

Water development in the San Jacinto Basin has significantly altered the groundwater systems and has caused changes in the groundwater budgets and flow directions. Under modern conditions in the basin, infiltration of excess irrigation water has become a large component of recharge to the basin-fill aquifers, and groundwater discharge is primarily withdrawals from wells (Wildermuth Environmental, Inc., 2000). Stable oxygen and hydrogen isotope ratios indicate that now groundwater in the basin is recharged from runoff derived from high-altitude precipitation in the San Jacinto Mountains, from low-altitude precipitation on the basin and hills, and from imported surface water (Williams and Rodoni, 1997, p. 1728). Aqueducts carrying State Project water from northern California and water from the Colorado River pass through the San Jacinto Basin. Lake Perris is adjacent to the Perris North subbasin and has served as a storage reservoir for northern California water since its construction in the 1970s. Both of these imported water sources have been utilized for irrigation and municipal supply. Recharge also occurs through seepage at retention basins, spreading basins, and percolation ponds filled with stormwater, imported surface water, and treated wastewater.

Table 2. Estimated groundwater budget for the basin-fill aquifer system in the San Jacinto Basin, California, under predevelopment and modern conditions.

[All values are in acre-feet per year. Estimates of natural recharge that are assumed to represent predevelopment conditions in the eastern subbasins are based on 1958–2001 averages determined for the area (Water Resources and Information Management Engineering Inc., 2003) and those in the western subbasins are from long term averages listed in a groundwater management plan for the west San Jacinto groundwater basin adopted in 1995 (Eastern Municipal Water District, 2005, appendix B, table 4-2), unless footnoted. The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components.]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Eastern subbasins			
Budget component	**Estimated recharge**		
Infiltration of precipitation on basin	3,400	3,400	0
Infiltration of streamflow	22,500	22,500	0
Subsurface inflow from mountain block	2,100	2,100	0
Infiltration of excess irrigation water and other artificial sources	0	16,600	16,600
Total recharge	**28,000**	**44,600**	**16,600**
Budget component	**Estimated discharge**		
Evapotranspiration and discharge to streams and springs	[1] 25,200	0	-25,200
Subsurface outflow to adjacent subbasins	[2] 2,800	[2] 2,800	0
Well withdrawals	0	55,000	55,000
Total discharge	**28,000**	**57,800**	**29,800**
Change in storage (total recharge minus total discharge)	0	-13,200	-13,200
Western subbasins			
Budget component	**Estimated recharge**		
Infiltration of precipitation on basin	6,400	6,400	0
Infiltration of streamflow	8,700	8,700	0
Subsurface inflow from adjacent subbasins	[2] 2,800	[2] 2,800	0
Infiltration of excess irrigation water and other artificial sources	0	[3] 20,300	20,300
Total recharge	**17,900**	**38,200**	**20,300**
Budget component	**Estimated discharge**		
Evapotranspiration and discharge to streams and springs	[1] 17,900	0	-17,900
Subsurface outflow to adjacent subbasins	0	0	0
Well withdrawals	0	[4] 18,000	18,000
Total discharge	**17,900**	**18,000**	**100**
Change in storage (total recharge minus total discharge)	0	20,200	20,200

[1] Assumed to be the difference between estimated predevelopment recharge and estimated discharge from subsurface outflow to adjacent subbasins.

[2] Listed as subsurface inflow from mountain boundaries in 1995 groundwater management plan (Eastern Municipal Water District, 2005, appendix B, table 4-2) and assumed to be mainly subsurface outflow from the eastern subbasins to the western subbasins.

[3] Includes only irrigation component of estimated recharge from deep percolation of applied water listed in the 1995 groundwater management plan (Eastern Municipal Water District, 2005, appendix B, table 4-2).

[4] Average well withdrawals from 1985–2004 (Metropolitan Water District of Southern California, 2007, table 17-3).

Prior to the importation of surface water to the San Jacinto Basin, water pumped from the basin-fill aquifer was a major source of public use and irrigation supply. The 1958–2001 average withdrawal rate from wells in the eastern subbasins was about 55,000 acre-ft/yr and estimated average recharge from excess irrigation water and artificial recharge sources was about 16,600 acre-ft/yr (Water Resources and Information Management Engineering Inc., 2003). Coupled with estimates of natural recharge that are assumed to be the same as under predevelopment conditions (28,000 acre-ft/yr) and subsurface outflow to the western subbasins (an estimated 2,800 acre-ft/yr), the difference in estimated recharge and discharge in the eastern subbasins is a deficit of about 13,000 acre-ft/yr. Although this is a rough estimate of overdraft that should not be used to calculate an accumulated volume of water removed from storage, it corresponds to lowered water levels and changes in flow directions in the eastern subbasins. The general directions of groundwater flow in 1973 and in 2006 in the San Jacinto Basin are shown on fig. 5.

Groundwater production in the western subbasins is monitored as part of the groundwater management plan and totaled about 18,700 acre-ft in 2004 (Eastern Municipal Water District, 2005, table 3.3), 16,800 acre-ft in 2005, and 23,100 acre-ft in 2006 (Eastern Municipal Water District, 2007a, table 3-5). Average production from wells in the western subbasins for 1985–2004 is listed at about 18,000 acre-ft/yr by the Metropolitan Water District of Southern California (2007, table 17-3) and is limited mainly by groundwater quality (elevated concentrations of dissolved solids) in parts of the area. Groundwater recharge to the western subbasins under modern conditions has increased due mostly to the infiltration of excess irrigation water and is estimated to be about 20,000 acre-ft/yr more than discharge based on estimates listed in table 2. These estimates were mostly from the groundwater management plan for the western part of the San Jacinto Basin adopted in 1995 (Eastern Municipal Water District, 2005, appendix B, table 4-2). Recharge from the infiltration of imported surface water and treated wastewater through storage ponds and reservoirs in the western subbasins is localized and variable (Kaehler and Belitz, 2003, p. 3). In this budget analysis, it is assumed that infiltration of excess irrigation water decreases as infiltration from these other sources increases due to land-use changes. Much uncertainty exists in the groundwater budgets for the San Jacinto Basin, and more information and analysis are needed to arrive at a better estimate for recharge and discharge components.

Water levels generally declined in the western subbasins from 1945 to the mid 1970s due to withdrawals from wells and periods of below-normal precipitation (Eastern Municipal Water District, 2005, appendix B, p. 4-6). Between 1973 and 2006, however, water levels typically rose in the Perris North, Perris South, and Menifee subbasins (fig. 5). The rise in water levels in these areas is attributed to decreased withdrawals and additional recharge of excess irrigation water, imported surface water, and treated wastewater to the basin-fill aquifer.

Effects of Natural and Human Factors on Groundwater Quality

The amount and source of recharge to the basin-fill deposits in the San Jacinto Basin affects the quality of the groundwater. Subbasins that receive a large percentage of recharge from mountain-front runoff carried by the San Jacinto River have groundwater quality that is typically similar to that of the recharged water. Some areas in the basin receive little recharge and others receive a large percentage from excess irrigation water. The concentration of dissolved minerals in this groundwater is generally elevated above background levels as a result of evapotranspiration. Other factors that affect groundwater quality in the basin are infiltration of water from overlying agricultural and urban areas, water/aquifer matrix reactions, movement of poorer quality water induced by withdrawals from wells, and the extensive use of imported water.

Water samples are collected annually from selected private, public-supply, and irrigation wells as part of water-quality monitoring programs in the San Jacinto Basin. In 2006, 102 wells were sampled in the western subbasins and 125 wells were sampled in the eastern subbasins (Eastern Municipal Water District, 2007a, p. 3-6 and 2007b, p. 27). NAWQA studies were done in 2001 in the San Jacinto Basin to help assess general water-quality conditions (samples were collected from 18 wells used for public supply and 5 wells used for irrigation) and to evaluate the susceptibility of public-supply wells to contamination by VOCs (samples were collected from 11 wells) (Hamlin and others, 2002) (fig. 6). Wells sampled by the NAWQA Program in the basin ranged in depth from 328 to 1,720 ft.

General Water-Quality Characteristics and Natural Factors

Groundwater quality in the San Jacinto Basin, in terms of the concentration of dissolved minerals, varies with the recharge source and location within the basin. The source of most recharge is runoff from the San Jacinto Mountains, a calcium-bicarbonate type water with low concentrations of dissolved solids (about 100 mg/L) (Anning and others, 2007, p. 103). Dissolved-solids concentrations in water from wells near the San Jacinto River and associated engineered recharge facilities near the mountain front in the Canyon and San Jacinto Upper Pressure subbasins were mostly less than about 500 mg/L (fig. 2). Dissolved-solids concentrations in water from wells in the Hemet North, Hemet South, San Jacinto Lower Pressure, Lakeview, and Perris North subbasins were generally higher, but usually less than 1,000 mg/L.

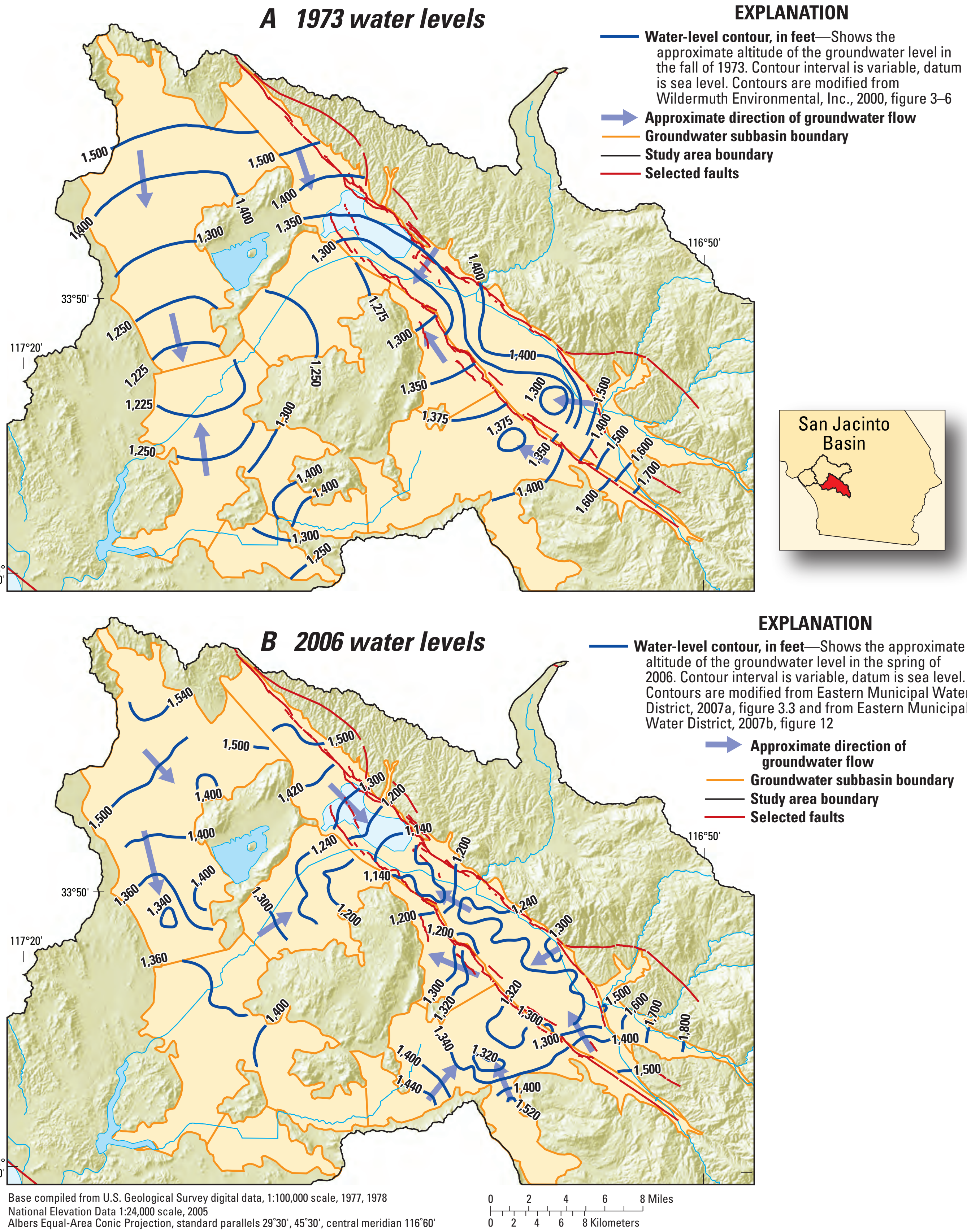


Figure 5. Groundwater levels and generalized flow directions in (*A*) 1973 and in (*B*) 2006 in the San Jacinto Basin, California.

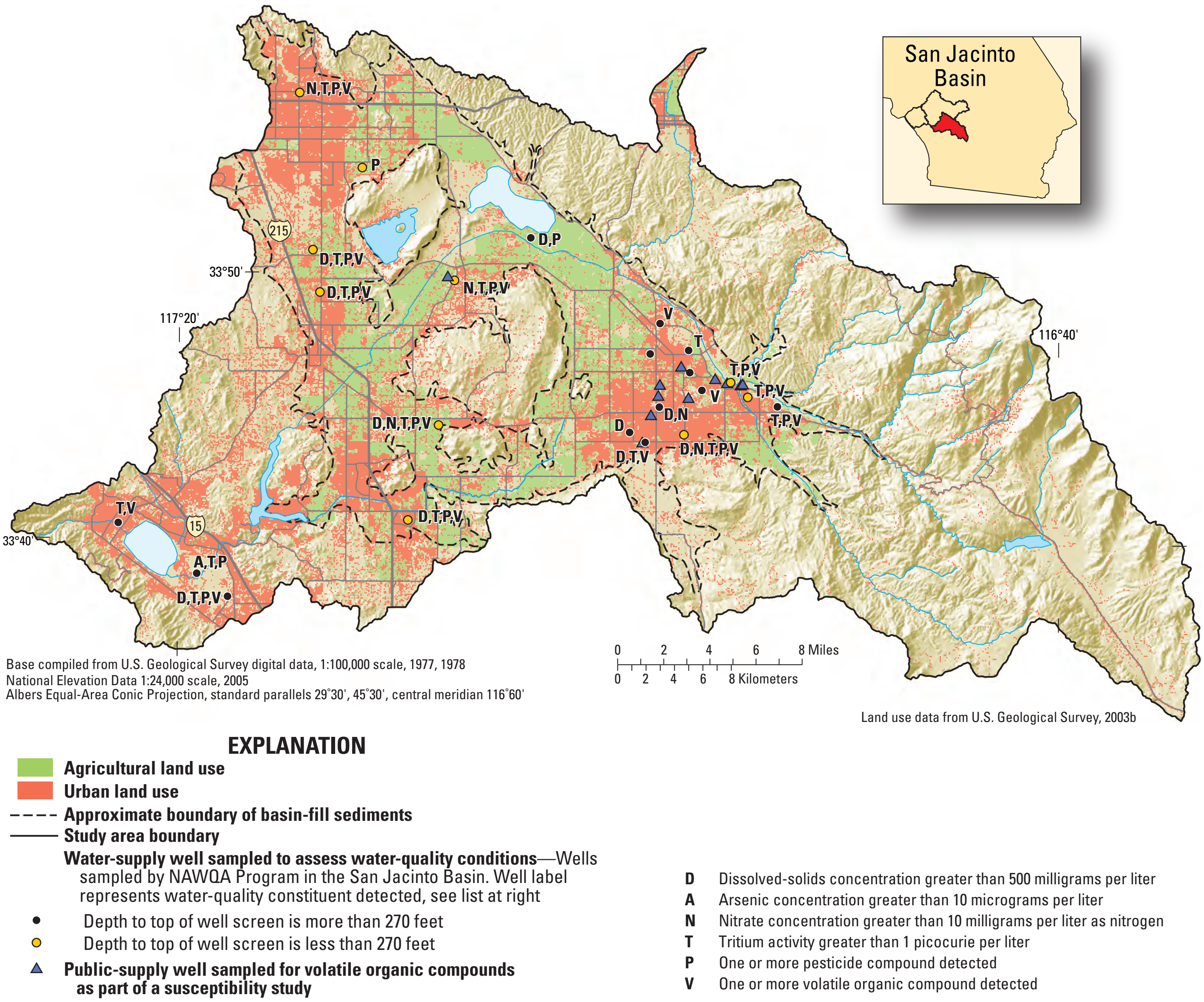


Figure 6. Locations of and chemical characteristics of water in wells sampled in the San Jacinto Basin, California, by the NAWQA Program, 2001.

These subbasins contain groundwater primarily recharged from mountain runoff and from excess irrigation. Concentrations in water from wells in the Perris South and Menifee subbasins, which are furthest from major sources of mountain-front recharge, were mostly greater than 1,000 mg/L (Kaehler and Belitz, 2003).

Before appreciable use of groundwater began, water levels in the lower parts of the San Jacinto Basin were near or at the ground surface, resulting in evapotranspiration and naturally high concentrations of dissolved solids in groundwater. Because of issues related to the use of high-salinity water, relatively few water-supply wells have been drilled in areas where the groundwater has concentrations of dissolved solids greater than 1,000 mg/L (Burton and others, 1996; Kaehler and others, 1998; Kaehler and Belitz, 2003). The highest concentration of dissolved solids measured in groundwater sampled in 2006, about 15,000 mg/L, was from a well in the Perris South subbasin (Eastern Municipal Water District, 2007a, table 3-3). Saline groundwater is pumped from wells and then treated by reverse osmosis at two desalination facilities in the Menifee Valley (Anning and others, 2007, p. 103; Eastern Municipal Water District, 2005, p. 88). The dissolved-solids concentration of treated wastewater recharged to the western subbasins is actually less than that of the local groundwater (Burton and others, 1996, p. 2).

Vertical differences in dissolved-solids and major ion concentrations were observed in water from a nested well site near the boundary between the Hemet South and Perris South subbasins (Kaehler and others, 1998, p. 32). Dissolved-solids concentrations in water sampled from 3 wells completed in shallower parts of the basin-fill aquifer (screened intervals from 72 to 236 ft below land surface) ranged from 1,620 to 3,380 mg/L. Water sampled from 2 wells screened at depths from 450 to 460 ft and from 630 to 640 ft below land surface had dissolved-solids concentrations of 595 and 483 mg/L, respectively. Evaporation and reactions with the aquifer matrix are likely causes of relatively high concentrations of dissolved solids in water in the shallower parts of the aquifer (Kaehler and others, 1998, p. 32).

The trace elements arsenic and uranium are present in the sampled groundwater, but are not contaminants of concern in the San Jacinto Basin. Concentrations of dissolved arsenic ranged from 0.3 to 19.4 µg/L, with a median of 1.2 µg/L, in samples collected by NAWQA from 23 water-supply wells in the San Jacinto Basin (Hamlin and others, 2002, appendix 6). Only one sample had an arsenic concentration greater than the drinking-water standard of 10 µg/L (U.S. Environmental Protection Agency, 2008). Concentrations of dissolved uranium in samples from the 23 water-supply wells ranged from less than 0.02 to 15.7 µg/L with a median value of 2.1 µg/L (Hamlin and others, 2002, appendix 8), all less than the drinking-water standard of 30 µg/L (U.S. Environmental Protection Agency, 2008).

The dissolved oxygen content of groundwater provides an indication of the oxidation-reduction (redox) environment within the aquifer, which affects the mobility of many constituents (McMahon and Chapelle, 2008). Suboxic and anoxic redox conditions are associated with concentrations of dissolved oxygen less than 0.5 mg/L and with nitrate reduction and(or) other reduction processes in the case of anoxic conditions (McMahon and Chapelle, 2008, table 1). Samples collected by NAWQA from wells in which the top of the screen was deeper than a median value of 270 ft below land surface had a median concentration of dissolved oxygen and nitrate (as nitrogen) of 0.4 mg/L and 1.7 mg/L, respectively, compared to wells with shallower open intervals in which the median concentration of dissolved oxygen and nitrate (as nitrogen) was 5.8 mg/L and 5.6 mg/L, respectively (table 1). These results suggest that the geochemical environment becomes more reducing with depth, which would cause consumption of dissolved oxygen and nitrate by biochemical processes.

Tritium, an isotope of hydrogen that is incorporated into the water molecule, is an indicator of young groundwater (see Section 1 of this report for a discussion of groundwater age and environmental tracers). Tritium was detected in 15 of 23 (65 percent) of the NAWQA sampled wells in the basin at activities greater than 1 pCi/L, indicating that a component of the groundwater in most of the samples was recharged since the early 1950s (young water) (Hamlin and others, 2005, p. 5). Shallow groundwater has higher tritium activities than deeper water (table 1) and therefore is assumed to be younger than the deeper water. This finding indicates that a major component of recharge is from the overlying land surface rather than by lateral flow from more distant areas.

Potential Effects of Human Factors

In addition to the natural factors described above, agricultural and urban land uses and activities and the extensive use of imported water also affect groundwater quality in the San Jacinto Basin. Withdrawals from wells have altered groundwater-flow directions in some areas, allowing new sources of recharge and the movement of poor quality groundwater to have a greater effect on water quality.

Groundwater monitoring programs in 2006 measured concentrations of nitrate (as nitrogen) ranging from not detected to 30 mg/L in the San Jacinto Basin (Eastern Municipal Water District, 2007a, table 3-3 and 2007b, table 9). The drinking-water standard for nitrate (as nitrogen) is 10 mg/L (U.S. Environmental Protection Agency, 2008). Water sampled from 12 of 58 public-supply wells (21 percent) in the basin had concentrations of nitrate (as nitrogen) greater than 10 mg/L (California Department of Water Resources, 2003a). Nitrate (as nitrogen) concentrations exceeded 10 mg/L in 5 of 23 of the NAWQA samples (22 percent) from wells used for municipal supply and irrigation (Hamlin and others, 2002). Contours of computed statistics representing concentrations of nitrate (as nitrogen) in groundwater in the San Jacinto Basin show areas with concentrations greater than 10 mg/L in the San Jacinto Upper Pressure, Hemet South, Lakeside, Perris North, and Perris South subbasins (Wildermuth Environmental, Inc., 2008, appendix B). Potential sources of nitrate in the basin are the infiltration of water affected by agricultural practices and wastewater from animal feeding facilities, septic tanks, and from municipal wastewater treatment plants (Rees and others, 1995).

Pesticides were detected in 14 of 23 of the NAWQA sampled wells (61 percent) at concentrations that were much lower than applicable drinking-water standards (Hamlin and others, 2002, appendix 9F). The most commonly detected pesticides in groundwater from the San Jacinto Basin were atrazine (9 samples), simazine (8 samples), and atrazine degradates (8 samples). VOCs were detected in 24 of 34 of the NAWQA sampled wells (71 percent) at low concentrations (Hamlin and others, 2002, appendixes 11G and 11H) well below applicable drinking-water standards. The most commonly detected VOCs were chloroform (21 samples) and perchloroethene (PCE, 7 samples).

Pesticides and VOCs were detected more frequently in water from 10 shallower wells (where the top of the well screen is within 270 ft of land surface) than in water from 10 deeper wells sampled by NAWQA (Hamlin and others, 2005, fig. 14) (fig. 7, table 1). One or more pesticides were detected in all of the wells and one or more VOCs were detected in 84 percent of the wells in which the top of the well screen was shallower than the median value of 270 ft below land surface (table 1). This finding compares to the detection of one or more pesticides in 30 percent and one or more VOCs in 67 percent of the wells with deeper open intervals. Pesticides were detected in more than 72 percent of the samples containing young (post-1950 recharge) water, but were detected in only 20 percent of the samples made up of older water (Hamlin and others, 2005, p. 17). Similarly, VOCs were detected in 83 percent of the samples containing young water and in 20 percent of the samples made up of older water. The differences in detection frequency based on depth are comparable to the differences based on age. The higher detection frequencies in shallower and younger groundwater suggest that these compounds have been introduced to the aquifer system since the early 1950s. Because the aquifers are generally unconfined, they are susceptible to contamination from sources at the land surface. The potential for contamination of groundwater by VOCs can be expected to increase as urban development in the basin continues.

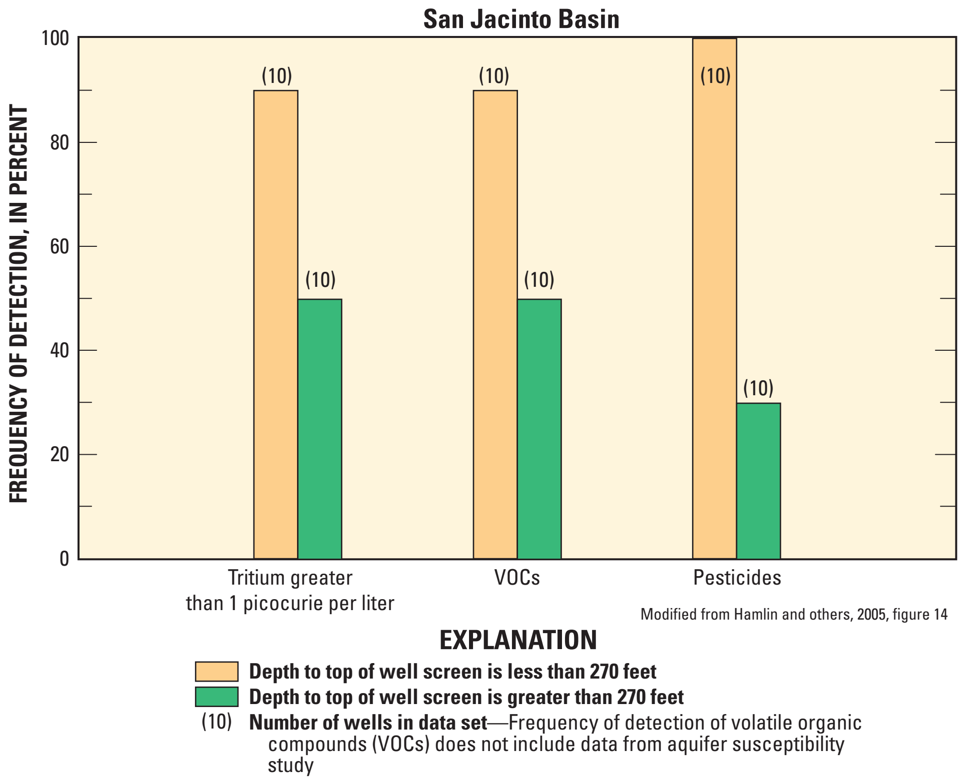


Figure 7. Detection frequencies of tritium, volatile organic compounds, and pesticides in water samples from wells in the San Jacinto Basin, California.

Inland Basin of the Santa Ana Basin

The Inland Basin covers about 655 mi^2 in the central part of the Santa Ana drainage basin (fig. 8) and is also called the Upper Santa Ana Valley Groundwater Basin (California Department of Water Resources, 2003b, p. 148). It is bounded by the San Bernardino Mountains on the northeast; the San Gabriel Mountains on the northwest; the Chino Hills and Santa Ana Mountains to the west; and various hills and relatively low altitude mountains to the south. Altitudes in the 1,484-mi^2 drainage basin range from 486 ft at the Prado Flood Control Basin dam to 11,499 ft in the San Bernardino Mountains. Alluvial fans extend from the mountain fronts into the basin and the watershed eventually drains to the Santa Ana River.

The Inland Basin has a semiarid climate, with hot dry summers and cooler, wetter winters. Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 16.4 in. over the groundwater basin as a whole (McKinney and Anning, 2009, table 1). Parts of the San Bernardino Mountains receive more than 50 in. of precipitation during the year. Most precipitation falls from October to March and most runoff results from winter storms. Natural recharge to the Inland Basin groundwater system is primarily from the infiltration of runoff originating in the mountains.

Analysis of LandScan population data for 2005 (Oak Ridge National Laboratory, 2005) indicated a population of 2.1 million in the groundwater basin (McKinney and Anning, 2009, table 1) and a population density of about 3,200 people/mi^2. About 68 percent of the basin was classified as urban and about 6 percent as irrigated agriculture in 2001 (U.S. Geological Survey, 2003b) (fig. 8). The largest use of water in the basin is for public supply, which was estimated using county-level water-use data disaggregated to a finer scale (McKinney and Anning, 2009, p. 9) at about 504,000 acre-ft in 2000 with about 74 percent supplied by groundwater. Water use for irrigation was estimated to be about 148,000 acre-ft in 2000, about 23 percent of which was pumped from wells. This method of determining water use in the basin may have a larger uncertainty because the Inland Basin is a relatively small part of San Bernardino County.

Conceptual Understanding of the Groundwater System in the Inland Basin

The Inland Basin is bounded on the east by the San Andreas Fault, which lies along the base of the San Bernardino Mountains; on the north by the Cucamonga Fault Zone, which follows the base of the San Gabriel Mountains; and on the west by the Chino Fault, which separates the basin from the Chino Hills (fig. 8). Other faults and consolidated rock constrictions divide the Inland Basin into groundwater subbasins. These interior faults locally restrict groundwater flow and control the location of natural groundwater discharge. The Chino, Cucamonga, Rialto-Colton, Bunker Hill, Yucaipa, San Timoteo, Riverside-Arlington, and Temescal groundwater subbasins within the Inland Basin are shown on figure 8 and are condensed from groundwater management zones used for water-quality monitoring within the basin (Wildermuth Environmental, Inc., 2008, fig. 1-1).

The groundwater basins are generally unconfined near the mountain fronts where precipitation and mountain runoff is distributed and recharged through natural streambeds and engineered recharge facilities. Confined conditions typically occur down gradient from the mountain fronts and at greater depths due to finer grained deposits interlayered with sand and gravel. The entire surface-water outflow from the Inland Basin is stored in the Prado Flood Control Basin before flowing in the Santa Ana River into the Coastal Basin. The Bunker Hill and Chino subbasins are the two largest subbasins and account for more than 50 percent of the basin-fill area in the Inland Basin (17 and 34 percent, respectively). The groundwater systems and water quality of the Bunker Hill and Chino subbasins are described in this section of the report because of their relatively large areas and relatively well understood flow systems. Thus, conceptual models of groundwater systems in the Inland Basin are based primarily on those of the Bunker Hill and Chino subbasins, which are discussed separately because of substantial differences in their hydrogeologic settings. Other subbasins, such as the Rialto-Colton subbasin, also have been extensively investigated (Woolfenden and Kadhim, 1997; Woolfenden and Koczot, 2001), but have characteristics that are represented by either the Bunker Hill or Chino subbasins.

The Bunker Hill subbasin, which covers 112 mi^2 in the San Bernardino area, is bounded by the San Bernardino Mountains and the San Jacinto Fault Zone in the northeastern part of the Inland Basin. It has a large mountain drainage area (466 mi^2) that contributes water to the subbasin. The sediments in the Bunker Hill subbasin generally consist of coarse-grained unconsolidated alluvial fan and stream deposits near the mountain fronts that become layered with finer grained material further away from the mountains. Although layers could be correlated only over short distances, Dutcher and Garrett (1963, plate 7) divided the basin-fill deposits into three water-bearing zones separated by intervals of primarily clay and silt (fig. 9). The thin Quaternary-age stream-channel deposits are among the most permeable sediments in the subbasin and allow large seepage losses from streams. Hydraulic conductivity values for these deposits range from about 40 to 100 ft/d (Dutcher and Garrett, 1963, p. 51-56). Basin-fill deposits near the land surface, but away from the streams, are generally less permeable and act to confine deeper groundwater.

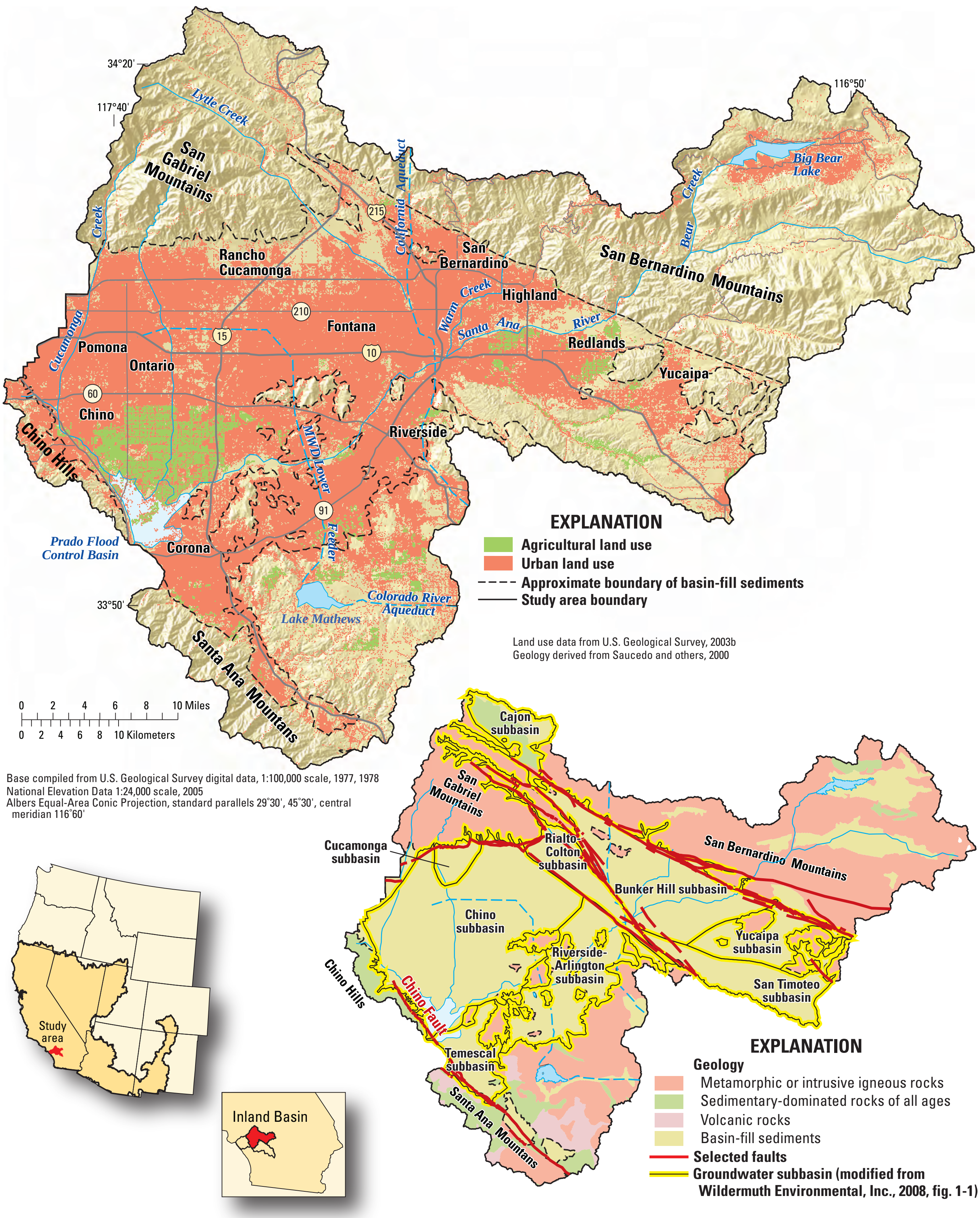


Figure 8. Physiography, land use, and generalized geology of the Santa Ana Inland Basin, California.

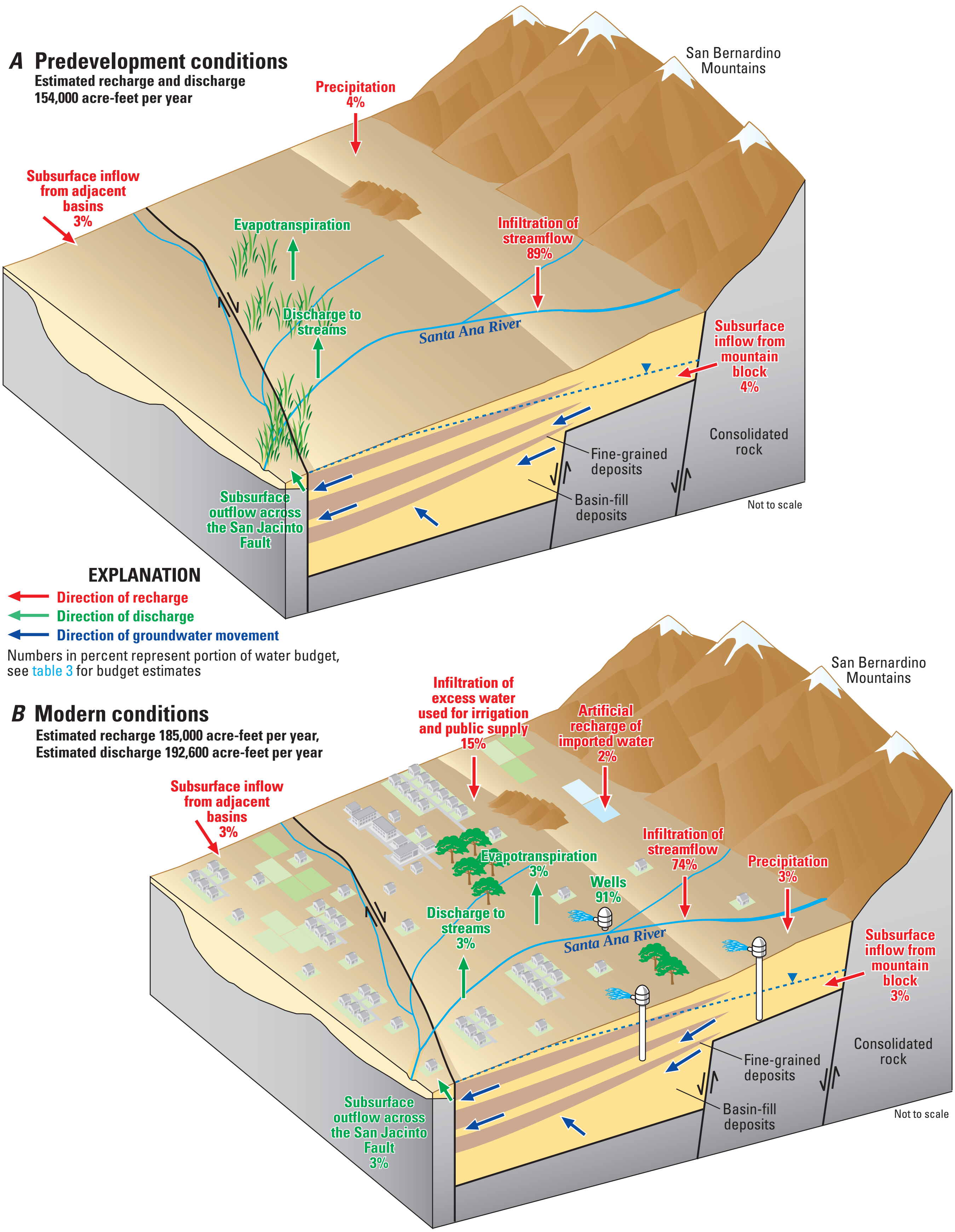


Figure 9. Generalized diagrams for the Bunker Hill subbasin, California, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

The upper and middle water-bearing zones contain layers of well-sorted sand and gravel that provide most of the water to public-supply and agricultural wells in the Bunker Hill subbasin. The hydraulic conductivity of the upper water-bearing zone is estimated to be about 60 ft/d on the basis of transmissivity values used in a numerical groundwater flow model of the subbasin (Danskin and others, 2006, fig. 40A). The upper and middle water-bearing zones are separated by as much as 300 ft of interbedded silt, clay, and sand in the central part of the subbasin. Although not as permeable as the adjacent water-bearing zones, the confining material does yield significant quantities of water to wells. As a result, most production wells are open to one or both of the water-bearing zones and to the intervening confining zone. The lower water-bearing zone is composed of poorly consolidated to partly cemented older Quaternary-age basin fill or older semiconsolidated to consolidated Tertiary-age deposits.

The Chino subbasin covers about 222 mi^2, but unlike the Bunker Hill subbasin, has a mountain drainage area of only 62 mi^2. It slopes from north to south towards the Santa Ana River. The upgradient Cucamonga subbasin receives much of the runoff from the San Gabriel Mountains as recharge, but faults cutting through the basin-fill deposits restrict groundwater flow to the Chino subbasin. Other faults bounding and within the subbasin impede the movement of groundwater to varying degrees. A detailed description of the geology and hydrostratigraphy of the basin is provided by the Chino Basin Watermaster (Wildermuth Environmental, Inc., 2007a).

The Chino subbasin is filled with an average of about 500 ft of unconsolidated sediment eroded from the surrounding mountains during the Pleistocene Epoch that is overlain by thinner, more recent flood plain and alluvial fan deposits associated with the mountain-draining streams and the Santa Ana River (California Department of Water Resources, 2003c). In the deepest parts of the subbasin, these sediments are greater than 1,000 ft thick. Laterally extensive and continuous layers of permeable sediment or of confining material are not present as a result of the alluvial fan depositional environment. The unconsolidated basin-fill deposits overlie semiconsolidated to consolidated Tertiary-age sediment, which overlie an irregular surface of igneous, metamorphic, and sedimentary rocks that are considered to be relatively impermeable.

The upper part of the unconsolidated basin-fill deposits is generally coarse grained and permeable. It ranges from having a thick unsaturated zone in the northern and eastern unconfined parts of the subbasin to being almost fully saturated in the southern and western semiconfined to confined parts, where the shallow deposits contain a larger fraction of silt and clay layers (fig. 10). Groundwater moving from higher altitude recharge areas in the northern and eastern parts of the subbasin becomes confined beneath fine-grained sediments in the western and southwestern parts of the subbasin. These fine-grained sediments are generally characterized by lower permeabilities and better water quality than that in the shallower deposits. The minimum extent of the confining layers in the southwestern part of the Chino subbasin is indicated by the area mapped by Mendenhall (1905a, plate 1) as having artesian conditions in the early 1900s (fig. 11).

Groundwater Budget

Sources of recharge to the basin-fill aquifer in the Bunker Hill subbasin under modern conditions are primarily infiltration of streamflow along the mountain front and infiltration of excess water used for irrigation and public supply. Infiltration of precipitation on the basin floor, artificial recharge of imported water, and subsurface inflow from the mountain block and adjacent groundwater subbasins are relatively minor sources of recharge to the basin-fill aquifer. The largest component of groundwater discharge in the subbasin is now withdrawals from wells (91 percent), mainly for public supply. Danskin and others (2006, p. 55) compiled a groundwater budget for the Bunker Hill subbasin for 1945–98, which is listed in table 3. The estimated amounts of recharge from natural sources such as precipitation on the basin, infiltration of streamflow along the mountain front, and subsurface inflow from the mountain block and from adjacent subbasins, are assumed to be unchanged from predevelopment to modern conditions.

Under modern conditions, about 74 percent (136,500 acre-ft) of the estimated average annual recharge to the basin-fill aquifer in the Bunker Hill subbasin is from the infiltration of mountain runoff (Danskin and others, 2006, table 11). Wetter-than-normal periods contribute large quantities of recharge to the groundwater system and result in higher water levels (Hardt and Freckleton, 1987, p. 14) and increased amounts of groundwater in storage. Much of the runoff is diverted into stormwater-detention basins in or adjacent to stream channels along the mountain front that also operate as groundwater recharge facilities. Some of these basins have been in operation since the early 1900s. The Seven Oaks Dam on the Santa Ana River, completed in 1999, allows for additional recharge of streamflow in the Bunker Hill subbasin through the storage of excess runoff and subsequent release to allow infiltration in the stream channel and artificial recharge basins (Danskin and others, 2006, p. 19).

Artificial recharge of imported surface water from northern California began in 1972 with generally decreasing amounts imported through 1986 (Danskin and others, 2006, figure 19). Recharge of imported water at engineered facilities is estimated to average about 3,000 acre-ft/yr from 1945–98 and about 6,000 acre-ft/yr from 1972–98. Higher rates in 1973–82 of artificial recharge of imported water may have contributed to rising water levels and flooding in formerly artesian areas in San Bernardino (Danskin and others, 2006, p. 29).

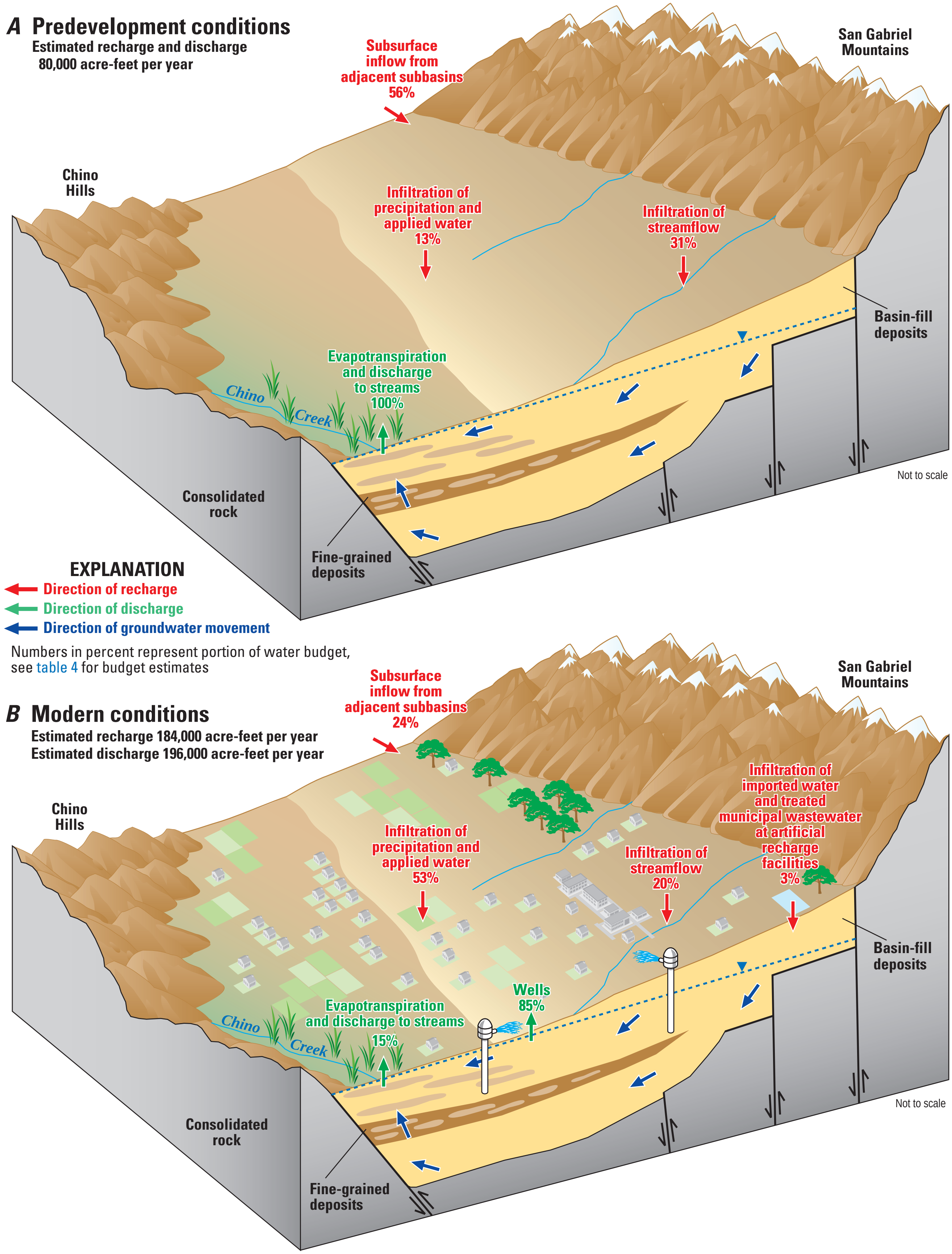
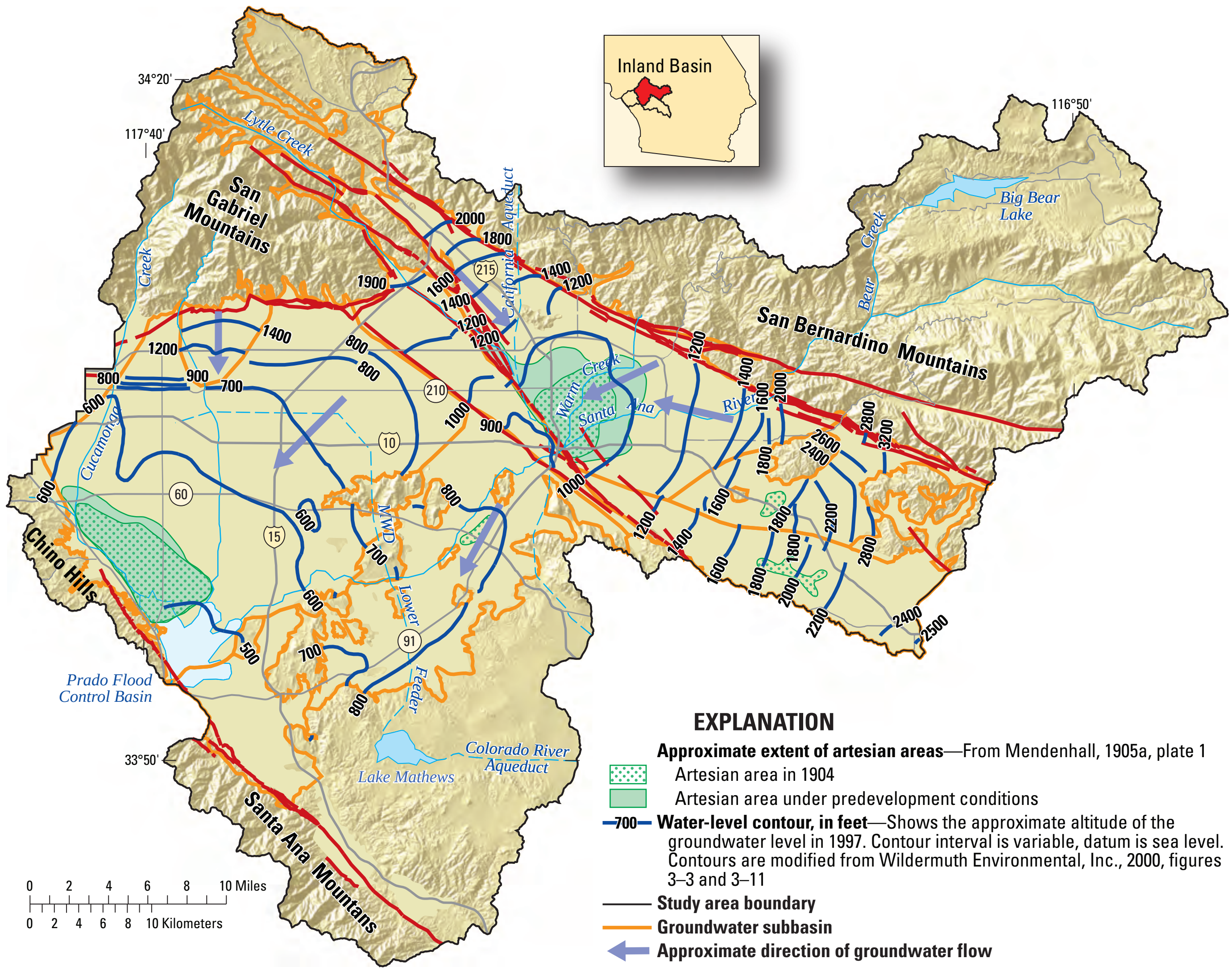


Figure 10. Generalized diagrams for the Chino subbasin, California, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.



Base compiled from U.S. Geological Survey digital data, 1:100,000 scale, 1977, 1978
National Elevation Data 1:24,000 scale, 2005
Albers Equal-Area Conic Projection, standard parallels 29°30', 45°30', central meridian 116°60'

Figure 11. Groundwater levels and generalized flow directions in 1997 in the Inland Basin, California.

Infiltration of excess water used for irrigation and public supply that recharges the aquifer system is mainly derived from pumped groundwater and is called return flow. Although water is extracted from different zones within the Bunker Hill basin-fill aquifer depending on well construction, return flow recharges only the shallow part of the aquifer through infiltration at the land surface. The downward movement of return flow is restricted by the presence of shallow confining layers in much of the area. For 1945–98, estimated recharge from infiltration of excess water used for irrigation and public supply averaged about 28,000 acre-ft/yr (Danskin and others, 2006, p. 42).

Seepage from consolidated rocks surrounding and underlying the basin-fill groundwater system commonly is assumed to be zero, but a heat-transport model suggested that as much as 15,000 acre-ft/yr of water could be contributed to the Bunker Hill subbasin from the consolidated-rock mountain block (Hughes, 1992). Danskin and others (2006, p. 53) noted that the inflow is greater than zero, though how much greater is unknown, and estimated about 6,000 acre-ft/yr of recharge by subsurface inflow from the mountain block.

Groundwater is discharged naturally by subsurface underflow out of the Bunker Hill subbasin, by upward flow into the lower reaches of Warm Creek, and by evapotranspiration. Underflow out of the subbasin near the Santa Ana River occurs only in the younger stream-channel deposits, which are about 100 ft thick (Danskin and others, 2006, p. 44). The river has eroded and redeposited these materials, removing most of the restriction to groundwater flow caused by the San Jacinto Fault (Dutcher and Garrett, 1963, p. 101). In the older, deeper deposits, fault gouge and the offset of permeable zones restrict groundwater flow.

Table 3. Estimated groundwater budget for the basin-fill aquifer system in the Bunker Hill subbasin, California, under predevelopment and modern conditions.

[All values are in acre-feet per year. Estimates of groundwater recharge and discharge are 1945–98 averages determined for the area and adjusted to compensate for the residual between recharge, discharge, and change in storage (Danskin and others, 2006, table 11) or are derived from these estimates. The budgets are intended only to provide a basis of comparison for overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components.]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Infiltration of precipitation on basin	[1] 6,500	6,500	0
Infiltration of streamflow near mountain front (includes engineered recharge under modern conditions)	[1] 136,500	136,500	0
Subsurface inflow from mountain block	[1] 6,000	6,000	0
Subsurface inflow from adjacent subbasins	[1] 5,000	5,000	0
Infiltration of excess irrigation water used for irrigation and public supply	0	28,000	28,000
Artificial recharge of imported water	0	3,000	3,000
Total recharge	**154,000**	**185,000**	**31,000**
Budget component		**Estimated discharge**	
Subsurface outflow across the San Jacinto Fault	[2] 51,400	6,600	-44,800
Evapotranspiration	[2] 51,300	6,000	-45,300
Discharge to streams	[2] 51,300	5,000	-46,300
Well withdrawals	0	175,000	175,000
Total discharge	**154,000**	**192,600**	**38,600**
Change in storage (total recharge minus total discharge)	0	-7,600	-7,600

[1] Assumed to be the same as estimated recharge under modern conditions.

[2] Assumed to be about one third of the estimated total recharge under predevelopment conditions.

Underflow near the Santa Ana River is mostly dependent on groundwater levels in the Bunker Hill subbasin. As groundwater levels in the subbasin rise, more water is forced out as underflow. As groundwater levels fall, less water leaves the subbasin as underflow. The water level in the lower (down gradient) part of the flow system controls the amount of storage in the subbasin, even though there is storage available in the upgradient part. Underflow out of the Bunker Hill subbasin was simulated at about 6,600 acre-ft/yr for 1945–98 (Danskin and others, 2006, table 11). Similarly, groundwater discharge to Warm Creek was shown to correspond to water levels in the aquifer with an average annual discharge from 1945–98 of about 5,000 acre-ft (Danskin and others, 2006, table 32).

Pumped groundwater in the Bunker Hill subbasin is used for agricultural, municipal, and industrial purposes. As the area has become urbanized, the quantity of water pumped for agricultural use has declined considerably. Withdrawals from wells for 1945–98 averaged about 175,000 acre-ft/yr, and ranged from about 123,000 acre-ft in 1945 to about 215,000 acre-ft in 1996 (Danskin and others, 2006, p. 47). As the San Bernardino area has become urbanized, wells have been installed higher on the alluvial fans, closer to the mountains and closer to the new urban demand.

Recharge to the groundwater system in the Chino subbasin is by infiltration of precipitation and water applied at the land surface, infiltration of streamflow (including stormwater runoff), subsurface inflow from adjacent subbasins, and infiltration of imported water and treated municipal wastewater at artificial recharge facilities such as spreading basins and storage ponds (table 4). Virtually all groundwater discharge under predevelopment conditions was by evapotranspiration and discharge to streams in areas where groundwater levels were at or near the land surface (fig. 10). Shallow bedrock in the gap between the Chino Hills and the Santa Ana Mountains forces groundwater to discharge to the river before it exits the subbasin. Groundwater discharge is now mostly from wells in the subbasin. Little information is available about groundwater conditions prior to development in the Chino subbasin, and estimates of recharge to and discharge from the aquifer presented in this section are intended only to provide a basis for comparison of change with development.

Table 4. Estimated groundwater budget for the basin-fill aquifer system in the Chino subbasin, California, under predevelopment and modern conditions.

[All values are in acre-feet per year. Estimates of groundwater recharge and discharge are from measurements and estimates for the July 1960 through June 2006 calibration period used by a groundwater flow model of the Chino subbasin (Wildermuth Environmental, Inc., 2007b, table 3-1) unless footnoted. Estimates for predevelopment conditions were not available. This budget is intended only to provide a basis of comparison for overall magnitudes of recharge and discharge between predevelopment and modern conditions, and does not represent a rigorous analysis of individual recharge and discharge components.]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Infiltration of streamflow	[1] 25,000	[2] 36,000	11,000
Subsurface inflow from adjacent subbasins	[3] 45,000	45,000	0
Infiltration of precipitation and applied water	[4] 10,000	[5] 98,000	88,000
Infiltration of imported water and treated municipal wastewater at artificial recharge facilities	0	5,000	5,000
Total recharge	**80,000**	**184,000**	**104,000**
Budget component		**Estimated discharge**	
Evapotranspiration	[6] 45,000	15,000	-30,000
Discharge to streams	[7] 35,000	14,000	-21,000
Well withdrawals	0	167,000	167,000
Total discharge	**80,000**	**196,000**	**116,000**
Change in storage (total recharge minus total discharge)	0	-12,000	-12,000

[1] Calculated as the residual of other components of recharge and discharge estimated for predevelopment conditions.

[2] Infiltration of streamflow includes the Santa Ana River and tributary streams. Recharge from the Santa Ana River has increased over time due to increased upstream streamflow while recharge from the tributaries has decreased due to channel lining (Wildermuth Environmental, Inc., 2007b, fig. 3-4).

[3]Assumed to be the same as the amount estimated for modern conditions.

[4] Estimated to be about 5 percent of average annual precipitation (16.4 inches per year) on the Chino subbasin (222 square miles).

[5] Could not separate out Temescal subbasin part of budget component.

[6] Estimated predevelopment evapotranspiration is based on an assumed evapotranspiration rate of 2 feet per year occurring over approximately 35 square miles, which includes the artesian area mapped by Mendenhall (1905a) and the area covered by the Prado Flood Control Basin within the Chino subbasin.

[7] Estimated to be the average groundwater outflow for 1930–40 calculated by French (1972, table 1). This period is affected by withdrawals from wells resulting in reduced groundwater discharge to streams.The estimate includes discharge by evapotranspiration to the lowland area along the Santa Ana River.

Recharge to the Chino subbasin from subsurface inflow from adjacent subbasins was simulated using a groundwater flow model of the subbasin calibrated to conditions from 1960 to 2006 and averaged about 45,000 acre-ft/yr (Wildermuth Environmental, Inc., 2007b, table 3-1). Much of the subsurface inflow moves across faults from the Cucamonga and Rialto-Colton subbasins. Infiltration of precipitation was estimated to be about 10,000 acre-ft/yr, or 5 percent of the average annual precipitation (16.4 in.) on the basin-fill area (222 mi^2). Recharge from excess irrigation water began in the 1800s and an average of about 98,000 acre-ft/yr of areal recharge, which includes infiltration of precipitation and excess water applied to the land surface, was specified in the groundwater model of the subbasin from 1960–2006 (Wildermuth Environmental, Inc., 2007b, table 3-1). This estimate includes a relatively small amount of recharge from the Temescal subbasin, which was not removed for this analysis.

Spreading and impounding local stormwater runoff in the northern part of the Chino subbasin began in the early 1900s. Flood control projects, mainly the lining of stream channels, have been constructed to capture and convey runoff in tributary streams to the Santa Ana River and out of the Chino subbasin. Urbanization has resulted in the creation of more impermeable surfaces in the basin that divert runoff to these lined channels. After 1987, minimal recharge to the groundwater system was modeled from tributary stream channels in the subbasin (Wildermuth

Environmental, Inc., 2007b, fig. 3-4). Infiltration from the Santa Ana River to the basin-fill aquifer has increased with time as a result of increased flow in the river associated with upstream urbanization. The infiltration of water from the channels of the Santa Ana River and its tributaries (including stormflow) specified in the groundwater flow model averaged about 36,000 acre-ft/yr from 1960–2006 (Wildermuth Environmental, Inc., 2007b, tables 3-1 and 3-2). Infiltration of streamflow under predevelopment conditions was estimated to be about 25,000 acre-ft/yr, the residual of estimated discharge and other estimated recharge components (table 4).

Urbanization has increased the volume of stormwater runoff that can be diverted to artificial recharge basins and has resulted in the artificial recharge of imported water and treated wastewater effluent in the subbasin. The infiltration of imported water and treated municipal wastewater at artifical recharge facilities began in 1978, and an average of about 5,000 acre-ft/yr for 1960–2006 was specified in the groundwater flow model (Wildermuth Environmental, Inc., 2007b, tables 3-1 and 3-3).

Evapotranspiration under predevelopment conditions was estimated to be about 45,000 acre-ft/yr (table 4) based on an approximate area of shallow groundwater within the Chino subbasin and an estimated evapotranspiration rate of 2 ft/yr. An area of about 23 mi^2 in which artesian conditions prevailed in the early 1900s was mapped in the western part of the subbasin (Mendenhall, 1905a, plate 1) indicating that the groundwater level was at or above the ground surface (fig. 11). Combined with other areas of shallow groundwater, including the area of the Prado Flood Control Basin, evapotranspiration is estimated to have occurred across approximately 35 mi^2 of the subbasin under predevelopment conditions. Evapotranspiration has decreased as groundwater levels have declined and was estimated to average about 15,000 acre-ft/yr from 1960–2006 (Wildermuth Environmental, Inc., 2007b, p. 3-1 and table 3-1).

Groundwater discharge to the Santa Ana River in the Chino subbasin was estimated by French (1972, table 3) to average about 35,000 acre-ft/yr from 1933 to 1939. French developed his water budget values from data for measured streamflow at the upstream and downstream ends of the subbasin, direct runoff to the river, inflows and outflows, evapotranspiration along the river, and inflow from the Temescal subbasin. Groundwater discharge to streams in the subbasin under predevelopment conditions was estimated at about 35,000 acre-ft/yr (table 4). This estimate is at best a very “rough” one, but is assumed to be reasonable if evapotranspiration near the river and inflow from the Temescal subbasin are accounted for. More discharge to the river resulting from higher groundwater levels and groundwater discharge to Chino Creek and other creeks draining the artesian area prior to development also must be accounted for. The difference in discharge in the Santa Ana River at upstream and downstream ends of the subbasin during the relatively dry summer months of July, August, and September were compiled by Post (1928, p. 357) for 6 years during 1891–1905. The average summer base flow determined from this data extended through a year equals about 16,000 acre-ft of streamflow that is assumed to be primarily groundwater discharge to the river under early development conditions. Groundwater discharge to streams averaged about 14,000 acre-ft/yr from 1960–2006 in the groundwater flow model of the Chino subbasin (Wildermuth Environmental, Inc., 2007b, p. 3-3).

Withdrawals from wells in the Chino subbasin averaged about 167,000 acre-ft/yr from 1960–2006 (Wildermuth Environmental, Inc., 2007b, table 3-4) (table 4). Land and water use in the Chino subbasin has progressively shifted from agricultural to urban. The area of urban land increased from 7 percent in 1933 (Wildermuth Environmental, Inc., 1999, table 2-1) to 75 percent of the subbasin area in 2001 (McKinney and Anning, 2009), mainly at the expense of irrigated and non-irrigated agricultural land. Groundwater withdrawals for agriculture, primarily in the southern part of the subbasin, decreased from about 54 percent of the total production in 1977–78 to about 18 percent in 2005–06 (Wildermuth Environmental, Inc., 2007a, p. 3-4). During the same period, withdrawals for municipal and industrial uses, mainly in the northern half of the basin, increased from about 40 percent of total production in 1977–78 to 80 percent in 2005–06. In 2005–06, about 119,000 acre-ft/yr was pumped for municipal and industrial use and about 29,000 acre-ft/yr was pumped for agricultural use (Wildermuth Environmental, Inc., 2007b, table 3-4).

The groundwater budget for modern conditions (1960–2006) in the Chino subbasin has more than doubled from the estimated budget for predevelopment conditions (table 4), mainly as a result of the infiltration of excess applied water. Under modern conditions, more water is discharged than is recharged to the aquifer, resulting in the removal of water from storage. This rough estimate of overdraft, however, should not be used to calculate an accumulated volume of water removed from storage.

Groundwater Flow

Groundwater in the Bunker Hill subbasin generally moves from dispersed areas of recharge along the base of the San Bernardino Mountains towards a more focused area of discharge where the Santa Ana River crosses the San Jacinto Fault Zone as it did under predevelopment conditions (Danskin and others, 2006, p. 56) (fig. 11). The convergence of flow paths is caused by the San Jacinto Fault acting as a partial barrier to groundwater flow. The mountain-front streams generally lose most or all of their water to the groundwater system as they enter the basin and flow across the alluvial deposits. Under predevelopment conditions, flow resumed further downstream as a result of groundwater, restricted from flowing across the less permeable San Jacinto Fault at depth, rising to the land surface and discharging to Warm Creek or as subsurface outflow near the Santa Ana River (Danskin and others, 2006, p. 15).

Historically, a downward vertical gradient was present in the recharge areas of the Bunker Hill subbasin, and an upward vertical gradient was present in the discharge area. Under predevelopment conditions, a large artesian area covered nearly one-third of the subbasin and extended east towards the San Bernardino Mountains (Mendenhall, 1905b) (fig. 11). Natural groundwater discharge occurred at extensive, but somewhat discontinuous bogs, swamps, and marshlands generally near the lower stream reaches.

The vertical pattern of groundwater flow in the Bunker Hill subbasin has changed significantly from predevelopment to modern conditions because of withdrawals from wells. As groundwater production increased, water was withdrawn from increasingly deeper parts of the basin-fill aquifer. Natural discharge to the land surface was replaced by discharge to pumping wells. Hydraulic head within the aquifer changed to reflect the change in groundwater flow patterns, and the upward vertical gradient was reduced or reversed. The size of the artesian area has fluctuated historically depending on the amount of water recharged and discharged from the groundwater system (Hardt and Freckleton, 1987, fig. 3). By 1992, the area of historically flowing wells east of the San Jacinto Fault had a downward vertical gradient.

Near the base of the San Bernardino Mountains, large rises or declines in water levels occur in response to changes in recharge from streams. During times of drought and increased pumping, water levels decline as much as 200 ft, which limits the ability of water purveyors to supply sufficient groundwater to meet demand (Danskin and others, 2006, p. 8). Water levels also fluctuate in the area where the Santa Ana River crosses the San Jacinto Fault. During a period of extensive groundwater extractions from 1950 to 1970, water levels fell by as much as 100 ft and induced land subsidence of as much as 1 ft (Miller and Singer, 1971). After 1970, both natural and artificial recharge increased, so that by 1980, water levels had risen to within a few feet of the land surface (Hardt and Freckleton, 1987).

Groundwater flow in the Chino subbasin generally follows surface drainage patterns and moves from the higher altitude alluvial fans flanking the San Gabriel Mountains towards lower altitude discharge areas near the Santa Ana River and Prado Flood Control Basin (fig. 11). Depth to water ranges from more than 500 ft below land surface in the northern part of the subbasin to near land surface in the southern part, with steeper gradients in the northern part. Water levels in the subbasin have declined since development began, with larger declines in the northern unconfined part. Withdrawals from wells has reversed or changed groundwater flow directions in some areas of the Chino subbasin. Water levels measured in 1997 are as much as 200 ft below land surface in the formerly artesian area (Wildermuth Environmental, Inc., 1999, fig. 2-23). Land subsidence in this area is attributed to compaction of the fine-grained material caused by local groundwater withdrawals. Changes in groundwater levels in the Chino subbasin in response to periods of above-normal-precipitation are small, on the order of a few feet, because of the relatively small mountain drainage area that is tributary to the subbasin. In addition, the relatively large size of the subbasin coupled with thick unsaturated zones in the unconfined recharge areas delay the effects of recharge on groundwater levels.

Effects of Natural and Human Factors on Groundwater Quality

Agricultural and urban development have caused groundwater quality changes in the Inland Basin, primarily increased concentrations of dissolved solids and nitrate and the presence of VOCs and perchlorate. The basin-fill aquifers are susceptible to water-quality changes because of the unconfined conditions in much of the area, and are vulnerable to contamination because of the overlying land uses that utilize chemicals and water.

General Water-Quality Characteristics and Natural Factors

Extensive analyses of water sampled from active production and monitoring wells have been made to determine groundwater quality in the Inland Basin, especially in the Chino subbasin (Wildermuth Environmental, Inc., 2005, fig. 4-1). Concentrations of dissolved solids in groundwater from much of the Inland Basin generally are less than about 500 mg/L, except in downgradient discharge areas near the Santa Ana River and south of the river in much of the Temescal and Riverside-Arlington subbasins (fig. 2 and Wildermuth Environmental, Inc., 2008, appendix B). Shallow groundwater (between land surface and the first major confining layer) in the area of the Chino subbasin near the Prado Flood Control Basin had higher concentrations of dissolved solids that were statistically computed from multiple values per well (about 1,000–1,800 mg/L) than deeper confined groundwater (about 300–500 mg/L) (Wildermuth Environmental, Inc., 2008, appendix B). Wildermuth Environmental, Inc. (2005, p. 4-8) states that most areas in the Chino subbasin with either significant irrigated land use or dairy waste disposal histories overlie groundwater with elevated concentrations of dissolved solids (see 'Potential Effects of Human Factors' section). Concentrations of dissolved solids in water sampled from 204 public-supply wells in the Bunker Hill subbasin range from 155 to 1,140 mg/L, with a mean value of 324 mg/L (California Department of Water Resources, 2003d). Shallow groundwater in the confined part of the subbasin near the San Jacinto Fault had higher statistically derived concentrations (about 350-600 mg/L) than deeper confined groundwater (about 250-400 mg/L) (Wildermuth Environmental, Inc., 2008, appendix B).

Most of the 50 water-supply wells in the Inland Basin sampled as part of NAWQA studies (fig. 12) produced calcium-bicarbonate type water (Hamlin and others, 2002), which primarily reflects the quality of recharge water originating in high-altitude areas of the adjacent San Gabriel and San Bernardino Mountains. Dissolved-solids concentrations in water sampled from these wells in areas of natural recharge or near recharge facilities primarily ranged from about 180 to 250 mg/L.

The trace elements arsenic and uranium are present in the groundwater sampled, but typically are not contaminants of concern in the Inland Basin. Concentrations of dissolved arsenic ranged from less than 1.0 to 10 µg/L with a median of 1.1 µg/L in samples collected by NAWQA from 29 production wells, used for both public supply and irrigation, in the Inland Basin (Hamlin and others, 2002, appendix 6). Along the flow paths sampled by NAWQA in the Bunker Hill subbasin, arsenic concentrations were less than 0.9 µg/L in the unconfined proximal parts compared to generally greater concentrations (up to 12.1 µg/L) in the confined distal part (Hamlin and others, 2002, appendix 6). The longer contact time between groundwater and aquifer material at the end of the flow path is a likely cause of the greater concentrations of arsenic. Only one of the samples collected by NAWQA in the Inland Basin had a concentration of uranium greater than the drinking-water standard of 30 µg/L (40.5 µg/L in water from an irrigation well) (Hamlin and others, 2002, p. 34 and appendix 8).

Water sampled from 26 wells along flow paths in the Bunker Hill subbasin by the NAWQA Program was classified as being withdrawn from unconfined or confined parts of the aquifer (table 1). The unconfined part of the aquifer corresponds to the area in which recharge occurs in the subbasin, whereas the confined part is at the confluence of many groundwater flow paths in the discharge area. The median concentration of dissolved solids was lower and the median concentration of nitrate was higher for water sampled from wells in the upgradient unconfined part of the aquifer than for water from the downgradient confined part. Lower concentrations of dissolved oxygen indicate likely nitrate-reducing conditions in the confined part of the aquifer, whereas higher concentrations of dissolved oxygen would limit denitrification in the unconfined part of the aquifer. The median concentration of the stable isotope deuterium also was lighter in water sampled from the unconfined part of the aquifer compared to the confined part (table 1) indicating that there is some recharge at lower altitudes in the subbasin.

Potential Effects of Human Factors

Elevated concentrations of dissolved solids and nitrate in groundwater have resulted from past and present agricultural practices and urban development in parts of the Inland Basin. Dissolved solids in Chino subbasin groundwater has increased primarily due to evaporative concentration after irrigation, from the leaching of fertilizer and manure in agricultural areas, and from the recharge of treated wastewater and imported water (Wildermuth Environmental, Inc., 2005, p. 4-8). In general, sources of nitrate to groundwater include leaching of fertilizers and animal wastes applied to the land, leakage from sewer pipes, and reuse of treated wastewater. Another source of nitrate in groundwater in the Chino subbasin is infiltration of wastewater from animal feeding facilities. Runoff from these facilities can have high concentrations of ammonia, which can in turn result in high concentrations of nitrate. Computed statistics representing nitrate (as nitrogen) concentrations in shallow groundwater in the area of the Chino subbasin between the Prado Flood Control Basin and Highway 60 ranged from about 20 mg/L to greater than 100 mg/L (Wildermuth Environmental, Inc., 2008, appendix B). Computed statistics representing nitrate (as nitrogen) concentrations for the deeper confined groundwater in this area are much lower than in the overlying shallow groundwater and range from 5 to 18 mg/L.

Water-quality monitoring during 1999–2004 for the Chino subbasin groundwater management plan showed concentrations of dissolved solids and nitrate (as nitrogen) mostly greater than the secondary drinking-water standard of 500 mg/L and the primary drinking-water standard of 10 mg/L, respectively, in water sampled from the upper part of the aquifer system in the southern part of the subbasin (Wildermuth Environmental, Inc., 2005, figs. 4-4 and 4-7). Concentrations measured from wells in the northern part of the Chino subbasin (north of Highway 60) during this period were typically less than 300 mg/L for dissolved solids and generally varied from less than 2 to greater than 10 mg/L for nitrate (as nitrogen).

Concentrations of nitrate (as nitrogen) were greater than 10 mg/L in water from 14 percent of the 29 production wells sampled by NAWQA in the Inland Basin. In samples from all 4 wells with nitrate concentrations greater than 10 mg/L, tritium was greater than 1 pCi/L, and VOCs and (or) pesticides were detected; dissolved-solids concentrations greater than 500 mg/L occurred in samples from 3 of these 4 wells.

Most of the agricultural land use in the southern part of the Chino subbasin (fig. 12) is projected to be converted to urban uses over the next 20 to 30 years. Groundwater pumped from this area will have to be treated before it can be used for public supply because of elevated concentrations of dissolved solids and nitrate. More of this poorer quality groundwater is projected to move toward and discharge to the Santa Ana River if withdrawals from wells in the area decrease. About 500 acres of constructed wetlands in the Prado Flood Control Basin were designed primarily to lower nitrate concentrations in the Santa Ana River below the Prado Dam.

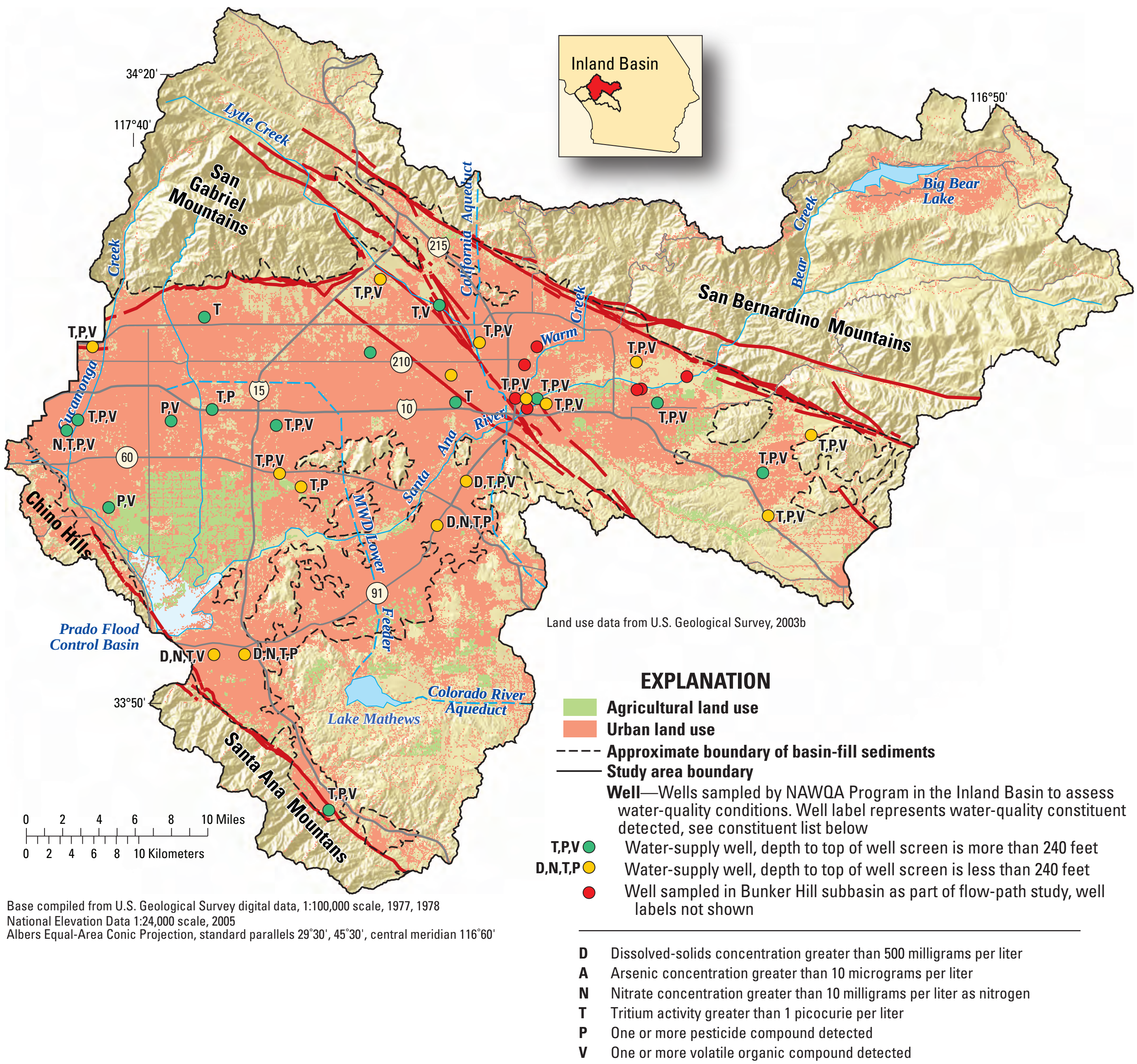


Figure 12. Locations of and chemical characteristics of water in wells sampled in the Inland Basin, California, by the NAWQA Program, 2000.

This surface water is used to recharge the basin-fill aquifer in the Coastal Basin. A plan to manage groundwater quality in the Chino subbasin has been developed that attempts to balance recharge to the subbasin with discharge from wells (Wildermuth Environmental, Inc., 2007b, p. 7-2). Wells and treatment plants (desalters) are being used to reduce the amount of groundwater discharge that contains elevated concentrations of dissolved solids and nitrate from the Chino subbasin to the Santa Ana River through the manipulation of hydraulic gradients. Desalters also are in operation in the Riverside-Arlington and Temescal subbasins (Santa Ana Watershed Authority, no date). As the demand for water increases with time, the artificial recharge of treated municipal wastewater (recycled water) in the subbasin is projected to increase from about 12,500 acre-ft in 2005 to about 58,000 acre-ft in 2010, corresponding to an equal reduction in the demand for imported water from northern California (Wildermuth Environmental, Inc., 2007b, p. 7-2). The change in concentrations of dissolved solids and nitrate in the recharge water must be accounted for as part of managing the groundwater system.

Numerous contaminant plumes (mainly VOCs and perchlorate) in the Inland Basin that are related to industrial activities extend miles from the source of contamination (Hamlin and others, 2005, fig. 3; Wildermuth Environmental, Inc., 2005, figs. 4-18 and 4-26; and California Department of Water Resources, 2003d). Since about 1980, these contaminants have become widespread in the groundwater, and have affected the operation of many public-supply wells in the basin (Danskin and others, 2006, p. 59; Santa Ana Watershed Project Authority, 2002, p. 3-10). Perchlorate contamination in Chino subbasin groundwater has been attributed to known point source releases and possibly from fertilizers historically used on citrus orchards in the northern part of the subbasin (Wildermuth Environmental, Inc., 2007a, p. 4-10).

Pesticides and VOCs were frequently detected, although mostly at very low concentrations (Hamlin and others, 2002, appendixes 9D, 9E, 10D, 10E, 11E, 11F, 12D, and 12E), in water sampled by NAWQA from 50 wells in the Inland Basin (table 1 and fig. 12). The large number of detections of these compounds in this small subset of existing wells probably reflects generally unconfined conditions in the groundwater system, present and past land uses, and the relatively low organic content of aquifer materials (Hamlin and others, 2005). The most frequently detected pesticides were atrazine; one of its degradation products, deethylatrazine; and simazine.

The most frequently detected VOCs in the samples collected by NAWQA were chloroform, trichloroethene (TCE), and perchloroethene (PCE). Some wells with VOC detections were near known VOC plumes emanating from industrial sites (Hamlin and others, 2005) and concentrations above drinking-water standards were detected in water sampled from 4 irrigation wells and 2 flow-path study monitoring wells (Hamlin and others, 2002, appendixes 9D, 9E, 11E, and 11F). Wells without VOC detections were generally deep or near recharge areas upgradient from urban areas along the mountain front. About 94 percent of the wells sampled by NAWQA at the lower end of the flow paths in the confined part of the Bunker Hill subbasin had a VOC detected compared to about 56 percent of the wells nearer to the mountain front (table 1). This suggests that either (1) high concentrations of these VOCs reached the groundwater at some time in the past in the unconfined, upgradient area of the flow paths and have moved downgradient or (2) VOCs are introduced all along the flow paths, even in the confined part of the aquifer. Methyl *tert*-butyl ether (MTBE) is an oxygenate added to gasoline to improve combustion and motor vehicle emissions. Its use in California was banned in 1999 because of groundwater contamination. The absence of MTBE in the unconfined part of the flow paths and its presence in the confined part suggest downward groundwater flow and that the confining units present in the distal part of the Bunker Hill subbasin do not prevent VOCs from reaching the aquifer (Dawson and others, 2003, p. 71).

The median depth to the top of the well-screen interval for the 29 production wells sampled by NAWQA in the Inland Basin was 240 ft below land surface. Samples from wells in which the top of the well-screen interval ranged in depth from 26 to 240 ft below land surface had a higher median concentration for dissolved solids and tritium and had more pesticide and VOC detections than did samples from wells in which the top of the well-screen interval ranged in depth from 250 to 650 ft below land surface (table 1 and fig. 13). Most of the wells in this sample set tap unconfined aquifers and therefore may be susceptible to receiving compounds generated by overlying land-use activities (Hamlin and others, 2002, p. 25). Wells with deeper screened intervals likely encounter more layers of fine-grained material that can impede the downward movement of water recharged at the land surface.

Since 1980, when extensive groundwater contamination by VOCs was discovered in the Bunker Hill subbasin, many new wells have been installed with perforations only below a depth of 200 to 300 ft below land surface. This change in construction, largely to avoid water-quality problems near the land surface, has further altered the vertical movement of groundwater in the subbasin. About the same amount of water is now pumped from the shallower and deeper parts of the aquifer in the Bunker Hill subbasin (Danskin and others, 2006, p. 49). The hydraulic head in the deeper part of the aquifer will decline with additional withdrawals. This may induce some groundwater flow to the deeper pumped zones from underlying older, more consolidated basin-fill deposits, through faults and fractures, and possibly from the surrounding and underlying bedrock.

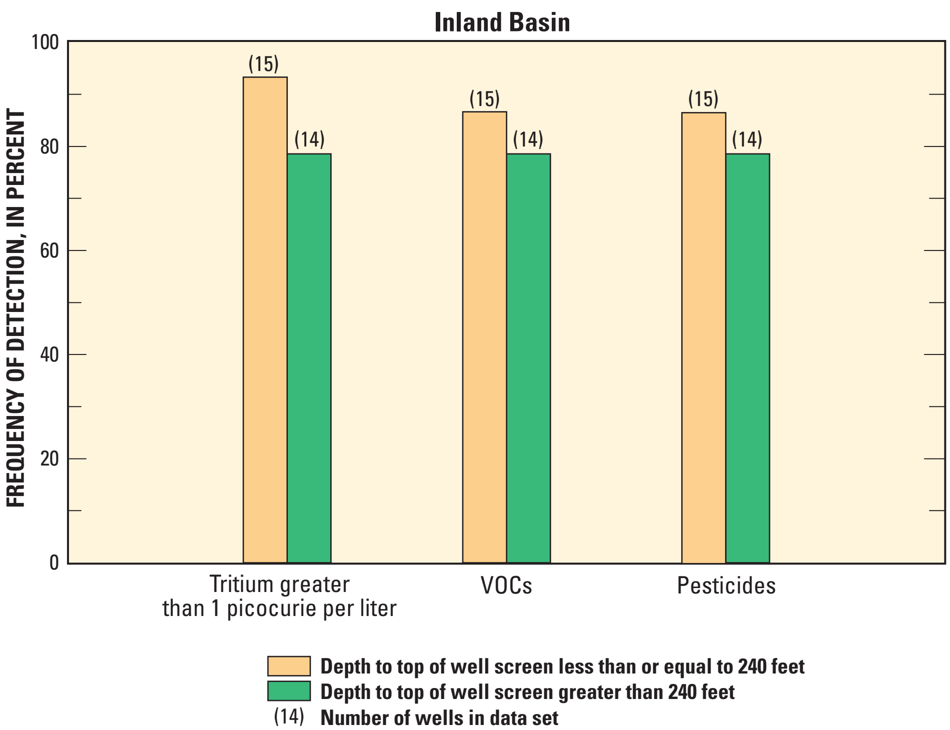


Figure 13. Detection frequencies of tritium, volatile organic compounds, and pesticides in water samples from wells in the Inland Basin, California.

Coastal Basin of the Santa Ana Basin

Artificial recharge to the groundwater system and pumping from wells has accelerated the movement of water through the Coastal Basin basin-fill aquifer. To a large extent, native water in the aquifer has been replaced by water recharged since the early 1950s (Belitz and others, 2004, p. 8). Groundwater quality in the basin is affected by the enhanced recharge of water from the Santa Ana River, including the infiltration of treated wastewater that now is a large component of base flow in the river, and the infiltration of imported water.

The approximately 800-mi^2 Los Angeles physiographic basin is subdivided into two groundwater basins on the basis of sources of recharge water—the Coastal Los Angeles Basin in the north and the Coastal Santa Ana Basin in the south. The hydrogeologic settings of the two groundwater basin are similar, in that each contains flow paths originating at focused engineered recharge facilities where major streams enter the basin and the water then moves radially outward toward dispersed areas of pumping in the confined part of the basin. The Coastal Santa Ana Basin is described in this section and is referred to as the Coastal Basin (fig. 14).

Analysis of modeled precipitation data for 1971–2000 (PRISM Group, Oregon State University, 2004) resulted in an average annual precipitation value of about 13.2 in. over the 340 mi^2 Coastal Basin (McKinney and Anning, 2009, table 1). Analysis of LandScan population data for 2005 (Oak Ridge National Laboratory, 2005) indicated a population of almost 2.6 million in the Coastal Basin and a population density of about 7,000 people/mi^2 (McKinney and Anning, 2009, table 1). Groundwater from the Coastal Basin supplies about 70 percent of the total water demand. The remaining 30 percent is obtained from water imported through the Colorado River Aqueduct and from northern California (Orange County Water District, 2008). Overall, the Coastal Basin has the highest percentage of urbanized land (94 percent) in 2001 (U.S. Geological Survey, 2003b) of the three NAWQA studied groundwater basins in the Santa Ana Basin. Groundwater use also is highest in the Coastal Basin, and in contrast to the Inland and San Jacinto Basins, most of the aquifer is confined and insulated from the effects of overlying land uses.

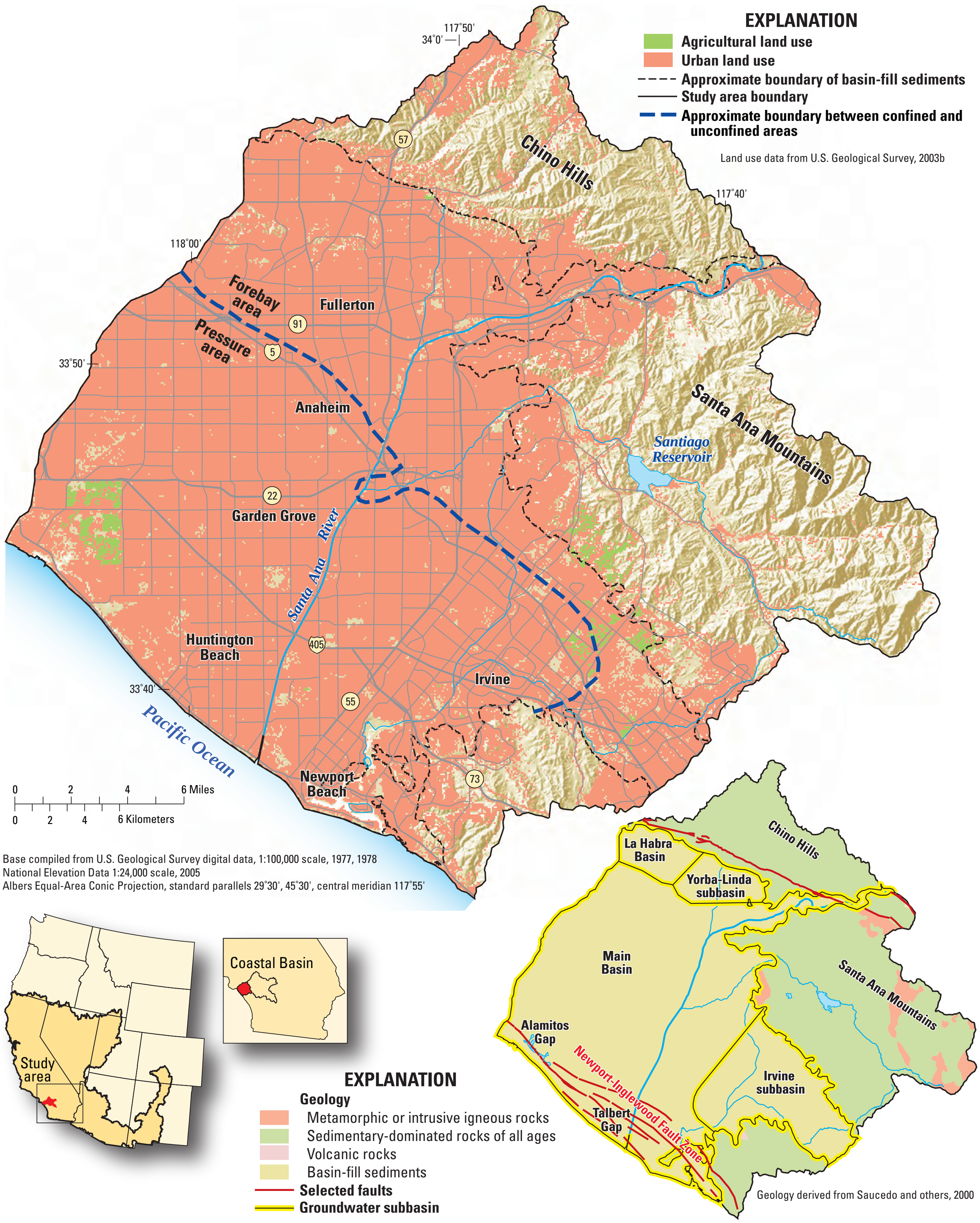


Figure 14. Physiography, land use, and generalized geology of the Coastal Santa Ana Basin, California.

Conceptual Understanding of the Groundwater System in the Coastal Basin

The Coastal Basin groundwater system is analogous to a bowl filled with sediment and water, the central part of which contains freshwater-bearing deposits up to 4,000 ft thick (California Department of Water Resources, 1967). Deposits along the margins of the basin, in the Irvine and Yorba Linda subbasins and the La Habra Basin, are less thick and less permeable than in the main part (Main Basin). The Coastal Basin has been divided into a forebay area and a pressure area on the basis of the relative abundance of shallow clay layers in the subsurface. The forebay area is small (38 percent of the basin) compared to the pressure area (62 percent) and occupies about 130 mi^2 along the west side of the Santa Ana Mountains, generally north and east of the Interstate-5 freeway (fig. 14). The unconsolidated sediments in the forebay area deepen to about 1,000 ft thick away from the basin margins and consist mainly of interbedded sands and gravels with occasional lenses of silt and clay derived from the mountains to the east and southeast and from marine deposits. Hydraulic conductivity values used in a groundwater flow model of part of the forebay area ranged from 150–300 ft/d (Tompson and others, 1999, p. 2985) and a value of about 600 ft/d was estimated for coarse-grained deposits from tracer studies (Davisson and others, 2004, p. 93). The fine-grained sediments in the forebay area are laterally discontinuous and generally do not restrict the vertical movement of groundwater, resulting in unconfined to semiconfined conditions and a downward hydraulic gradient. A hydraulic conductivity value of 1 ft/d was used to simulate fine-grained material in the forebay area (Tompson and others, 1999, table 4). Groundwater recharge in the Coastal Basin primarily occurs in the forebay area.

The pressure area extends from the western edge of the forebay area to the Pacific Ocean. The pressure area contains relatively continuous, thick layers of silt and clay that confine underlying sands and gravels and typically impede the vertical movement of groundwater. The Newport-Inglewood Fault Zone parallels the coastline and generally forms a barrier to groundwater flow. This impedance to flow causes hydrostatic pressure within the aquifer and upward vertical gradients in the western part of the basin. Seawater intrusion can occur where the less permeable uplifted rocks are breached by gaps that are filled with alluvium coupled with lower water levels upgradient of the barrier, such as at the Alamitos and Talbert Gaps (California Department of Water Resources, 2003e).

A simplified conceptual model for the groundwater system in the Coastal Basin consists of an upper (shallow) aquifer system, a middle (principal) aquifer system, and a lower (deep) aquifer system (California Department of Water Resources, 2003e; Orange County Water District, 2004, fig. 2-2) (fig. 15). Water recharged in the generally unconfined forebay area moves to each of these aquifer systems, although vertical flow is impeded by discontinuous layers of silt and clay and horizontal flow is affected by faults and bedrock structure in the subsurface. The shallow aquifer system, generally the uppermost 200 ft of basin-fill deposits, provides about 5 percent of the total groundwater production in the basin, mainly for irrigation use (Orange County Water District, 2004, p. 2-2).

Most of the groundwater withdrawals in the Coastal Basin are from wells completed in the principal aquifer system, with the main production zones generally between 300 and 1,500 ft below land surface. The main production zones in the confined part of the principal aquifer system can be overlain by 300 to 500 ft of sediment containing large amounts of silt and clay, which typically impede vertical groundwater flow (Herndon and others, 1997). As of 2004, few wells have been drilled and completed in the deep aquifer system (Orange County Water District, 2004, p. 2-2) due to depth and aesthetic issues with the water, such as color and odor. The deep aquifer system is tapped by wells in the southwest part of the basin at depths of about 600–1,200 ft below land surface. These wells reduce the upward pressure and migration of colored water into the principal aquifer system in the area (Mesa Consolidated Water District, 2005).

Water Budget

Recharge to the Coastal Basin groundwater system under predevelopment conditions was primarily from the infiltration of water through the channels of the Santa Ana River and smaller streams flowing into the forebay area. Groundwater recharge under predevelopment conditions is estimated to be about 163,000 acre-ft/yr (table 5). Gross estimates of recharge prior to water development in the Coastal Basin from the infiltration of Santa Ana River water, precipitation on the basin, and inflow along the mountain fronts are derived from estimates for modern conditions. Little information is available on groundwater conditions prior to development in the basin and these estimates of recharge to and discharge from the aquifer are intended only to provide a basis for comparison of change with development.

Under modern-day conditions, flow in the Santa Ana River to the Coastal Basin consists predominantly of perennial base flow that is mostly treated wastewater (Mendez and Belitz, 2002) and intermittent stormflow that includes runoff from urban and agricultural land. According to the Orange County Water District (2004, p. 5-5), the Santa Ana River loses about 100 ft^3/sec of flow (72,400 acre-ft/yr) to the groundwater system along a 6-mile segment near where it enters the groundwater basin. Downstream from this reach, a low permeability clay layer in the subsurface impedes infiltration of water from the river to the aquifer (Orange County Water District, 2004, p. 5-3).

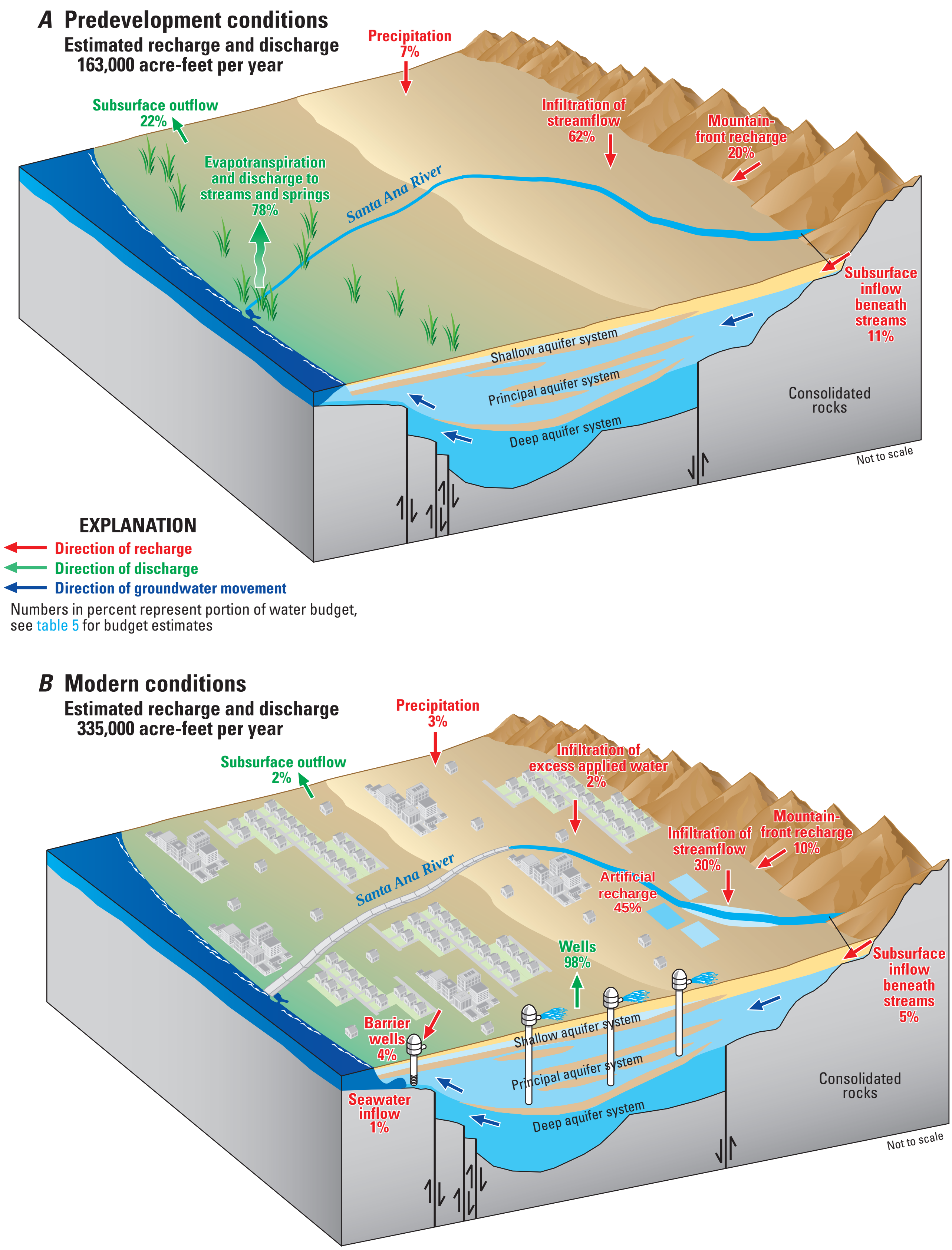


Figure 15. Generalized diagrams for the Coastal Basin, California, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

Table 5. Estimated groundwater budget for the basin-fill aquifer system in the Coastal Basin, California, under predevelopment and modern conditions.

[All values are in acre-feet per year. Estimates of groundwater recharge and discharge under modern conditions are from the Orange County Water District (2004, table 2-1). Estimates for predevelopment conditions are derived from those for modern conditions or were estimated as described in the footnotes and text. The budgets are intended only to provide a basis of comparison for overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components.]

	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Mountain-front recharge	33,000	33,000	0
Infiltration of precipitation on basin	12,000	12,000	0
Infiltration of streamflow	[1] 100,000	100,000	0
Subsurface inflow beneath major streams	18,000	18,000	0
Infiltration of excess applied water	0	5,500	5,500
Artificial recharge in forebay area	0	150,000	150,000
Seawater inflow through coastal gaps	0	2,000	2,000
Seawater intrusion barrier wells	0	14,500	14,500
Total recharge	**163,000**	**335,000**	**172,000**
Budget component		**Estimated discharge**	
Evapotranspiration and discharge to streams and springs	[2] 127,000	0	-127,000
Subsurface outflow across county line	36,000	8,000	-28,000
Well withdrawals	0	327,000	327,000
Total discharge	**163,000**	**335,000**	**172,000**
Change in storage (total recharge minus total discharge)	0	0	0

[1] Estimated natural base flow and stormflow in the Santa Ana River.

[2] Calculated to be the difference between predevelopment recharge and discharge from subsurface outflow out of the Coastal Basin at the Los Angeles/Orange County line.

Estimated base flow in the Santa Ana River below Prado Dam has increased from about 40,000 acre-ft in 1970 to about 155,000 acre-ft in 2001, owing primarily to increases in treated wastewater discharge in upstream basins (Orange County Water District, 2004, fig. 5-3). Summer base flow (monthly flow in July, August, and September) from 1878 to 1928 in the Santa Ana River below the location of Prado Dam averaged about 10,000 acre-ft (Post, 1928, p. 356). Assuming that this summer flow is not affected by storm runoff and is consistent throughout the year, base flow prior to major urban development in the upper part of the watershed is estimated at about 40,000 acre-ft/yr. This amount of base flow in the Santa Ana River is assumed to approximate predevelopment conditions and to have recharged the aquifer in the Coastal Basin.

Annual recharge to the aquifer from captured Santa Ana River stormflow is estimated to average about 60,000 acre-ft (Orange County Water District, 2004, p. 2-7). Although the amount of stormflow in the river entering the Coastal Basin prior to the construction of Prado Dam may have been greater, average recharge from the Santa Ana River under predevelopment conditions is assumed to be the sum of the estimates for base flow (40,000 acre-ft/yr) and stormflow (60,000 acre-ft/yr), about 100,000 acre-ft/yr (table 5). Climatic conditions greatly influence annual recharge to the Coastal Basin, and wet periods result in more recharge to the groundwater system due to expansion of the river in its floodplain and more runoff from precipitation on and flowing to the forebay area.

Stable-isotope data for older groundwater sampled from the pressure area of the Coastal Basin indicate that in the predevelopment state, recharge from near-coastal precipitation was minor compared to recharge from the Santa Ana River (Williams, 1997, p. 241). Areal recharge from precipitation on the basin floor is estimated to average about 12,000 acre-ft/yr. Infiltration of direct precipitation to the aquifer system was estimated to be about 10 percent of annual average precipitation (0.11 ft/yr) on the forebay area (about 9,000 acre-ft/yr) and 2 percent on the pressure area (about 3,000 acre-ft/yr), based on the occurrence of fine-grained sediment layers near land surface. Tompson and others (1999, p. 2985) estimated areal recharge from precipitation in the forebay area at 0.1 ft/yr. Recharge from infiltration of excess applied irrigation water in the basin is estimated at about 5,500 acre-ft/yr, the difference between recharge from rainfall and irrigation estimated by the Orange County Water District (2004, table 2-1) and the estimate for recharge from precipitation on the basin.

The Santa Ana River and Santiago Creek are estimated to lose about 8,000 and 10,000 acre-ft/yr, respectively, as subsurface inflow to the basin and as infiltration through their channels near the mountain front (Orange County Water District, 2004, table 2-1). Recharge from ephemeral streams and runoff from consolidated rocks in the hills and mountains bounding the Coastal Basin (mountain-front recharge) is estimated to be about 33,000 acre-ft/yr (Orange County Water District, 2004, table 2-1). These estimates of natural recharge are assumed to represent both predevelopment and modern conditions (table 5).

In a water budget constructed by the Orange County Water District (2004, table 2-1), estimated recharge and discharge to the modern Coastal Basin aquifer system is about 335,000 acre-ft/yr. The groundwater basin is managed to maintain an overall balance over many years that incorporates periods of above- and below-average precipitation. The balanced budget is based on the following assumptions: (1) average precipitation, (2) recharge at the Santa Ana River and at recharge facilities in the forebay area (both natural and engineered) is held to the current maximum capacity of 250,000 acre-ft/yr, and (3) withdrawals from wells are adjusted so that total groundwater inflows and outflows are equal (Orange County Water District, 2004, p. 2-6).

Currently, water managers utilize almost all of the base flow and much of the stormflow in the Santa Ana River that enters the Coastal Basin to recharge the aquifer system. About 100,000 acre-ft/yr of water is estimated to infiltrate from the Santa Ana River to the aquifer naturally (the same amount of recharge from the river as estimated for predevelopment conditions). Additional streamflow, mostly from increased base flow and stormflow, is introduced artificially at engineered recharge facilities in and along the Santa Ana River channel and at a smaller facility on Santiago Creek. Imported Colorado River and northern California water also have been artificially recharged in the forebay (Herndon and others, 1997). Artificial recharge in the forebay area is estimated at about 150,000 acre-ft/yr (table 5), or about 45 percent of the total recharge to the basin.

Under modern-day water-level conditions, seawater has intruded into the aquifer. The Orange County Water District (2004, table 2-1) estimates about 2,000 acre-ft/yr of seawater flows into the basin-fill aquifer through coastal gaps. To limit this seawater intrusion, about 14,500 acre-ft/yr of freshwater is injected into the aquifer in the Talbert and Alamitos Gaps (Orange County Water District, 2004, table 2-1).

The Orange County Water District and the Orange County Sanitation District have developed an additional source of water that began recharging the aquifer in the Coastal Basin in 2008. About 42,000 acre-ft/yr of treated wastewater is processed using microfiltration, reverse osmosis, and advanced oxidation processes for infiltration at a recharge facility near the Santa Ana River (Orange County Water District, 2004, 5-14). Another 30,000 acre-ft/yr of treated wastewater is injected into seawater intrusion barrier wells at the Talbert Gap to prevent further movement of seawater into the aquifer. These sources of recharge are not included in the groundwater budget listed in table 5.

Under natural, predevelopment conditions, the Coastal Basin was full to overflowing, with discharge occurring primarily by evapotranspiration and springs, including submarine springs (Poland and Piper, 1956, p. 50). Natural discharge varied in response to changes in precipitation on the Santa Ana River watershed and the resulting recharge to the groundwater basin. Subsurface flow out of the basin across the Orange/Los Angeles County line as a result of higher water levels under predevelopment conditions was simulated at about 36,000 acre-ft/yr (Orange County Water District, 2004, fig. 2-4).

Increased withdrawals from large-capacity wells pumped for irrigation, coupled with generally below-normal precipitation from 1917–36, cumulatively caused water levels in wells to drop to near or below sea level in much of the Coastal Basin by 1936 (Poland, 1959, p. 11). Seawater intruded into the aquifer along the western margin of the basin due to the decline in groundwater levels, and injection wells were installed in the bedrock gaps to create a hydraulic barrier between the seawater and the basin-fill aquifer containing freshwater. Generally above-normal precipitation from 1937–44 resulted in water level rises and the return of flowing-well conditions to some areas near the coast.

Beginning in about 1940, the urban population began to steadily increase along with water use for municipal and industrial purposes. Agricultural water uses in Orange County decreased from 100,000 acre-ft in 1954 to 10,000 acre-ft in 2004, while the population increased from 300,000 to 2,300,000 (Orange County Water District, 2004, p. 4-1). Withdrawals from wells increased steadily from about 150,000 acre-ft in 1954 to about 350,000 acre-ft in 2002 (Orange County Water District, 2004, fig. 1-3). There are about 500 active production wells in the Coastal Basin with approximately 200 large-capacity public-supply wells accounting for 97 percent of the total production in 2001–02 (Orange County Water District, 2004, p. 2-8). Discharge from the groundwater system under modern conditions is almost completely through pumping from wells (327,000 acre-ft/yr) in the balanced water budget (Orange County Water District, 2004, table 2-1) (table 5). The remaining 8,000 acre-ft/yr flows out of the basin in the subsurface across the Orange/Los Angeles County line (Orange County Water District, 2004, table 2-1).

Groundwater Flow

On a regional scale, groundwater in the Coastal Basin moves from areas of unconfined conditions in the forebay area westward to areas of confined conditions in the pressure area. This pattern of groundwater flow in the Coastal Basin can be conceptualized as a slice of pie, starting from a small area at the Santa Ana River and its recharge facilities and expanding outward toward the coast (Shelton and others, 2001, p. 13). Under predevelopment conditions, recharge entered the relatively thin, coarse-grained basin-fill deposits along the mountain-front stream channels, moved laterally and vertically into thicker deposits in the middle of the basin, and eventually was forced towards the land surface by the sedimentary rock offset along the Newport-Inglewood Fault Zone near the ocean (fig. 15). Layers of fine-grained sediment serve to confine the aquifer system in the pressure area resulting in artesian conditions where groundwater levels once were at or above the land surface in wells, springs, and seepage areas.

The artesian area for the Coastal Basin groundwater system under predevelopment conditions (prior to about 1870) was estimated by Mendenhall (1905a, plate 1) to cover about 154 mi^2, almost 75 percent of the pressure area, and extended more than 10 mi inland from the coastline in the central part of the basin (fig. 16). By August 1904, the artesian area in the Coastal Basin had decreased to about 111 mi^2, corresponding to a reduction in artesian pressure in the aquifer caused by the installation of many flowing wells.

The groundwater surface for the principal aquifer in the Coastal Basin for 2005 constructed by the Orange County Water District (2006, plate 1) indicates that recharge occurring near and along the Santa Ana River and Santiago Creek moves southwestward towards the coast (fig. 16). Water levels were below land surface throughout the basin and below sea level in approximately the western third of the basin. This is in contrast to the extent of the artesian area described in 1904 by Mendenhall (1905a, plate 1). Water-surface gradients are relatively steep along the northeast and southeast margins of the basin where little recharge occurs.

Artificial recharge and withdrawals from wells have resulted in very large vertical and lateral rates of groundwater flow through the basin-fill deposits in parts of the forebay and pressure areas. Water-quality data show that water that entered the ground at the recharge facilities extends over 11 mi into the aquifer along a studied flow path (Dawson and others, 2003, p. 37). Apparent ages of water sampled from 300–500 ft below land surface along a flow path originating at recharge basins near the Santa Ana River were determined using the tritium-helium-3 (^3H-^3He) dating method. The age distribution indicates that groundwater less than 5 years old had traveled more than one mile from the recharge basins, implying a mean linear groundwater velocity of around 2,000 ft/yr (Davisson and others, 2004, p. 89). Groundwater ages progressively increased to more than 20 years at a distance of approximately 5-6 mi west of the recharge basins (Davisson and others, 2004, fig. 28b). The decrease in linear velocity of groundwater flow with distance from the recharge basins is due to the increasing aquifer width and thickness away from the recharge basins (Clark and others, 2004, p. 170). Vertically, groundwater is less than one year old more than 500 ft below the recharge basins on the basis of ^3H-^3He age determinations (Davisson and others, 2004, p. 89). The water can move quickly into and laterally through the aquifer because a thin unsaturated zone underlies the recharge basins. Layers of lower permeability sediment, however, slow the vertical movement of water at depths of about 1,000 ft near the recharge basins.

Shallow groundwater ages beneath the Santa Ana River channel near the artificial recharge facilities varied from 1 to 10 years old (Davisson and others, 2004, fig. 29b). The large volume of annual recharge infiltrating through the channel is likely “held up” at shallow depths (less than 100 ft below land surface) by discontinuous layers of less transmissive sediments. A subsurface fault may impede the westward movement of groundwater at depth and force older groundwater upward, which also would restrict the downward movement of recharged river water. Extensive lateral flow parallel to the river, dominated by flow paths near the water table, likely moves most of the recharged river water away from the channel and into the aquifer system (Davisson and others, 2004, p. 91).

Effects of Natural and Human Factors on Groundwater Quality

Groundwater in the Coastal Basin’s forebay area has primarily been recharged since the early 1950s, and its chemical characteristics are influenced by the recharge sources. In the pressure area nearer to the forebay area, those characteristics reflect historical variations in recharge water quality and mixing with older groundwater. The quality of groundwater at the lower end of the flow system near the coast typically represents predevelopment conditions (native groundwater) but, in some areas, may be affected by seawater intrusion.

Extensive analyses of water sampled from active production and monitoring wells have been made to determine groundwater quality in the Coastal Basin. The Orange County Water District (2004, p. 3-10) collects water samples from about 200 potable-supply wells and about 225 non-potable production wells annually to meet regulatory requirements, to gain a better understanding of the aquifer system, and for special studies. Several NAWQA studies were done from 1999–2001 to assess general water-quality conditions in the Coastal Basin aquifer system and to characterize variation in groundwater quality as it moves from recharge facilities in the forebay area toward natural discharge areas near the coast (Hamlin and others, 2002, p. 14) (fig. 17, table 1).

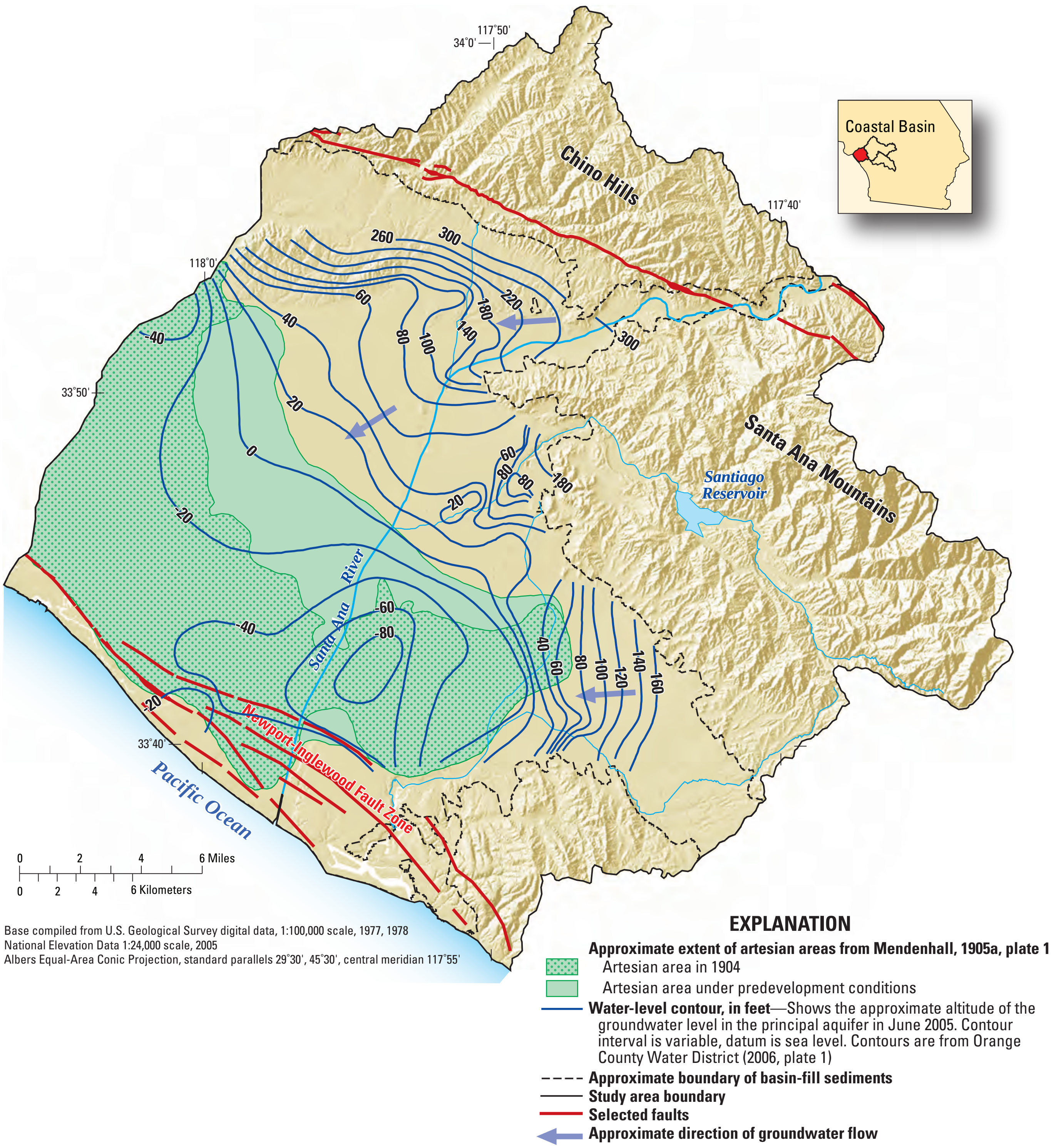


Figure 16. Groundwater levels and generalized flow directions in the Coastal Basin, California, in 2005, and artesian areas in 1904 and under predevelopment conditions.

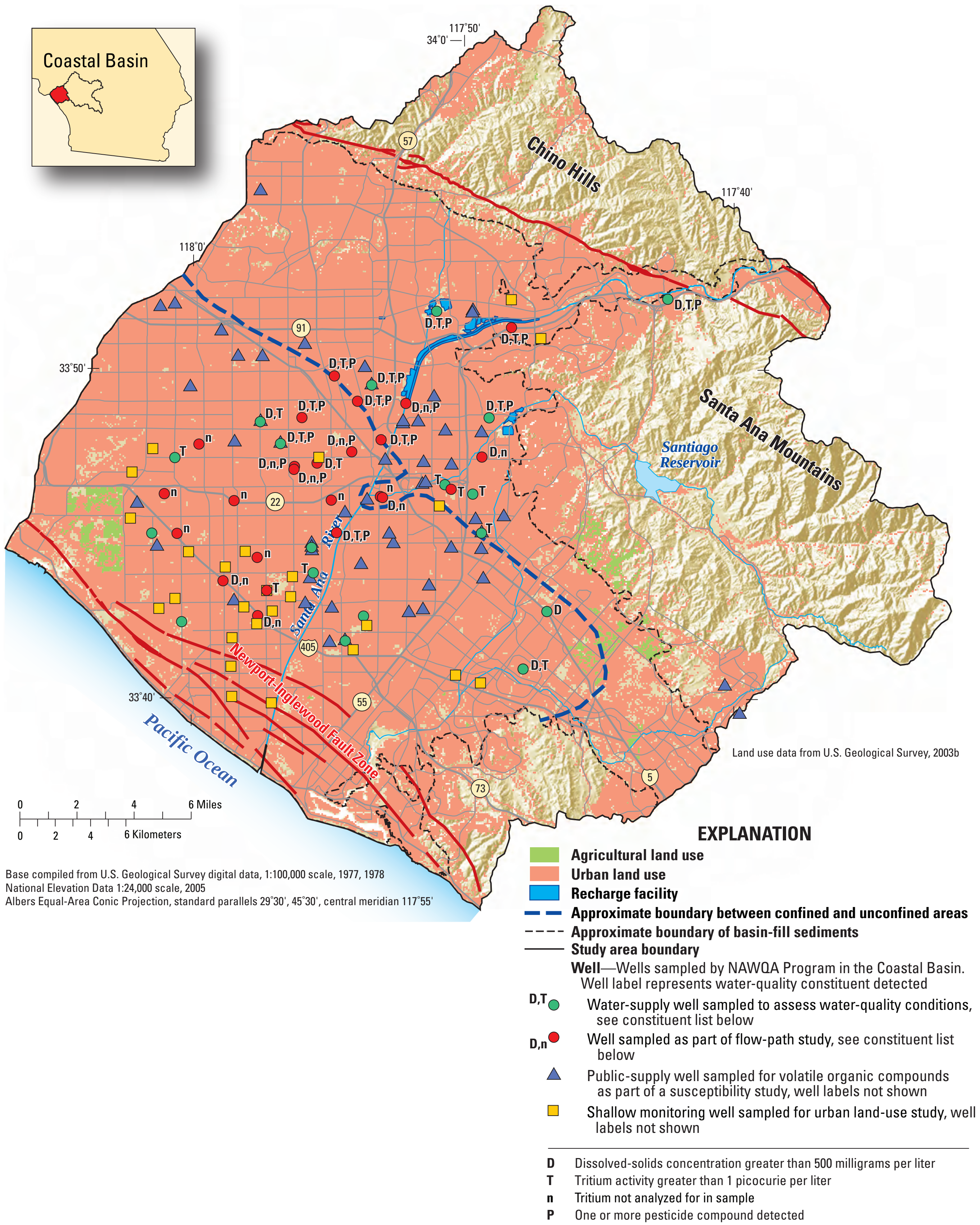


Figure 17. Locations of and chemical characteristics of water in wells in the Coastal Basin, California, sampled by the NAWQA Program, 1999–2001.

A set of 56 public-supply wells was sampled for analysis of VOCs in the water in collaboration with the California State Water Resources Control Board as part of its California Aquifer Susceptibility Program (Shelton and others, 2001; Hamlin and others, 2002). Groundwater samples collected from production wells in the Coastal Basin as part of NAWQA studies were grouped on the basis of well location in the unconfined forebay area or in the confined pressure area (table 1).

General Water-Quality Characteristics and Natural Factors

Groundwater used for public supply within the Coastal Basin is primarily a sodium-calcium bicarbonate type (California Department of Water Resources, 1967). The concentration of dissolved solids in groundwater varies with depth and location and its general spatial distribution in the used part of the aquifer is shown in figure 2. Although the basin is highly urbanized, wells in the pressure area are screened in confined aquifers that are protected to a degree from the effects of overlying land uses by layers of clay and silt. Deeper groundwater in much of the western part of the pressure area has concentrations of dissolved solids less than 400 mg/L that are associated with recharge that occurred prior to development in the basin (Orange County Water District, 2004, fig. 6-4) and are indicative of recharge from mountain-front and storm runoff in the forebay area. The highest concentrations of dissolved solids in groundwater are found in the Irvine area and along the coast in association with seawater intrusion. The concentration of dissolved solids in groundwater recharged along the mountain front in the Irvine area exceeds 1,000 mg/L, due to leaching of salts from marine sediments in the Santa Ana Mountains (Singer, 1973). High concentrations of dissolved solids in the Irvine area also reflect the effects of past and current agricultural practices.

The deep aquifer system in the western part of the basin sometimes produces water colored with an amber tint imparted by natural organic material buried with the coastal plain deposits, an unpleasant odor due to the presence of hydrogen sulfide, and a slightly warmer than average temperature. On the basis of its position along regional flow paths, this water was recharged prior to development in the basin and contains relatively low concentrations of dissolved solids (a median value of 240 mg/L is listed for water from 7 wells by the Orange County Water District (2004, table 6-5)). Some water from the deep aquifer system is now treated to remove color and odor and is distributed for public supply (Mesa Consolidated Water District, 2005).

Water from 41 deeper wells in the Coastal Basin sampled by NAWQA (well classes G and H in table 1) had dissolved-solids concentrations that ranged from 215 to 868 mg/L (water from one of the wells in class H was not analyzed for dissolved solids). The range in dissolved-solids concentrations in water from 26 shallow monitoring wells sampled by NAWQA is large (432- 25,500 mg/L with a median value of 2,390 mg/L) and is affected by activities at the land surface, by seawater intrusion, and by the upward movement of water from deeper aquifers in the pressure area (Hamlin and others, 2002, p. 21). The monitoring wells, which were constructed to sample the upper 10 to 15 ft of the aquifer system, ranged in depth from 18.5 to 143.5 ft with a median depth of 24 ft (well class I in table 1).

Two naturally occurring elements, arsenic and uranium, can affect the suitability of water for drinking. Concentrations of dissolved arsenic in water sampled by NAWQA from 20 water-supply wells to assess water-quality conditions in the basin ranged from less than 1.0 to 5.7 µg/L with a median value of 1.4 µg/L. However, 5 of the 25 shallow monitoring wells (well class I) sampled for arsenic had concentrations greater than 10 µg/L (11.2 to 37.4 µg/L) (Hamlin and others, 2002, appendix 6). Concentrations of dissolved uranium in the 20 water-supply wells ranged from less than 1.0 to 16.1 µg/L with a median value of 4.4 µg/L (Hamlin and others, 2002, appendix 8). Water from 48 percent of the shallow monitoring wells sampled for uranium had concentrations greater than 30 µg/L (43.2 to 312 µg/L). These wells are in a historically marshy area in which geochemical conditions and evaporation may tend to concentrate some trace elements (Hamlin and others, 2002, p. 34).

Potential Effects of Human Factors

Groundwater near the recharge basins and the Santa Ana River reflects the quality of recently recharged water. Concentrations of dissolved solids have increased in water from public-supply wells in much of the basin as a result of recharge water with relatively high concentrations of dissolved solids from the Santa Ana River and imported from the Colorado River. Streamflow in the Santa Ana River is affected by increased urban development in its watershed and by greater discharges of treated wastewater resulting from increases in population. The Orange County Water District began large-scale recharge to the Coastal Basin using water imported from the Colorado River in the early 1950s, and that water was the dominant source of recharge from about 1957 to 1971 (Wildermuth Environmental, Inc., 2000, p. 6-4). The imported water historically had higher concentrations of dissolved solids (about 700 mg/L) than the native groundwater, and as a consequence, concentrations of dissolved solids in groundwater began to increase (Herndon and others, 1997). Subsequently, alternative water supplies with lower concentrations of dissolved solids were developed to minimize the use of Colorado River water for aquifer recharge. During 1995–96, the Orange County Water District recharged water imported from northern California with an average dissolved-solids concentration of 321 mg/L (Herndon and others, 1997). Although imported water from northern

California has a lower concentration of dissolved solids than Colorado River water, it contains higher concentrations of organic carbon that may produce trihalomethanes (includes the compound chloroform) when the water is disinfected by chlorination. Local increases in concentrations of dissolved solids may also be related to the downward migration of shallow groundwater that has been affected by past agricultural and industrial activity (Orange County Water District, 2004, p. 6-6).

Sources of nitrate in water from public-supply wells in the Coastal Basin include recharge from the Santa Ana River (nitrate (as nitrogen) concentrations range from about 2 to 8 mg/L) and infiltration of water affected by past and present-day human activities (Herndon and others, 1997). Past agricultural land uses, such as pastures, livestock holding, cropland, vineyards, and orchards, are a major cause of elevated concentrations of nitrate detected in Coastal Basin groundwater (Orange County Water District, 2004, p. 6-1). Concentrations of nitrate (as nitrogen) typically range from 1 to 4 mg/L in the confined pressure area and from 4 to 7 mg/L in the unconfined forebay area (Orange County Water District, 2004, p. 6-4). The deeper production wells sampled by NAWQA in the confined part of the aquifer had a median concentration of nitrate (as nitrogen) of 1.4 mg/L compared to 4.3 mg/L for water sampled from production wells in the unconfined part (table 1), but none of the concentrations exceeded the drinking-water standard of 10 mg/L. Only one third of the shallow monitoring wells sampled by NAWQA had concentrations of nitrate (as nitrogen) greater than 1.0 mg/L, likely due to reducing conditions in parts of the shallow aquifer system.

Pesticides were detected in 56 percent of the NAWQA-sampled production wells in the Coastal Basin forebay area and in 23 percent of the production wells in the pressure area (table 1 and fig. 18). While all of the pesticide concentrations were very small (Hamlin and others, 2002, appendixes 9A, 9C, 10A, and 10C) and well below applicable drinking-water standards, concentrations and detection frequencies generally were highest in groundwater in the forebay area near the recharge facilities and decreased downgradient along the flow paths. In addition, the number of pesticides detected per well was significantly higher in the forebay area than in the pressure area. The occurrence of trace concentrations of pesticides in water from wells completed in the unconfined forebay area may be related to recharge at the spreading basins that utilize water from the Santa Ana River and to applications of pesticides in the forebay area. The lower detection frequency for wells in the confined pressure area probably results from mixing of younger water with pesticides with older water without pesticides and possibly degradation and adsorption of pesticides along the longer flow paths.

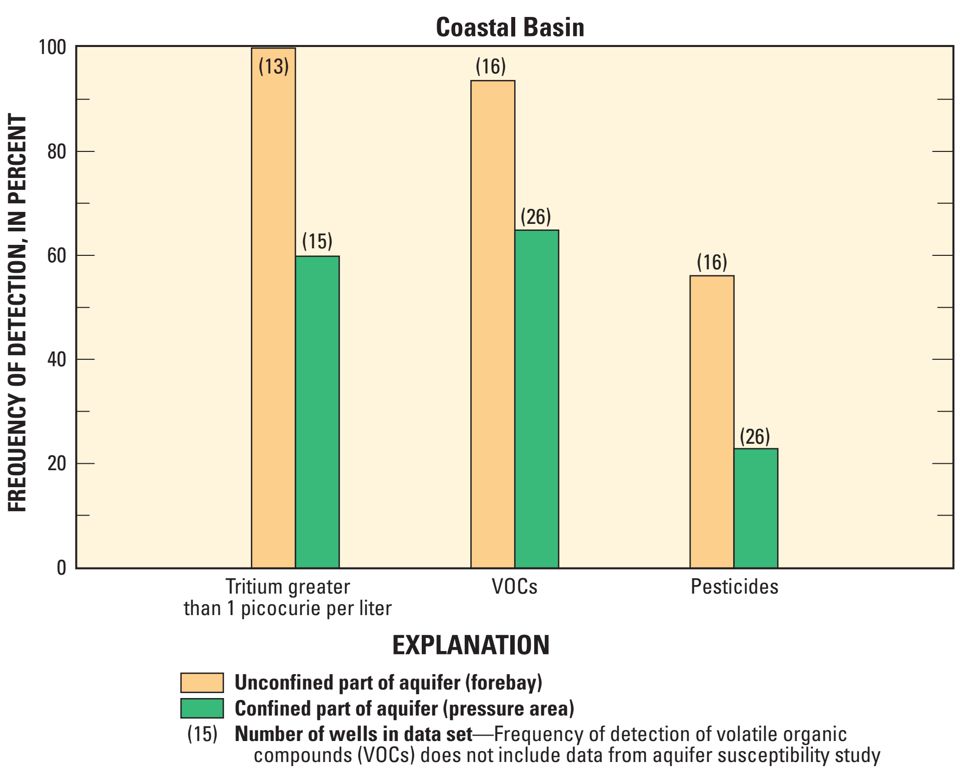


Figure 18. Detection frequencies of tritium, volatile organic compounds, and pesticides in water samples from wells in the Coastal Basin, California.

The most commonly detected pesticides in Coastal Basin groundwater (atrazine, simazine, tebuthiuron, and the degradation product deethylatrazine) were among the most frequently detected pesticides in the Santa Ana River below Prado Dam (Izbicki and others, 2000). Pesticides were detected in 69 percent of the shallow monitoring wells sampled by NAWQA in the basin (table 1) at very small concentrations (Hamlin and others, 2002, appendix 9B). The detection of prometon and tebuthiuron in water from the shallow wells, pesticides commonly used in urban areas, reflects the urban land use in the vicinity of these wells.

Many of the water-supply wells in the Coastal Basin have been sampled by the Orange County Water District for analyses of VOCs (Orange County Water District, 2004, p. 3-13). Areas with concentrations of VOCs near or above the drinking-water standards, mainly the chlorinated solvents trichloroethene (TCE) and perchloroethene (PCE), have been delineated in the shallow part of the aquifer system in the forebay area of the basin (Orange County Water District, 2004, p. 6-15). Work is underway to prevent the further movement of VOC-contaminated groundwater in the area.

Water samples collected by the NAWQA Program from production wells in the unconfined forebay areas had a higher detection frequency of VOCs (80 percent) than wells in the adjacent confined pressure area of the aquifer system (64 percent) (table 1). All of the detections were below drinking-water standards, most at very small concentrations (Hamlin and others, 2002, appendixes 11A, 11C, 11D, 12A, 12C, and 12D). Many of the production wells are downgradient from engineered recharge facilities along the Santa Ana River. VOCs may be introduced in the coarse-grained forebay areas either at the recharge facilities or in other sources of recharge that have encountered point or nonpoint contaminant sources (Shelton and others, 2001; Dawson and others, 2003). However, because of changes in the source and quality of recharge water over time, the chemical characteristics of the groundwater in the flow system is not the same as that of water currently entering the aquifer at the recharge facilities.

The most commonly detected VOCs in samples collected by NAWQA from production wells in the forebay area were chloroform, 1,1,1-trichloroethane (TCA), and the gasoline additive methyl *tert*-butyl ether (MTBE). The most commonly detected VOCs in samples from productions wells in the pressure area were chloroform and the refrigerants CFC-113 and CFC-11. Chloroform and MTBE were the most frequently detected VOCs in shallow monitoring wells in the basin sampled by NAWQA. The source of chloroform is likely chlorinated water and the source of MTBE is probably atmospheric deposition and proximity to leaky underground storage tanks.

The spatial distribution of VOCs detected above the laboratory reporting level (LRL) in groundwater sampled by NAWQA (fig. 19) was quantified in terms of the distance between recharge facilities and the location of the well. This distance is assumed to be the distance traveled along a flow path and is used as a surrogate for the time of travel (Shelton and others, 2001, p. 14). Samples with 2 or more VOC detections in the Coastal Basin are from wells within about 11 mi of the recharge facilities, with one exception, and only one or no VOCs were detected in wells sampled beyond this distance (Shelton and others, 2001, fig. 4).

Statistical analysis indicates a significant difference in the number of VOC detections in groundwater with depth in the forebay area of the Coastal Basin, but not in the pressure area (Shelton and others, 2001, p. 18). This indicates that there could be a vertical component of transport in the forebay area, but that the greater thickness of fine-grained layers likely impedes the downward movement of VOCS in the pressure area.

Stable isotope data presented by Shelton and others (2001, fig. 7) support the interpretation that VOCs detected in groundwater in the forebay area are associated with water introduced at the recharge facilities. Stable isotope composition indicates that groundwater containing VOCs is a mixture of local precipitation, runoff, and water that is isotopically lighter than the local sources. The isotopically lighter water could either be Colorado River water or northern California water, both of which have been imported to the basin and used as a source of groundwater recharge.

Tritium activity in water greater than 1 pCi/L is widespread in the Coastal Basin aquifer system, but is more prevalent in the unconfined part (fig. 18) indicating that groundwater in the forebay area is younger than groundwater in the downgradient pressure area. Data from the NAWQA studies indicate that pesticides were detected in almost 40 percent of the younger water samples, but in none of the older samples. VOCs were detected in more than 90 percent of the samples containing tritium, but in only 50 percent of the samples with tritium activities less than 1 pCi/L (Hamlin and others, 2005, p. 27). Pumping and engineered recharge in the Coastal Basin have caused the lateral rate of flow in the aquifer system to increase and are likely the dominant factors in controlling the distribution of VOCs in active public-supply wells.

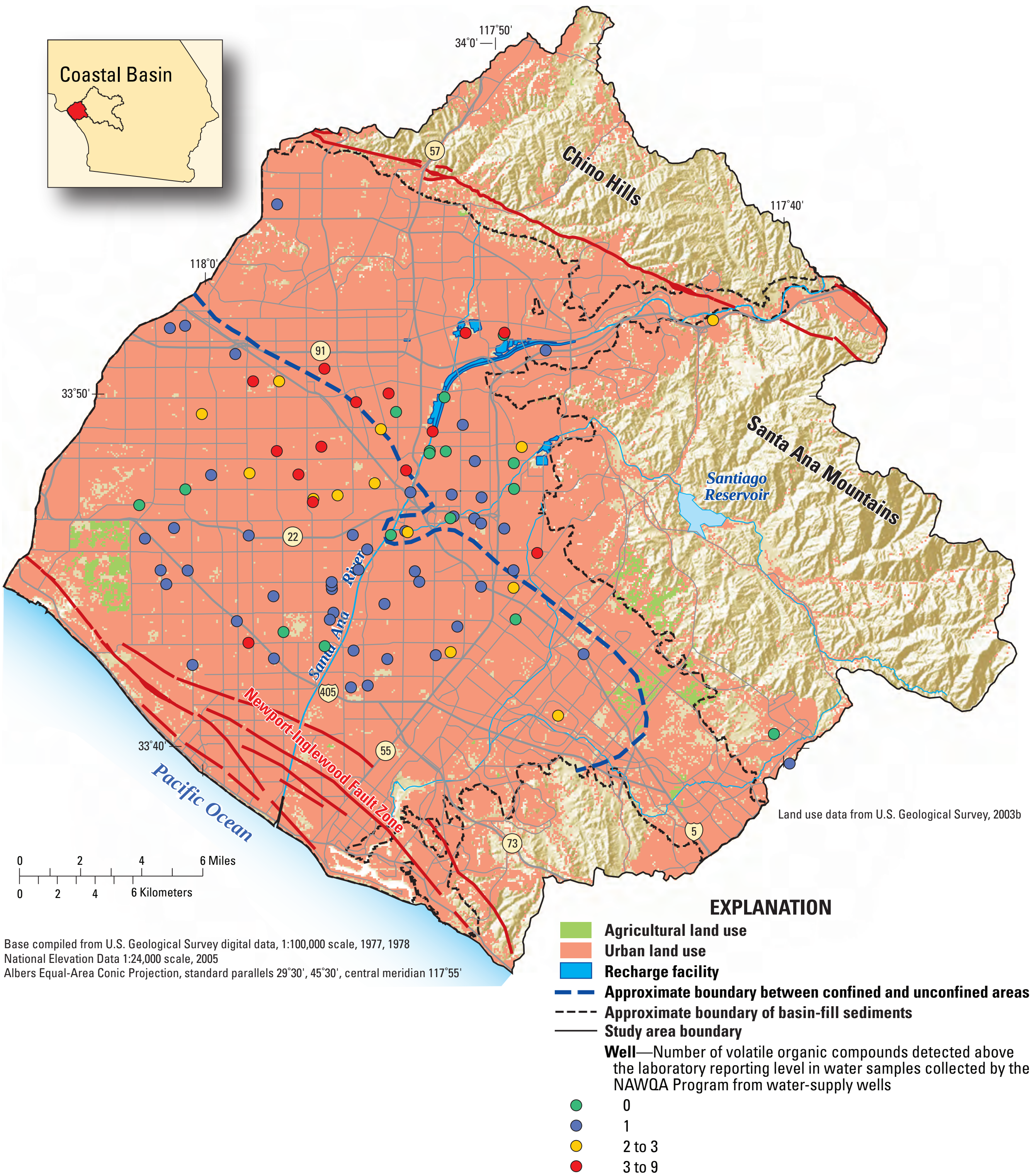


Figure 19. Public-supply wells sampled in the Coastal Basin, California, and the number of volatile organic compounds detected at concentrations above the laboratory reporting level in each well.

Summary

The hydrologic cycles of the groundwater basins in the Santa Ana Basin are greatly affected by human activities as a result of the semiarid climate and the water demands of the large urban population. The drainage basin has one of the fastest growing populations in California and was home to about 5.1 million people in 2000. Groundwater pumped from the basin is the major water supply, providing about two-thirds of the total water used. Imported water from northern California and the Colorado River also are important sources of the water supply. Pumping and additional sources of recharge have accelerated groundwater flow and the transport of dissolved constituents through the aquifers in the three distinct groundwater basins within the larger Santa Ana Basin—the San Jacinto, Inland, and Coastal Basins. Major water-quality issues in the Santa Ana Basin are elevated concentrations of dissolved solids, nitrate, perchlorate, and VOCs in groundwater.

The groundwater system in the San Jacinto Basin is largely unconfined and the overlying land use is a mixture of undeveloped rangeland, urban, and agricultural land. The amount and source of recharge to the basin-fill aquifer affects the quality of the groundwater. Subbasins that receive a large percentage of recharge from mountain-front runoff carried by the San Jacinto River have groundwater quality that is typically similar to that of the recharged water. Some areas in the basin receive little recharge and others receive a large percentage from excess irrigation water. Groundwater in these areas is affected by mineral concentration resulting from evapotranspiration. Agricultural and urban land uses and the extensive use of imported water also affect groundwater quality in the San Jacinto Basin. Withdrawals from wells have altered groundwater-flow directions in areas, allowing new sources of recharge and the movement of poorer quality groundwater to have an effect on water quality.

Faults bound and divide the mostly urban Inland Basin into several groundwater subbasins. These interior faults locally restrict groundwater flow and control the location of natural groundwater discharge. The groundwater basins are generally unconfined near the mountain fronts, where mountain runoff is distributed and recharged through natural streambeds and engineered recharge facilities. Confined conditions typically occur downgradient from the mountain fronts and at greater depths due to finer grained deposits interlayered with more permeable sand and gravel. The Bunker Hill subbasin covers 112 mi^2 in the northeastern part of the Inland Basin, but has a mountain drainage area of 466 mi^2. Three-fourths of the estimated average annual recharge to the basin-fill aquifer is from the infiltration of mountain runoff. The Chino subbasin covers about 222 mi^2, but unlike the Bunker Hill subbasin, has a mountain drainage area of only 62 mi^2. Recharge to the groundwater system in the Chino subbasin is by infiltration of precipitation and water applied at the land surface, infiltration of streamflow (including stormwater runoff), subsurface inflow from adjacent subbasins, and infiltration of imported water and treated municipal wastewater at artificial recharge facilities.

Agricultural and urban development have caused changes in groundwater quality in the Inland Basin, primarily increased concentrations of dissolved solids and nitrate and the presence of VOCs and perchlorate. The basin-fill aquifers are susceptible to water-quality changes because of the unconfined conditions in much of the area and are vulnerable to contamination because of the overlying land uses and activities that utilize chemicals and water. Dissolved solids in Chino subbasin groundwater has increased primarily due to evaporative concentration after irrigation, from the leaching of fertilizer and manure in agricultural areas, and from the recharge of treated wastewater and imported water. In general, sources of nitrate to groundwater include leaching of fertilizers and animal wastes applied to the land, leakage from sewer pipes, and reuse of treated wastewater. Another source of nitrate in groundwater in the Chino subbasin is infiltration of wastewater from animal feeding facilities.

Numerous contaminant plumes (mainly VOCs and perchlorate) in the Inland Basin that are related to industrial activities extend several miles from the source of contamination and have affected the operation of many public-supply wells in the basin. Pesticides and VOCs were frequently detected in water sampled by NAWQA from wells in the Inland Basin. The large number of detections of these compounds probably reflects generally unconfined conditions in the groundwater system, present and past land uses, and the relatively low organic content of aquifer materials.

The mostly urban Coastal Basin includes a relatively small unconfined recharge area and a relatively large confined area where pumping is now the predominant form of discharge from the groundwater system. The groundwater quality is affected by enhanced recharge of water from the Santa Ana River, including the infiltration of treated wastewater that now is a large component of base flow in the river, and the infiltration of imported water. On a regional scale, groundwater in the Coastal Basin moves from unconfined conditions in the forebay area westward to confined conditions in the pressure area. Wells in the pressure area are screened in confined aquifers that are protected to a degree from the effects of overlying land uses by layers of clay and silt.

Artificial recharge and pumping have resulted in very large vertical and lateral rates of groundwater flow through the basin-fill deposits in parts of the forebay and pressure areas. Water-quality data show that water that entered the ground at the recharge facilities extends over 11 miles into the aquifer system along a studied flow path. The quality of groundwater at the lower end of the flow system near the coast typically represents predevelopment conditions but, in some areas, may be affected by seawater intrusion. Groundwater quality in the pressure area nearer to the forebay area reflects historical variation in recharge water quality and mixing with naturally recharged groundwater. Concentrations of dissolved solids have increased in water from public-supply wells in much of the basin as a result of recharge water with relatively high concentrations of dissolved solids from the Santa Ana River and imported from the Colorado River. Sources of nitrate in water from public-supply wells in the Coastal Basin include recharge from the Santa Ana River and infiltration of water affected by past and present human activities.

Production wells sampled by NAWQA in the unconfined forebay areas of the Coastal Basin had a higher detection frequency of VOCs (82 percent) than wells in the adjacent confined pressure area of the aquifer system (65 percent). Groundwater samples with 2 or more VOC detections are from wells within about 11 miles of the recharge facilities, with one exception, and only one or no VOCs were detected in wells sampled beyond this distance in the basin.

References Cited

Anning, D.W., Bauch, N.J., Gerner, S.J., Flynn, M.E., Hamlin, S.N., Moore, S.J., Schaefer, D.H., Anderholm, S.K., and Spangler, L.E., 2007, Dissolved solids in basin-fill aquifers and streams in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2006–5315, 168 p. Available at http://pubs.usgs.gov/sir/2006/5315/.

Belitz, K., Hamlin, S.N., Burton, C.A., Kent, R.H., Fay, R.G., and Johnson, T.J., 2004, Water quality in the Santa Ana Basin California, 1999–2001: U.S. Geological Survey Circular 1238, 37 p. Available at http://pubs.usgs.gov/circ/2004/1238/.

Burton, C.A., Kaehler, C.A., and Christensen, A.H., 1996, Well-construction, water-quality, and water-level data, and pond-infiltration estimates, for three ground-water subbasins, Riverside County, California: U.S. Geological Survey Water-Resources Investigations Report 96–4294, 114 p.

California Department of Water Resources, 1967, Progress report on ground water geology of the Coastal Plain of Orange County, 138 p.

California Department of Water Resources, 2003a, California's groundwater—San Jacinto groundwater basin: Bulletin 118, updated 1/20/06, accessed December 1, 2010 at http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/8-5.pdf.

California Department of Water Resources, 2003b, California's groundwater—South Coast hydrologic region: Bulletin 118, updated 2003, accessed December 1, 2010 at http://www.water.ca.gov/pubs/groundwater/bulletin_118/california's_groundwater__bulletin_118_-_update_2003_/bulletin118_4-sc.pdf.

California Department of Water Resources, 2003c, California's groundwater—Upper Santa Ana Valley groundwater basin, Chino Subbasin: Bulletin 118, updated 1/20/06, accessed December 1, 2010 at http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/8-2.01.pdf.

California Department of Water Resources, 2003d, California's groundwater—Upper Santa Ana Valley groundwater basin, Bunker Hill Subbasin: Bulletin 118, updated 2/27/04, accessed July 7, 2009 at http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/8-2.06.pdf.

California Department of Water Resources, 2003e, California's groundwater—Coastal Plain of Orange County groundwater basin: Bulletin 118, updated 2/27/2004, accessed December 1, 2010 at http://www.water.ca.gov/pubs/groundwater/bulletin_118/basindescriptions/8-1.pdf.

Clark, J.F., Hudson, G.B., Davisson, M.L., Woodside, Greg, and Herndon, Roy, 2004, Geochemical imaging of flow near an artificial recharge facility, Orange County, California: Ground Water, vol. 42, no. 2, p. 167–174.

Danskin, W.R., McPherson, K.R., and Woolfenden, L.R., 2006, Hydrology, description of computer models, and evaluation of selected water-management alternatives in the San Bernadino area, California: U.S. Geological Survey Open-File Report 2005–1278, 178 p. and 2 plates. Available at http://pubs.usgs.gov/of/2005/1278/.

Davisson, M.L., Hudson, G.B., Clark, J.F., Woodside, Greg, and Herndon, Roy, 2004, Final report on isotope tracer investigations in the forebay of the Orange County groundwater basin: Lawrence Livermore National Laboratory UCRL–TR–201735, 115 p.

Dawson, B.J.M., Belitz, K., Land, M.T., and Danskin, W.R., 2003, Stable isotopes and volatile organic compounds along seven ground-water flow paths in divergent and convergent flow systems, southern California, 2000: U.S. Geological Survey Water-Resources Investigations Report 03–4059, 79 p. Available at http://water.usgs.gov/pubs/wri/wrir034059/.

Dutcher, L.C., and Garrett, A.A., 1963, Geologic and hydrologic features of the San Bernardino area, California—With special reference to underflow across the San Jacinto Fault: U.S. Geological Survey Water-Supply Paper 1419, 114 p.

Eastern Municipal Water District, 2005, Eastern Municipal Water District urban water management plan 2005, accessed April 29, 2008 at http://www.emwd.org/news/pubs_uwmp.html.

Eastern Municipal Water District, 2007a, West San Jacinto groundwater basin management plan—2006 annual report, accessed April 29, 2008 at http://www.emwd.org/news/pubs_sj-subbasin.html.

Eastern Municipal Water District, 2007b, Hemet/San Jacinto Water Management Area—2006 annual report, accessed April 29, 2008 at http://www.emwd.org/news/pubs_hemet-subbasin.html.

French, J.J., 1972, Ground-water outflow from the Chino Basin, Upper Santa Ana Valley, southern California: U.S. Geological Survey Water-Supply Paper 1999–G, 28 p.

Hamlin, S.N., Belitz, K., Kraja, S., and Dawson, B.J., 2002, Ground-water quality in the Santa Ana Watershed, California: Overview and data summary: U.S. Geological Survey Water-Resources Investigations Report 02–4243, 137 p. Available at http://pubs.usgs.gov/wri/wri02-4243/.

Hamlin, S.N., Belitz, K., and Johnson, T., 2005, Occurrence and distribution of volatile organic compounds and pesticides in ground water in relation to hydrogeologic characteristics and land use in the Santa Ana Basin, southern California: U.S. Geological Survey Scientific Investigations Report 2005–5032, 40 p. Available at http://water.usgs.gov/pubs/sir/2005/5032/.

Hardt, W.F., and Freckleton, J.R., 1987, Aquifer response to recharge and pumping, San Bernardino ground-water basin, California: U.S. Geological Survey Water-Resources Investigations Report 86–4140, 69 p.

Herndon, R.L., Brukner, D.B., and Sharp, G., 1997, Groundwater systems in the Orange County groundwater basin, Phase 1A Task 2.2 Report, prepared for the Santa Ana Watershed Project Authority, TIN/TOS Task Force: Orange County Water District, 12 p.

Hughes, C.L., 1992, 2–D modeling of heat flow in the San Bernardino Valley, southern California: Riverside, University of California, M.S. thesis, 169 p.

Izbicki, J.A., Mendez, G.O., Burton, C.A., 2000, Stormflow chemistry in the Santa Ana River below Prado Dam and at the diversion downstream from Imperial Highway, southern California, 1995–98: U.S. Geological Survey Water-Resources Investigations Report 2000–4127, 92 p.

Kaehler, C.A., and Belitz, K., 2003, Tracing reclaimed water in the Menifee, Winchester and Perris-South ground-water subbasins, Riverside County, California: U.S. Geological Survey Water-Resources Investigations Report 03–4039, 49 p. Available at http://water.usgs.gov/pubs/wri/wri034039/.

Kaehler, C.A., Burton, C.A., Rees, T.F., and Christensen, A.H., 1998, Geohydrology of the Winchester Subbasin, Riverside County, California: U.S. Geological Survey Water-Resources Investigations Report 98–4102, 90 p.

Kent, Robert, and Belitz, Kenneth, 2004, Concentrations of dissolved solids and nutrients in water sources and selected streams of the Santa Ana Basin, California, October 1998 to September 2001: U.S. Geological Survey Water-Resources Investigations Report 03–4326, 61 p. Available at http://pubs.usgs.gov/wri/wrir034326/.

MacRostie, W. and Dolcini, A.J., 1959, Santa Ana River investigation: California Department of Water Resources Bulletin No. 15, 194 p.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239, 16 p. Available at http://pubs.usgs.gov/sir/2008/5239/.

McMahon, P.B. and Chapelle, F.H., 2008, Redox processes and water quality of selected principal aquifer systems: Ground Water, vol. 46, p. 259–271.

Mendez, G.O., and Belitz, Kenneth, 2002, Identifying sources of base flow in the Santa Ana River, CA: American Water Resources Association 2002 summer specialty conference proceedings—Ground water/surface water interactions, p. 567–572.

Mendenhall, W.C., 1905a, Development of underground waters in the central coastal plain region of southern California: U.S. Geological Survey Water-Supply and Irrigation Paper 138, 162 p.

Mendenhall, W.C., 1905b, Hydrology of San Bernardino Valley, California: U.S. Geological Survey Water-Supply Paper 142, 124 p.

Mesa Consolidated Water District, 2005, Colored water treatment facility and wells 6 and 11 (revised May 2005): Mesa Consolidated Water District Well Fact Sheet, accessed on February 29, 2008 at http://www.mesawater.org/pdf/fact_sheets/factsheet6_11.pdf.

Metropolitan Water District of Southern California, 2007, Groundwater assessment study: Metropolitan Water District of Southern California Report Number 1308, accessed October 2, 2008 at http://www.mwdh2o.com/mwdh2o/pages/yourwater/supply/groundwater/gwas.html.

Miller, R.E., and Singer, J.A., 1971, Subsidence in the Bunker Hill-San Timoteo area, southern California: U.S. Geological Survey Open-File Report 71–209, 28 p.

Morton, D.M., and Miller F.K., 2006, Geologic map of the San Bernardino and Santa Ana 30' x 60' quadrangles, California, U.S. Geological Survey Open File Report 2006–1217. Available at http://pubs.usgs.gov/of/2006/1217.

Oak Ridge National Laboratory, 2005, LandScan Global Population Database, accessed May 17, 2010 at http://www.ornl.gov/landscan/.

Orange County Water District, 2004, Ground-water management plan, accessed May 17, 2010 at http://www.ocwd.com/Publications—-Newsletters/ca-43.aspx.

Orange County Water District, 2006, 2004–2005 Engineer's report on groundwater conditions, water supply and basin utilization in the Orange County Water District.

Orange County Water District, 2008, Orange County Water District: OCWD Fact Sheet—May 2008, accessed on May 6, 2009 at http://www.ocwd.com/Publications—-Newsletters/ca-43.aspx.

Planert, M., and Williams, J.S., 1995, Ground water atlas of the United States—California, Nevada: U.S. Geological Survey Hydrologic Atlas HA 730–B.

Poland, J.F., 1959, Hydrology of the Long Beach-Santa Ana area, California: U.S. Geological Survey Water-Supply Paper 1471, 257 p.

Poland, J.F. and Piper, A.M., 1956, Ground-water geology of the coastal zone, Long Beach-Santa Ana area, California: U.S. Geological Survey Water-Supply Paper 1109, 162 p.

Post, W.S., 1928, Santa Ana investigation—Flood control and conservation: California Department of Public Works Bulletin No. 19, 357 p. accessed June 23, 2009 at http://www.archive.org/details/santaanainvestig00postrich.

PRISM Group, Oregon State University, 2004, Precipitation-elevation regression on independent slopes model (PRISM), Oregon State University, accessed May 17, 2010 at http://www.ocs.oregonstate.edu/prism/.

Rees, T.F., Bright, D.J., Fay, R.G., Christensen, A.H., Anders, Robert, Baharie, B.S., and Land, M.T., 1995, Geochemistry in the saturated and unsaturated zones beneath various land uses, Riverside and San Bernardino Counties, California, 1991–93: U.S. Geological Survey Water-Resources Investigations Report 94–4127, 267 p.

Reichard, E.G., Land, M., Crawford, S.M., Johnson, T., Everett, R.R., Kulshan, T.V., Halford, K.J., Johnson, T.A., Paybins, K.S., and Nishikawa, T., 2003, Geohydrology, geochemistry, and ground-water simulation-optimization of the Central and West Coast Basins, Los Angeles County, California: U.S. Geological Survey Water-Resources Investigations Report 03–4065, 184 p. Available at http://pubs.usgs.gov/wri/wrir034065/wrir034065.html.

Santa Ana Watershed Project Authority, 2002, Santa Ana integrated watershed plan—Volume 1 water resources component, accessed on April 7, 2009, at http://www.sawpa.org/pdf/IWRP_Complete.pdf.

Santa Ana Watershed Project Authority, 2005, Santa Ana integrated watershed plan—2005 update, 183 p.

Santa Ana Watershed Authority, no date, Major water infrastructure—Treatment and delivery: Santa Ana Watershed Authority, accessed on July 9, 2009 at http://sawpa.org/documents/MapGallery/1421_WaterRelatedInfrastructure.pdf.

Saucedo, G.L., Bedford, B. R., Raines, G. L., Miller, R. J., and Wentworth, C. M., 2000, Modified from California Division of Mines and Geology, CD-ROM 2000–007, GIS data for the geologic map of California.

Scott, M.B., 1977. Development of water facilities in the Santa Ana River Basin, California, 1810–1968, a compilation of historical notes derived from many sources describing ditch and canal companies, diversions, and water rights: U.S. Geological Survey Open-file Report 77–398, 231 p.

Schlehuber, M.J., Lee, T.C., and Hall, B.S., 1989, Groundwater level and hydrochemistry in the San Jacinto Basin, Riverside County, California: Journal of Hydrology, vol. 106, p. 79–98.

Shelton, J.L., Burow, K.R., Belitz, K., Dubrovsky, N.M., Land, M., and Gronberg, J., 2001, Low-level volatile organic compounds in active public supply wells as ground-water tracers in the Los Angeles physiographic basin, California, 2000: U.S. Geological Survey Water-Resources Investigations Report 01–4188, 29 p., accessed September 23, 2008 at http://ca.water.usgs.gov/archive/reports/wrir014188/.

Singer, J.A., 1973, Geohydrology and artificial-recharge potential of the Irvine area, Orange County, California: U.S. Geological Survey Open-File Report 73–264, 41 p.

Tompson, A.F.B., Carle, S.F., Rosenberg, N.D., and Maxwell, R.M., 1999, Analysis of ground water migration from artificial recharge in a large urban aquifer: A simulation perspective: Water Resources Research, vol. 35, no. 1, p. 2981–2998.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs, accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003a, Principal aquifers in National Atlas of the United States of America, accessed on August 26, 2009 at http://water.usgs.gov/ogw/aquiferbasics/alphabetical.html.

U.S. Geological Survey, 2003b, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

U.S. Geological Survey, 2004, Estimated use of water in the United States; county-level data for 2000: U.S. Geological Survey web site, accessed on May 5, 2010 at http://water.usgs.gov/watuse/data/2000/index.html.

Waring, G.A., 1919, Ground water in the San Jacinto and Temecula basins, California: U.S. Geological Survey Water-Supply Paper 429, 113 p.

Water Resources and Information Management Engineering Inc., 2003, Operational yield study, Hemet/San Jacinto Groundwater Management Area: prepared in coordination with the California Department of Water Resources, 36 p.

Wildermuth Environmental, Inc., 1999, Optimum Basin Management Program, Phase I Report: prepared for the Chino Basin Watermaster, August 19, 1999, accessed May 9, 2008 at http://www.cbwm.org/docs/engdocs/obmpphas1rep/Text/OBMP_Ph1_Report.pdf.

Wildermuth Environmental, Inc., 2000, TIN/TDS Phase 2A: Tasks 1 through 5, TIN/TDS study of the Santa Ana Watershed, Technical Memorandum, July 2000.

Wildermuth Environmental, Inc., 2005, Chino Basin Optimum Basin Management Program, State of the Basin Report—2004: prepared for the Chino Basin Watermaster, July 2005, accessed May 9, 2008 at http://www.cbwm.org/rep_engineering.htm.

Wildermuth Environmental, Inc., 2007a, Chino Basin Optimum Basin Management Program, State of the Basin Report—2006: prepared for the Chino Basin Watermaster, July 2007, accessed May 9, 2008 at http://www.cbwm.org/rep_engineering.htm.

Wildermuth Environmental, Inc., 2007b, 2007 CBWM groundwater model documentation and evaluation of the Peace II project description: prepared for the Chino Basin Watermaster, November 2007.

Wildermuth Environmental, Inc., 2008, Recomputation of ambient water quality in the Santa Ana Watershed for the period 1987 to 2006: Final Technical Memorandum prepared for the Basin Monitoring Program Task Force, August 2008.

Williams, A.E., 1997, Stable isotope tracers: natural and anthropogenic recharge, Orange County, California: Journal of Hydrology, vol. 201, p. 230–248.

Williams, A.E., and Rodoni, D.P., 1997, Regional isotope effects and application to hydrologic investigations in southwestern California, Water Resources Research, vol. 33, no. 7, p. 1721–1729.

Woolfenden, L.R., and Kadhim, D., 1997, Geohydrology and water chemistry in the Rialto-Colton Basin, San Bernadino County, California: U.S. Geological Survey Water-Resources Investigations Report 97–4012, 101 p.

Woolfenden, L.R., and Koczot, K.M., 2001, Numerical simulation of ground-water flow and assessment of the effects of artificial recharge in the Rialto-Colton Basin, San Bernadino County, California: U.S. Geological Survey Water-Resources Investigations Report 00–4243, 148 p. Available at http://pubs.usgs.gov/wri/wri004243/.

This page intentionally left blank.

Section 13.—Conceptual Understanding and Groundwater Quality of the Basin-Fill Aquifer in the Central Valley, California

By Susan A. Thiros

Basin Overview

The Central Valley aquifer system is contained within the basin-fill deposits in the Central Valley of California. The distribution of water in the valley has been modified to even out differences between where it naturally occurs and where the agricultural and urban demand exist. Surface water that under natural conditions mostly flowed out of the valley is now used for irrigation within the valley resulting in additional recharge to the aquifer system. Groundwater also is used extensively for irrigation and public supply. This water development has resulted in major changes to the groundwater flow system in the Central Valley, such as reversals in vertical and lateral directions of flow, which in turn, affect the groundwater quality.

The Central Valley is roughly 400 mi long, averages about 50 mi in width, and comprises about 20,000 mi^2. The Sacramento Valley occupies the northern third of the Central Valley and the San Joaquin Valley the southern two-thirds (fig. 1). The San Joaquin Valley is made up of the San Joaquin Basin in the northern part, which is drained by the San Joaquin River, and the internally drained Tulare Basin in the southern part. The Sacramento and San Joaquin Valleys are separated by a low-lying area called the Delta, where the Sacramento and San Joaquin Rivers converge and discharge through a natural outlet into San Francisco Bay on the Pacific Ocean. This is the only natural outlet for surface water from the Central Valley.

Topographically, the Central Valley is relatively flat and at low altitude compared to the surrounding mountains. The only feature of prominent relief within the valley is Sutter Buttes, a volcanic plug that rises about 2,000 ft above the valley floor near the center of the Sacramento Valley. The altitude of the boundary between unconsolidated basin-fill deposits in the valley and consolidated rock of the mountains is about 500 ft along much of the east side of the valley and ranges from 50 to 350 ft on the west side. The drainage area for the Central Valley is almost 49,000 mi^2 and includes the crest of the Sierra Nevada to the east and the Coast Ranges to the west.

The Central Valley has a Mediterranean climate, with hot, dry summers and cool, wet winters. Average annual precipitation, a value developed from the Parameter-elevation Regressions on Independent Slopes Model (PRISM) temperature data for September 1961 through September 2003, mostly ranges from 13 in. to 26 in. in the Sacramento Valley and from 6 to 18 in. in the San Joaquin Valley, and decreases from the northeast to the southwest (Faunt, Hanson, and Belitz, 2009, fig. A5). About 85 percent of the precipitation falls during November through April, and rainfall varies greatly from year to year. Average annual precipitation in the Sierra Nevada ranges from about 40 in. to more than 90 in., and increases with altitude. The Coast Ranges are not as high and have much less precipitation and smaller drainage areas available to sustain streamflow. The western part of the Central Valley is in the rain shadow of the Coast Ranges and is therefore drier than the eastern part.

The Sacramento River drains the Sacramento Valley and has more flow than the San Joaquin River. Major tributaries include the Feather, American, and Yuba Rivers. The major tributaries of the San Joaquin River include the Mokelumne, Stanislaus, Tuolumne, and Merced Rivers. The Tulare Basin in the southern part of the Central Valley receives streamflow from the Kings, Kaweah, and Kern Rivers. The natural flow of these rivers over thousands of years has deposited sediment on the slopes of alluvial fans and terminated in the topographically closed sinks Tulare Lake, Kern Lake, and Buena Vista Lake. The estimated amount of streamflow entering the Central Valley around its perimeter ranged from 10 million acre-ft in 1977 to more than 78 million acre-ft in 1983, with a median inflow of about 29 million acre-ft/yr for the period 1961–2003 (Faunt, Hanson, and Belitz, 2009, p. 46). Streamflow in the Central Valley is highly variable from year to year and is influenced by variability in climate. Most of the flow originates as snowmelt runoff from the Sierra Nevada during January through June and most of the surface-water flow is controlled by dams, which capture and store the water for use during the dry season. Below the dams, a complex network of streams and canals distribute the water throughout the valley.

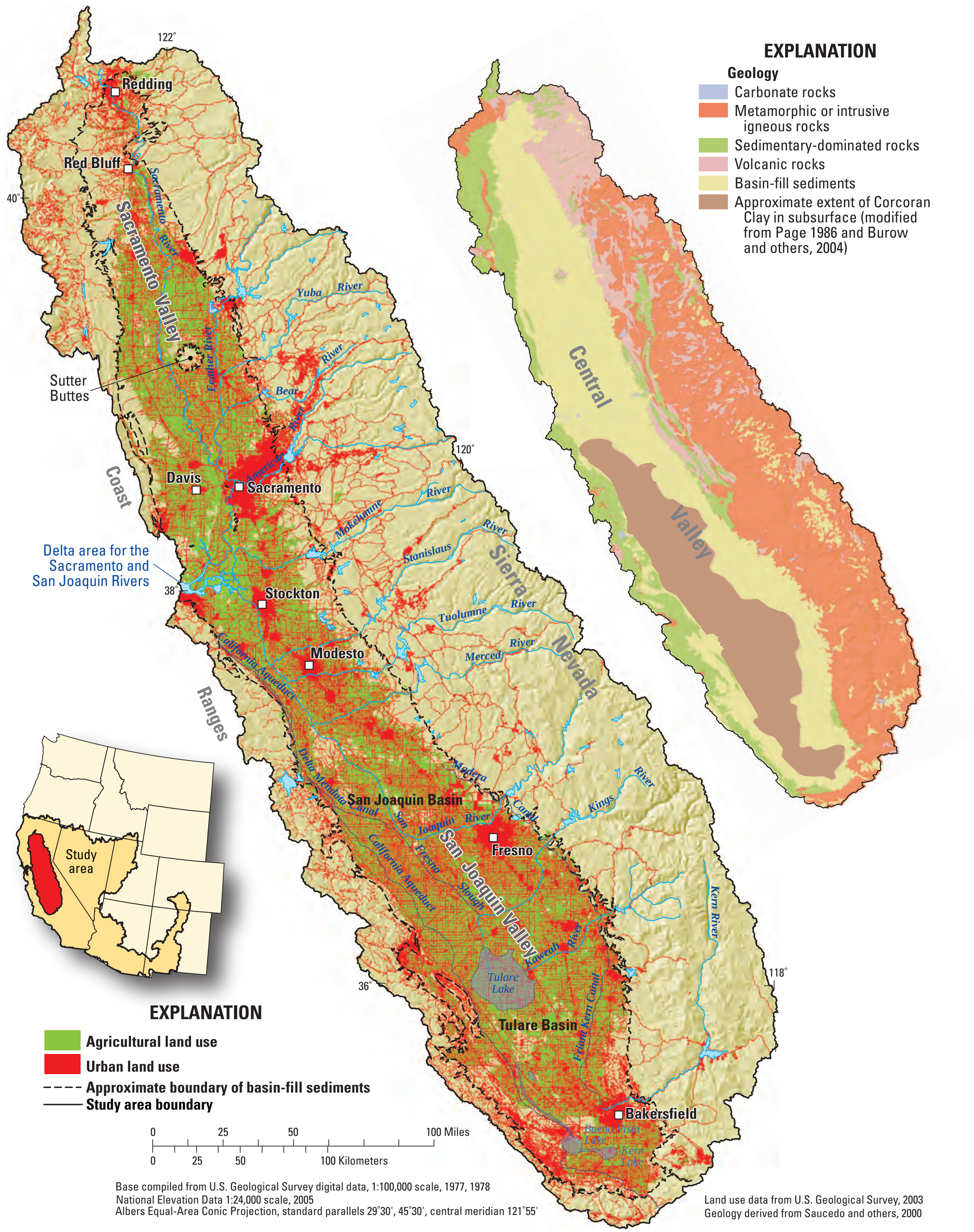


Figure 1. Physiography, land use, and generalized geology of the Central Valley, California.

Agriculture is the predominant land use in the Central Valley (fig. 1). About 57 percent of the total land area in the valley was agricultural in 2000, with about 1.76 million acres of irrigated crops in the Sacramento Valley and about 5.46 million irrigated acres in the San Joaquin Valley (McKinney and Anning, 2009). Major crop types include grains, hay, cotton, tomatoes, vegetables, citrus and other fruits, nuts, grapes, corn, and rice. Groundwater withdrawals from the Central Valley aquifer system were the second largest for a principal aquifer in the United States (after the High Plains aquifer), accounting for 13 percent of total withdrawals in 2000 (Maupin and Barber, 2005, p. 24). The withdrawals supplied about 10.7 million acre-ft (43 percent) of the water used for agriculture and public supply in the Central Valley in 2000 (McKinney and Anning, 2009, table 1) and are especially important in dry years because they supplement the variable surface-water supplies in the valley.

The population in the Central Valley more than doubled from about 2.7 million in 1970 (U.S. Census Bureau, 1995) to about 6.0 million people in 2005 (McKinney and Anning, 2009). Large urban areas include Sacramento, Fresno, Bakersfield, Stockton, and Modesto. Urban land use in the Central Valley has increased from 3 percent in 1961 to 7 percent in 2000 (Faunt and others, 2009, table C3) at the expense of both undeveloped and agricultural lands. Nearly every city in the San Joaquin Valley uses groundwater as its main source for municipal and industrial supplies (Faunt, Belitz, and Hanson, 2009, p. 62).

The U.S. Geological Survey (USGS) has recently reported on the availability and use of groundwater in the Central Valley as part of its Ground-Water Resources Program (Faunt, Hanson, and Belitz, 2009). Information on the regional groundwater flow system compiled and developed as part of that study is described in this section of the report.

Water Development History

Water development in the Central Valley began in about 1790 with the diversion of surface water for irrigation (Williamson and others, 1989, p. D44). Early farming was concentrated close to the delta formed by the San Joaquin and Sacramento Rivers and in other areas where the water table was near the land surface throughout the year. Agriculture in the San Joaquin Valley increased in the late 1850s with the drainage and reclamation of river bottom lands and by 1900, an extensive system of canals and ditches had been built and much of the flow of the Kern River and the entire flow of the Kings River had been diverted to irrigate lands in the southern San Joaquin Valley (Nady and Larragueta, 1983). Because no large storage facilities were built along with these early diversions, the agricultural water supply, and therefore crop demand, was largely limited by the amount of summer base flow in streams. By 1910 nearly all of the available surface-water supply in the San Joaquin Valley had been diverted, leading to more extensive development of groundwater resources.

Groundwater was first used in the Central Valley in about 1880 in areas where artesian conditions were present and flowing wells could be drilled, particularly near the central part of the San Joaquin Valley and around the terminal lake basins. After 1900, the yields of flowing wells were reduced due to declining water levels, and it became necessary to install pumps in the wells to sustain flow rates. Around 1930, the development of an improved deep-well turbine pump and rural electrification enabled additional groundwater development for irrigation (Galloway and Riley, 1999). Years of pumping in the valley for irrigation has caused large declines in the water table, resulting in many wells going dry and thousands of acres of farmland taken out of production (Faunt, Belitz, and Hanson, 2009, ch. B, p. 60).

In 1935, as part of the Federal Central Valley Project, planning began to use water from the San Joaquin and Sacramento Rivers to irrigate about 12 million acres in the San Joaquin Valley (Faunt, Belitz, and Hanson, 2009, p. 60). The need to prevent groundwater overdraft in the Central Valley and for additional water to support population growth in southern California prompted construction of the State Water Project. These two projects resulted in the storage of most of the tributary streamflow behind dams for use throughout the year. Surface water was diverted for the Central Valley Project for irrigation and transported to the southern San Joaquin Valley through the Madera and Friant-Kern Canals beginning in the mid-1940s and the State Water Project delivered water to the west side of the valley through the Delta-Mendota Canal in 1951. The Central Valley relies on a combination of local and imported surface water and local groundwater. Generally, most farms near surface water distribution canals use surface water. When surface water is not available later in the growing season or during drought, groundwater is used.

Hydrogeology

The Central Valley is a large structural trough filled with sediment that is bounded by primarily granitic and metamorphic rocks in the Sierra Nevada that were probably uplifted between Late Jurassic and Late Cretaceous time on the east (Planert and Williams, 1995) and a complex assemblage of late Jurassic- to Quaternary-age marine and continental rocks in the Coast Ranges on the west (Gronberg and others, 1998, p. 5). The northeastern corner of the valley is at the southern end of the Cascade Range and contains material derived from volcanic rocks (Planert and Williams, 1995, p. B16). The east side of the Central Valley is underlain by a westward sloping surface of consolidated rocks that are the subsurface continuation of the Sierra Nevada to the east. The trough tilts to the south and has been filled with marine and continental deposits of Tertiary age and continental deposits of Quaternary age. The continental sediments consist mostly of sand and gravel interbedded and mixed with clay and silt deposited by streams and lakes. Depending on location, deposits of fine-grained materials—mostly clay and silt—make up as much as 50 percent of the thickness of the basin-fill sediments (Planert and Williams, 1995, p. B17).

Alluvial fans have formed on all sides of the Central Valley with coarse-grained material deposited close to the valley margins and finer grained detritus transported farther toward the valley axis. On the east side of the valley, shifting stream channels have created coalescing fans consisting of broad sheets of inter-fingering, wedge-shaped lenses of gravel, sand and finer sediment (Faunt, Hanson, and Belitz, 2009, p. 18).

The basin-fill deposits in the Sacramento and San Joaquin Valleys have somewhat different depositional environments and textural compositions. A three-dimensional model of the percentage of sediments with coarse-grained texture in the Central Valley was developed by Faunt, Hanson, and Belitz, (2009, p. 2) from information on drillers' logs. The model shows significant heterogeneity in the texture of the sediments, although sediments in the Sacramento Valley are generally finer grained than in the San Joaquin Valley (Faunt, Hanson, and Belitz, 2009, figs. A12 and A14). Fine-grained sediments likely associated with nearby volcanic activity, relatively low energy drainage basins, and the lack of glacially derived deposits are interbedded with coarse-grained alluvial sediments in and near river channels, flood plains, and alluvial fans in the Sacramento Valley. No extensive layers of fine-grained sediments have been found in the Sacramento Valley (Faunt, Hanson, and Belitz, 2009, p. 20).

Areas of coarse-grained sediments are more widespread in the San Joaquin Valley, especially on the east side, and occur along the major rivers. Alluvial fans in the southern San Joaquin Valley are derived from glaciated parts of the Sierra Nevada and are much coarser grained than the alluvial fans to the north (Faunt, Hanson, and Belitz, 2009, p. 2). Generally thin, discontinuous lenses of fine-grained sediments (clay, sandy clay, sandy silt, and silt) are distributed throughout the San Joaquin Valley. The shales and marine deposits of the Coast Ranges generally yield finer grained sediments than the crystalline rocks of the Sierra Nevada and contribute to the sediments of the western San Joaquin Valley being finer grained overall than the eastern part. Alluvium derived from the Coast Range and the Sierra Nevada interfinger near the surface at the valley bottom. The large percentage of fine-grained sediments in the western San Joaquin Valley impedes the downward movement of groundwater and may contribute to agricultural drainage problems and to land subsidence in the area (Faunt, Hanson, and Belitz, 2009, p. 40).

The areally extensive lake-deposited Corcoran Clay of Pleistocene age underlies as much as 6,600 mi^2 of the San Joaquin Valley, extending to near the valley's western margin (Page, 1986; Faunt, Hanson, and Belitz, 2009, p. 21) (fig. 1). An analysis of well logs by Burow and others (2004, p. 29) indicates that the eastern extent of the Corcoran Clay lies approximately parallel to the axis of the valley and thins eastward or was eroded by the rivers draining the Sierra Nevada in the Modesto area. The top of the Corcoran Clay is up to 900 ft deep and the clay is as much as 200 ft thick beneath the Tulare Lake bed (Davis and others, 1959; Page, 1986).

Conceptual Understanding of the Groundwater Flow System

The main source of groundwater in the valley is the upper 1,000 ft of basin-fill deposits (Page, 1986). Granitic, volcanic, and metamorphic rocks that crop out and underlie the eastern part of the valley form an almost impermeable boundary for the basin-fill groundwater system. Little water flows through the extensive deposits of consolidated marine and mixed marine and continental rocks that overlie the crystalline rocks and bound the western part of the valley because of low permeability. Most of the freshwater (water with less than 1,000 mg/L of dissolved solids) is contained in continental deposits in the Sacramento Valley, where the depth to the base of freshwater is as much as 2,500 ft (Planert and Williams, 1995 p. B20). In the San Joaquin Valley, most of the freshwater is within continental deposits, but also is in marine rocks on the southeast side of the valley. The sediments in the San Joaquin Valley saturated with freshwater range in thickness from 100 to more than 4,000 ft. Saline water (water with a minimum dissolved-solids concentration of 2,000 mg/L) occurs at depth throughout the Central Valley, usually as connate water in marine sediments and rocks.

The general conceptual model for groundwater flow in the Central Valley is that of a heterogeneous aquifer system comprising confining units and unconfined, semiconfined, and confined aquifers (Williamson and others, 1989, p. D14; Faunt, Hanson, and Belitz, 2009, p. 20). Alluvial sediments transported from the surrounding Sierra Nevada and Coast Ranges make up the aquifer system. Unconfined (water table) or semiconfined conditions occur in shallower deposits and along the margins of the valley. The aquifer system becomes confined in most areas within a few hundred feet of land surface because of numerous overlapping lenses of fine-grained sediments (fig. 2). Generally, these lenses are discontinuous and are not vertically extensive or laterally continuous. An exception is the Corcoran Clay that separates the basin-fill deposits over a large area in the central, western, and southern parts of the San Joaquin Valley into an upper unconfined to semiconfined zone and a lower confined zone (Williamson and others, 1989, p. D16; Burow and others, 2004) (fig. 2). The drilling of thousands of large-diameter irrigation wells through and perforated above and below the Corcoran Clay has connected the upper and lower zones, resulting in a substantial increase in downward leakage (Bertoldi and others, 1991).

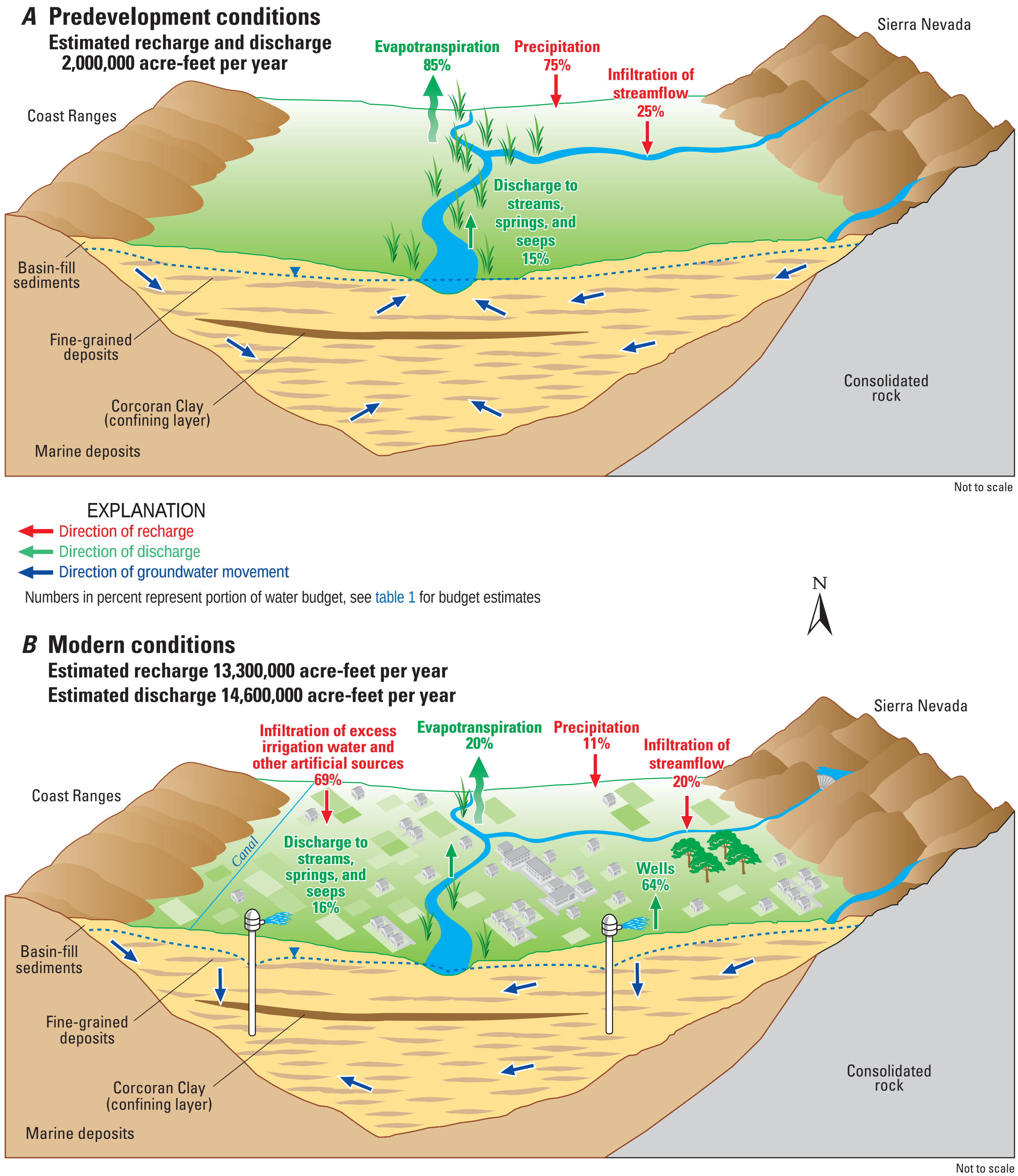


Figure 2. Generalized diagrams for the Central Valley, California, showing the basin-fill deposits and components of the groundwater system under (*A*) predevelopment and (*B*) modern conditions.

The considerable variability in hydraulic properties, both laterally and vertically, within the Central Valley aquifer system reflects the various depositional environments of the sediments. The water-transmitting properties of aquifer sediments are functions of lithology and differ according to grain size and the degree of sorting of the sediments. Hydraulic conductivity values used in a recent numerical groundwater flow model of the Central Valley aquifer system were assumed to be correlated to sediment texture, which was determined from the fraction of coarse-grained material recorded on multiple drillers' logs. Calibrated hydraulic conductivities ranged from 0.075 ft/d for fine-grained material to 670 ft/d for coarse-grained material in the Sacramento Valley and from 0.024 ft/d for fine-grained material to 330 ft/d for coarse-grained material in the San Joaquin Valley (Faunt and others, 2009, p. 156). For both valleys, the distributions of horizontal and vertical conductivities are the same as those for the sediment texture (Faunt, Hanson, and Belitz, 2009, fig. A12).

Groundwater Budget and Flow

Under predevelopment conditions, before surface-water diversions and irrigation began to affect the groundwater system in the Central Valley in about 1850 (Williamson and others, 1989, p. D32), recharge occurred naturally from the infiltration of precipitation on the valley floor and from stream losses in the upper parts of the alluvial fans, where the major streams enter the valley (fig. 2). Streams carrying runoff from the Sierra Nevada provided most of the water lost to the groundwater system. The volume of precipitation on the valley floor that infiltrates to the groundwater system is presumed to be significantly larger during wetter years (Faunt, Belitz, and Hanson, 2009, table B1). Estimates of selected components of the groundwater budget for subbasins within the Central Valley are presented where available by the California Department of Water Resources (2003).

Estimates of recharge and discharge to the Central Valley groundwater system under predevelopment conditions are presented in table 1. Because of a paucity of data before water development began, these values are considered to be rough estimates and represent recharge and discharge to both shallow, local aquifers and the deeper, more regionally extensive part of the groundwater system (Williamson and others, 1989, p. D38 and D57). Under predevelopment conditions, groundwater recharge was balanced by groundwater discharge, which occurred primarily through evapotranspiration and by leakage to streams in the bottom of the valley (fig. 2). Before water development substantially affected the aquifer, groundwater generally moved from recharge areas along the valley margins toward topographically low areas in the center of the valley and to the Sacramento or San Joaquin Rivers (fig. 3*A*). The vertical gradient was downward around the margins of the valley and upward in the center of the valley. The areas of natural discharge in the central part of the valley generally coincided with a large artesian area that was documented prior to 1900 (Hall, 1889; Mendenhall and others, 1916). The direction of groundwater flow in the southern San Joaquin Valley was toward Tulare Lake, an area of natural groundwater discharge that existed prior to water development in the area.

The natural patterns of groundwater movement and the rates of recharge and discharge throughout the Central Valley have been substantially altered by groundwater development and the diversion and redistribution of surface water for irrigation. These modifications have changed the amount and distribution of recharge to the aquifer system, which has affected the configuration of the water table in parts of the valley (fig.3*B*). Streams that naturally would have recharged the aquifer are now diverted to irrigate crops in other areas or the water is stored for seasonal release. Recharge from excess irrigation water and discharge from wells for irrigation and public supply, simulated to average about 9,200,000 and 9,300,000 acre-ft/yr from 1962 to 2003, respectively (Faunt, Belitz, and Hanson, 2009, table B2), are much larger than natural sources of recharge and discharge (table 1 and fig. 2). Groundwater withdrawals have lowered water levels, altered the direction and rates of groundwater flow, and have caused the land to subside in some areas (Williamson and others, 1989, p. D52).

Withdrawals from wells in the Central Valley averaged 11.5 million acre-ft/yr during the 1960s and 1970s, and during the drought of 1976–77, withdrawals increased to a high of about 15 million acre-ft (Bertoldi and others, 1991, p. A22). More surface water is available for irrigation during years with average or above average precipitation, resulting in a decrease in withdrawals from wells and a rise in groundwater levels. During drought years, less surface water is available for irrigation and wells are more heavily pumped, leading to water-level declines. Most of the approximately 100,000 high-capacity wells in the Central Valley are used for either irrigation or public supply (Bertoldi and others, 1991, p. A22). Well depths in the San Joaquin Valley range from about 100 to 3,500 ft, and the deepest wells are in the

Table 1. Estimated groundwater budget for the Central Valley basin-fill aquifer system, California, under predevelopment and modern conditions.

[All values are in acre-feet per year (acre-ft/yr). Estimates of groundwater recharge and discharge under predevelopment conditions are from Williamson and others (1989, fig. 19). Estimates for modern conditions are derived from averages listed for 1962–2003 by Faunt, Belitz, and Hanson (2009, table B2 and figure B1). The budgets are intended only to provide a basis for comparison of the overall magnitudes of recharge and discharge between predevelopment and modern conditions, and do not represent a rigorous analysis of individual recharge and discharge components]

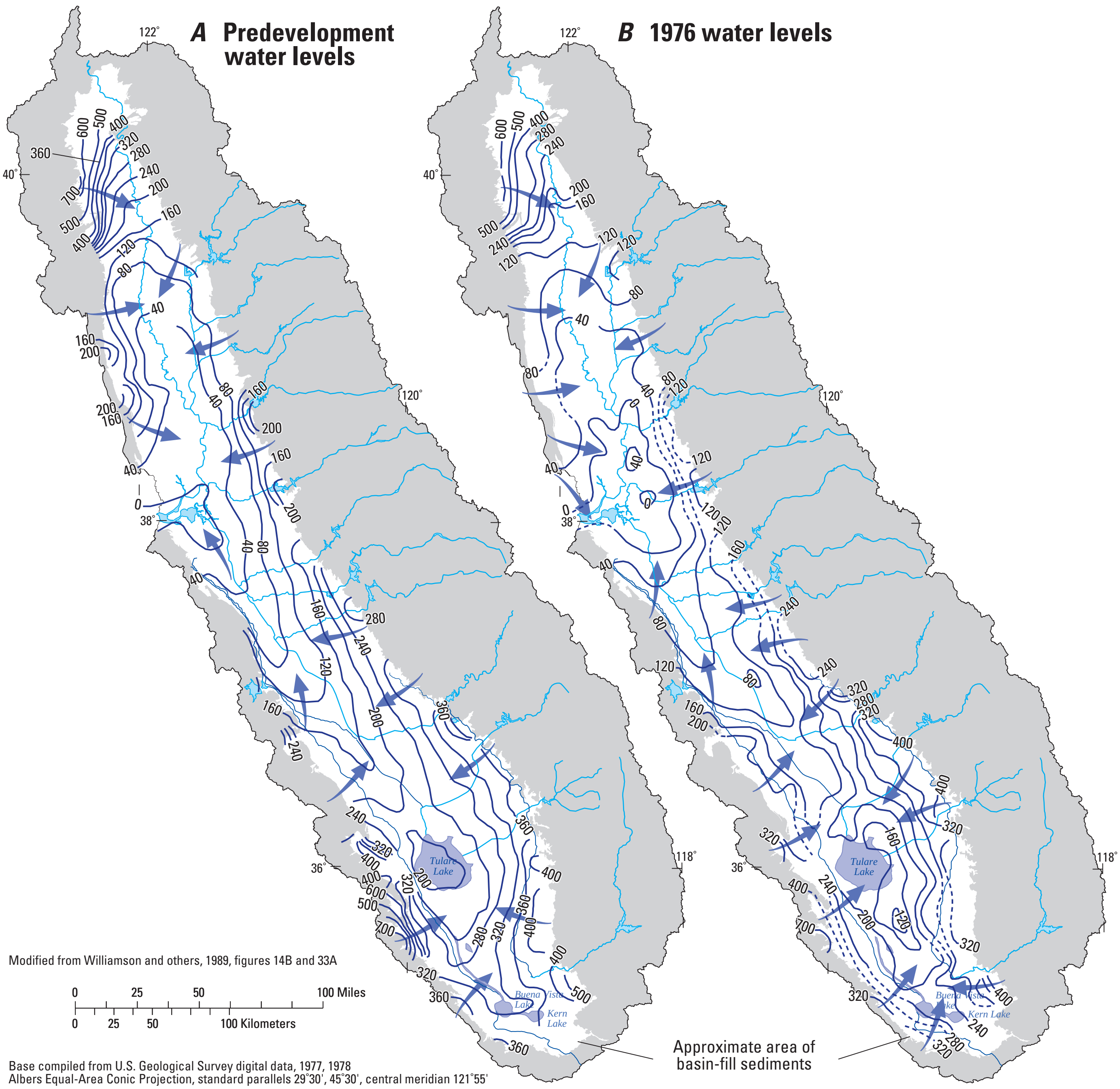
	Predevelopment conditions	Modern conditions	Change from predevelopment to modern conditions
Budget component		**Estimated recharge**	
Infiltration of precipitation on basin	1,500,000	[1]1,500,000	0
Infiltration of streamflow	500,000	2,600,000	2,100,000
Infiltration of excess irrigation water and other artificial sources	0	[1]9,200,000	9,200,000
Total recharge	**2,000,000**	**13,300,000**	**11,300,000**
Budget component		**Estimated discharge**	
Evapotranspiration	1,700,000	3,000,000	1,300,000
Discharge to streams, springs, and seeps	300,000	[2]2,300,000	2,000,000
Well withdrawals	0	9,300,000	9,300,000
Total discharge	**2,000,000**	**14,600,000**	**12,600,000**
Change in storage (total recharge minus total discharge)	0	-1,300,000	-1,300,000

[1] The simulated average recharge for 1962–2003 from landscape processes (includes infiltration of precipitation and excess irrigation water) was 10,700,000 acre-ft/yr (Faunt, Belitz, and Hanson, 2009, table B2). To fit the components in this table, recharge from the infiltration of precipitation on the basin was assumed to be the same as under predevelopment conditions (Williamson and others, 1989) and the remainder was assumed to be from excess irrigation water and from other artificial sources.

[2] Includes a simulated average discharge of 100,000 acre-ft/yr to the San Joaquin River Delta (Faunt, Belitz, and Hanson, 2009, fig. B1 and table B2).

west-central and south-central parts of the valley. Many of the wells are constructed with long perforated or screened intervals that connect several water-bearing layers and thus increase the vertical hydraulic connection through the aquifer system (Bertoldi and others, 1991, p. A23). Public-supply wells typically have long intervals open to the deeper part of the aquifer system. Vertical flow between permeable layers, either upward or downward, can be substantial in many unpumped and unsealed abandoned wells.

Recharge from excess irrigation water and discharge from wells for irrigation and public supply have increased the amount of water flowing vertically in the aquifer system from that under predevelopment conditions. Under modern conditions with water development, the combination of increased recharge to the water table and increased pumping from the lower confined zone has reversed the direction of the hydraulic gradient from upward to downward in the center of the valley (Williamson and others, 1989). In addition, groundwater moving along a lateral flow path may be extracted by wells and reapplied at the surface multiple times before reaching the natural discharge area in the valley bottom (Phillips and others, 2007, p. 4-7) (fig. 4). Under modern conditions in some areas, groundwater flows beneath the river toward pumping centers on the west side of the valley rather than discharging to the river (Bertoldi and others, 1991, p. A21).



EXPLANATION

Predevelopment water levels—Water-level altitude, in feet. Contour interval is variable. Datum is National Geodetic Vertical Datum of 1929

Study area boundary

Direction of water flow

EXPLANATION

Spring 1976 water levels—Water-level altitude, in feet. Contour interval is variable, dashed where approximately located. Datum is National Geodetic Vertical Datum of 1929

Study area boundary

Direction of water flow

Figure 3. Groundwater levels in the unconfined part of the aquifer system in the Central Valley, California (*A*) estimated for predevelopment conditions, and (*B*) in 1976.

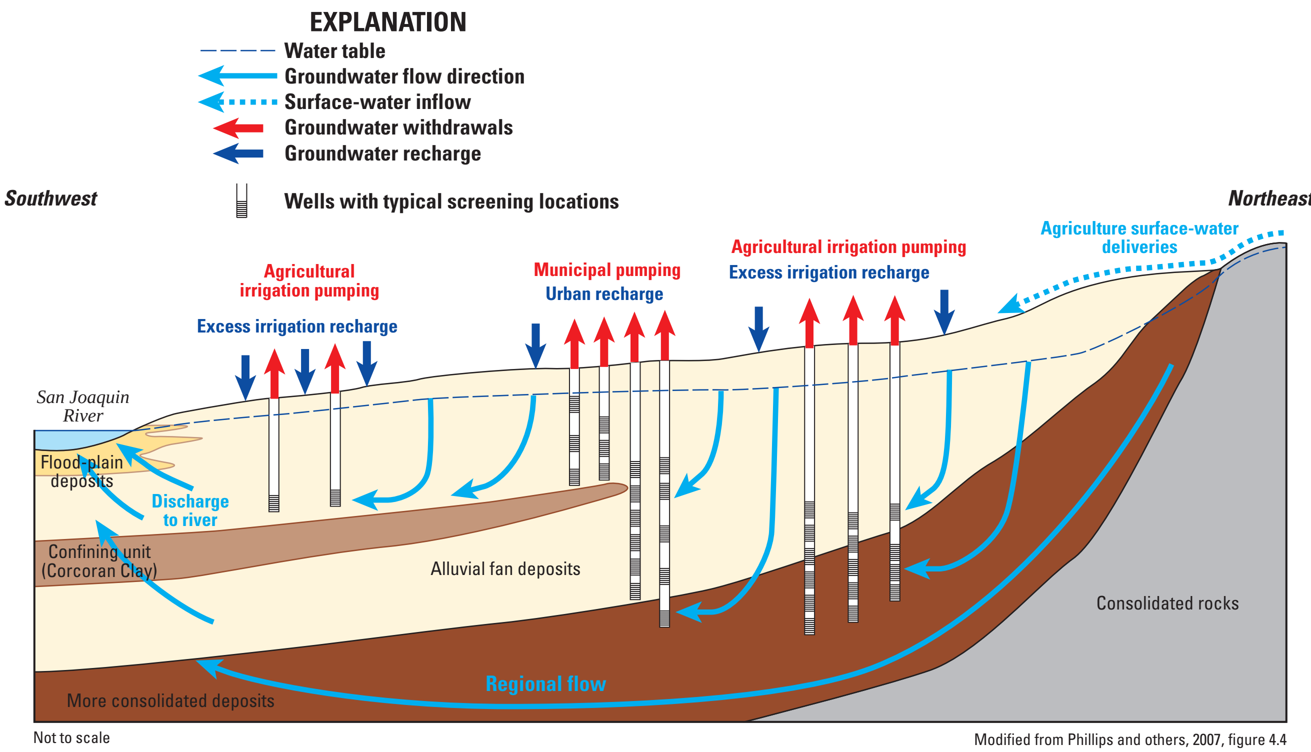
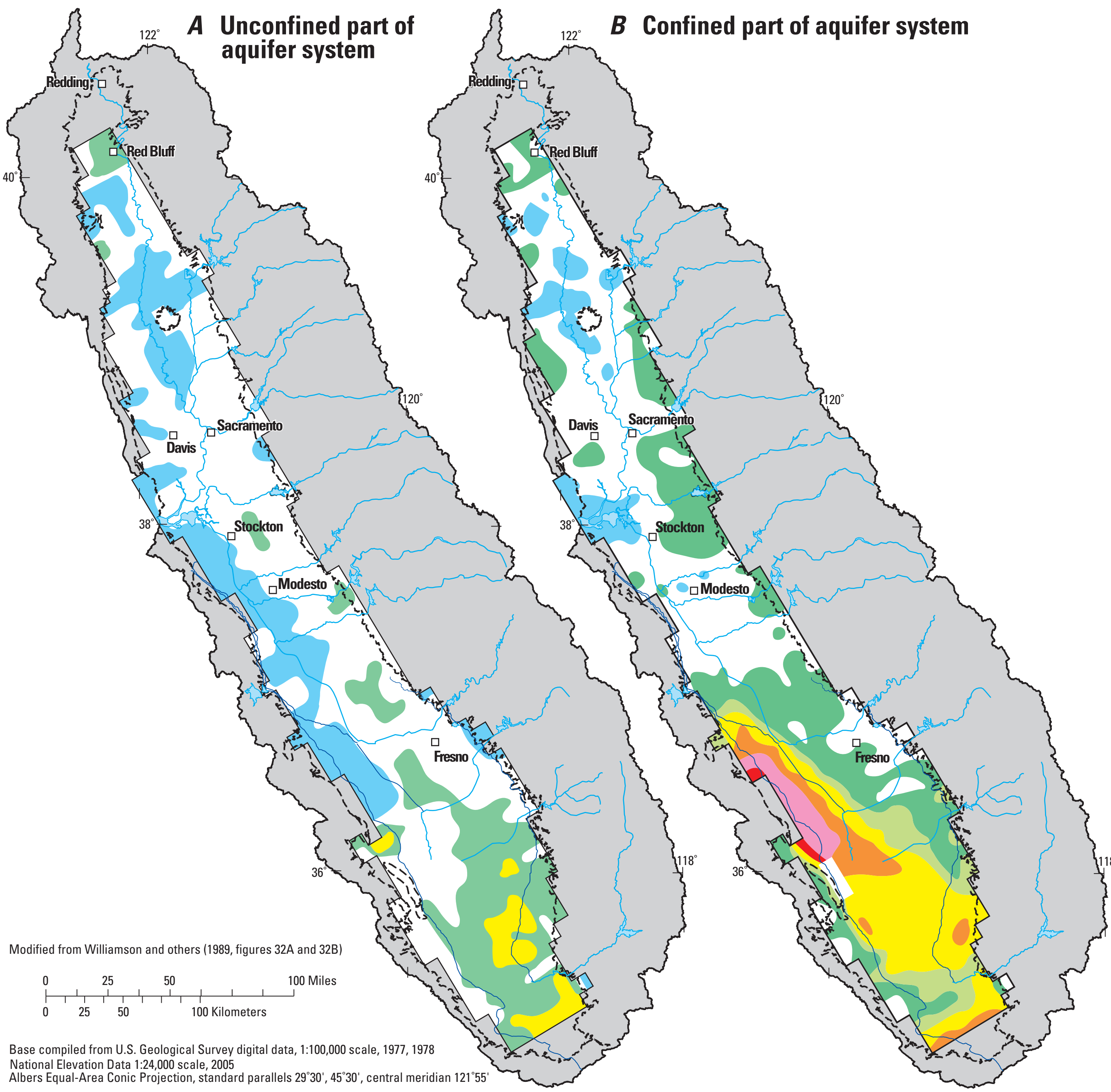


Figure 4. Conceptual diagram showing effects of pumping and irrigation on vertical groundwater flow, and natural discharge areas near Modesto, California.

From predevelopment conditions to 1961, water levels generally declined less than 100 ft in unconfined parts of the aquifer system and more than 100 ft in the deeper confined parts of the system in the western and southern San Joaquin Valley (Williamson and others, 1989, figs. 32A and 32B) (figs. *5A* and *5B*). Water levels have dropped more than 300 ft in some westside areas. Pumping in the western and southern San Joaquin Valley was reduced in 1967 when delivery of surface water through the California Aqueduct to farms in the area began. Increased surface-water delivery and decreased groundwater withdrawals caused water levels to rise in both the upper and lower zones in much of the area (Faunt, Belitz, and Hanson, 2009, p. 97). The water table also rose in much of the Sacramento Valley due to the recharge of excess irrigation water.

The decrease in groundwater stored in the Central Valley from predevelopment conditions to 1977 was calculated with a numerical model to be about 60 million acre-ft (Williamson and others, 1989, p. D95). About 1,400,000 acre-ft/yr less groundwater was simulated in storage using average annual conditions from 1962 to 2003 (Faunt, Belitz, and Hanson, 2009, p. 70), about the difference between estimated recharge and discharge for modern conditions listed in table 1. This depletion of water in storage is made up of three components: long-term decline of the water table; inelastic compaction of the aquifer (permanent reduction of pore space resulting in land subsidence); and elastic storage (compression of sediments and expansion of water). Although a large amount, this long-term decrease in aquifer storage is only a small part of the more than 800 million acre-ft of freshwater estimated to be stored in the upper 1,000 ft of sediments in the Central Valley (Williamson and others, 1989, p. D96).

The area affected by land subsidence includes much of the southern San Joaquin Valley, smaller areas in the Sacramento Valley, and in the delta area for the Sacramento and San Joaquin Rivers. Large groundwater withdrawals and associated water-level declines, mainly in deeper parts of the aquifer system during the 1950s and 1960s, caused about 75 percent of the total volume of land subsidence in the San Joaquin Valley. In 1970, subsidence in excess of 1 ft had affected more than 5,200 mi^2 of irrigable land in the San Joaquin Valley (Poland and others, 1975). The maximum subsidence was more than 28 ft near Mendota, about 30 mi west of Fresno in the bottom of the valley. Water levels in deeper parts of the aquifer system recovered as much as 200 ft in the 6 years from 1967 to 1974 (Ireland and others, 1984) and subsidence slowed or stopped over much of the affected area. Subsidence is likely to resume in the future if groundwater withdrawals cause water levels to drop below the previous low levels.



EXPLANATION

Estimated change in water levels, predevelopment to 1961, in the unconfined part of the aquifer system

- Rise from 0 to 40 feet
- Decline from 0 to 40 feet
- Decline from 41 to 100 feet
- Decline of more than 100 feet
- **Approximate boundary of basin-fill sediments**
- **Study area boundary**

EXPLANATION

Estimated change in water levels, predevelopment to 1961, in the confined part of the aquifer system

- Rise from 0 to 10 feet
- Decline from 0 to 40 feet
- Decline from 41 to 80 feet
- Decline from 81 to 120 feet
- Decline from 121 to 200 feet
- Decline from 201 to 300 feet
- Decline from 301 to 400 feet
- Decline of more than 400 feet
- **Approximate boundary of basin-fill sediments**
- **Study area boundary**

Figure 5. Changes in groundwater levels in the Central Valley, California, from predevelopment conditions to spring 1961 in the (*A*) unconfined part of the aquifer system and (*B*) confined part of the aquifer system.

Effects of Natural and Human Factors on Groundwater Quality

Many factors influence the quality of groundwater in the Central Valley aquifer system, but the predominant factors are the bedrock geology and chemistry of soils derived from bedrock, land use, and water use. Activities associated with agricultural land and water use have affected groundwater quality across the Central Valley. The infiltration of water pumped from deeper parts of the aquifer system and surface water applied to fields for irrigation has resulted in recharge to the shallow unconfined part of the aquifer system with water that has been exposed to agricultural chemicals and natural salts concentrated by evapotranspiration. Excess irrigation water has become a major source of recharge to the Central Valley aquifer system and may contribute to elevated concentrations of nutrients, pesticides, volatile organic compounds (VOCs), major ions, and trace elements in groundwater.

Groundwater-quality data has been collected in the Central Valley as part of several local- and regional-scale studies by the USGS, including the San Joaquin Valley Drainage Program (Gilliom and others, 1989), the Regional Aquifer Systems Program (Bertoldi and others, 1991), and the National Water-Quality Assessment (NAWQA) Program. The California Ground-Water Ambient Monitoring and Assessment (GAMA) Program also is collecting data from parts of the Central Valley aquifer system that will be used to identify the natural and human factors affecting groundwater quality (Belitz and others, 2003; Kulongoski and Belitz, 2006). Groundwater used for public drinking-water supplies was sampled in the southern Sacramento Valley (Milby Dawson and others, 2008) and the northern San Joaquin Valley (Bennett and others, 2006) in 2005; the southeastern San Joaquin Valley (Burton and Belitz, 2008) in 2005–06; and the middle Sacramento Valley (Schmitt and others, 2008), the Kern County part of the southern San Joaquin Valley (Shelton and others, 2008), and the central part of the eastern San Joaquin Valley (Landon and Belitz, 2008) in 2006. Water-quality data collected by the USGS are stored in the National Water Information System (NWIS) database.

Studies made as part of the NAWQA Program on groundwater quality in parts of the Sacramento Valley (Domalgalski and others, 2000) and in the eastern San Joaquin Valley (Dubrovsky and others, 1998) have provided information on the factors affecting water quality in a portion of the Central Valley. The location of the NAWQA sampled wells are shown on figure 6. Water was sampled from 61 wells (59 domestic wells and 2 monitoring wells) completed in alluvial fan deposits along the east side of the valleys as part of regional aquifer studies to assess the concentration and distribution of major chemical constituents, nutrients, pesticides, VOCs, trace elements, and radon. The study in the southeastern Sacramento Valley is described by Dawson (2001a) and the study in the eastern San Joaquin Valley is described by Burow and others (1998a). Because domestic wells are typically screened in the upper part of the aquifer system, this dataset generally represents the water quality of the unconfined part of the aquifer system in the eastern San Joaquin Valley and southeastern Sacramento Valley.

Shallow groundwater was sampled from 28 monitoring wells completed near the water table beneath or near rice fields in the central Sacramento Valley (Dawson, 2001b) and from 19 monitoring wells beneath recently urbanized areas of Sacramento (Shelton, 2005) to determine the water chemistry of recently recharged groundwater and the effects of these land uses on water quality. Three land-use studies were done in the eastern San Joaquin Valley: in agricultural areas dominated by vineyards, in almond orchards, and in areas in which corn, alfalfa, and vegetables were grown in rotation (Burow and others, 1998b). Combined, these three crop groups account for 47 percent of the agricultural land in the eastern San Joaquin Valley.

A local-scale network of 20 monitoring wells near Fresno along an approximately horizontal groundwater flow path was designed and sampled by NAWQA investigators to characterize the spatial and temporal distribution of water quality in relation to groundwater flow in a vineyard land-use setting (Burow and others, 1999). A network of 23 monitoring wells in the zone of contribution to a public-supply well in Modesto was designed and sampled as part of the NAWQA Transport of Anthropogenic and Natural Contaminants (TANC) to public-supply wells topical study (Jurgens and others, 2008).

General Water-Quality Characteristics and Natural Factors

The quality of groundwater in the Central Valley is influenced by the surface water that enters the valley from the surrounding mountains. Runoff and snowmelt from the Sierra Nevada have low concentrations (less than 200 mg/L) of dissolved solids (fig. 7) because of the low solubility of the quartz and feldspar minerals that comprise the granitic bedrock and sediment derived from this rock. In contrast, the rocks and sediments of the Coast Ranges in the western part of the valley contain highly soluble minerals. Of particular importance are marine sedimentary formations with soluble calcium, sodium, and magnesium sulfates, and elevated concentrations of various nitrogen-containing minerals and trace elements (Gronberg and others, 1998, p. 27). Precipitation on the Coast Ranges dissolves these constituents and the resulting runoff has elevated concentrations of dissolved solids and other minerals. Chemical constituents also may be concentrated in the soil and in shallow groundwater by evapotranspiration.

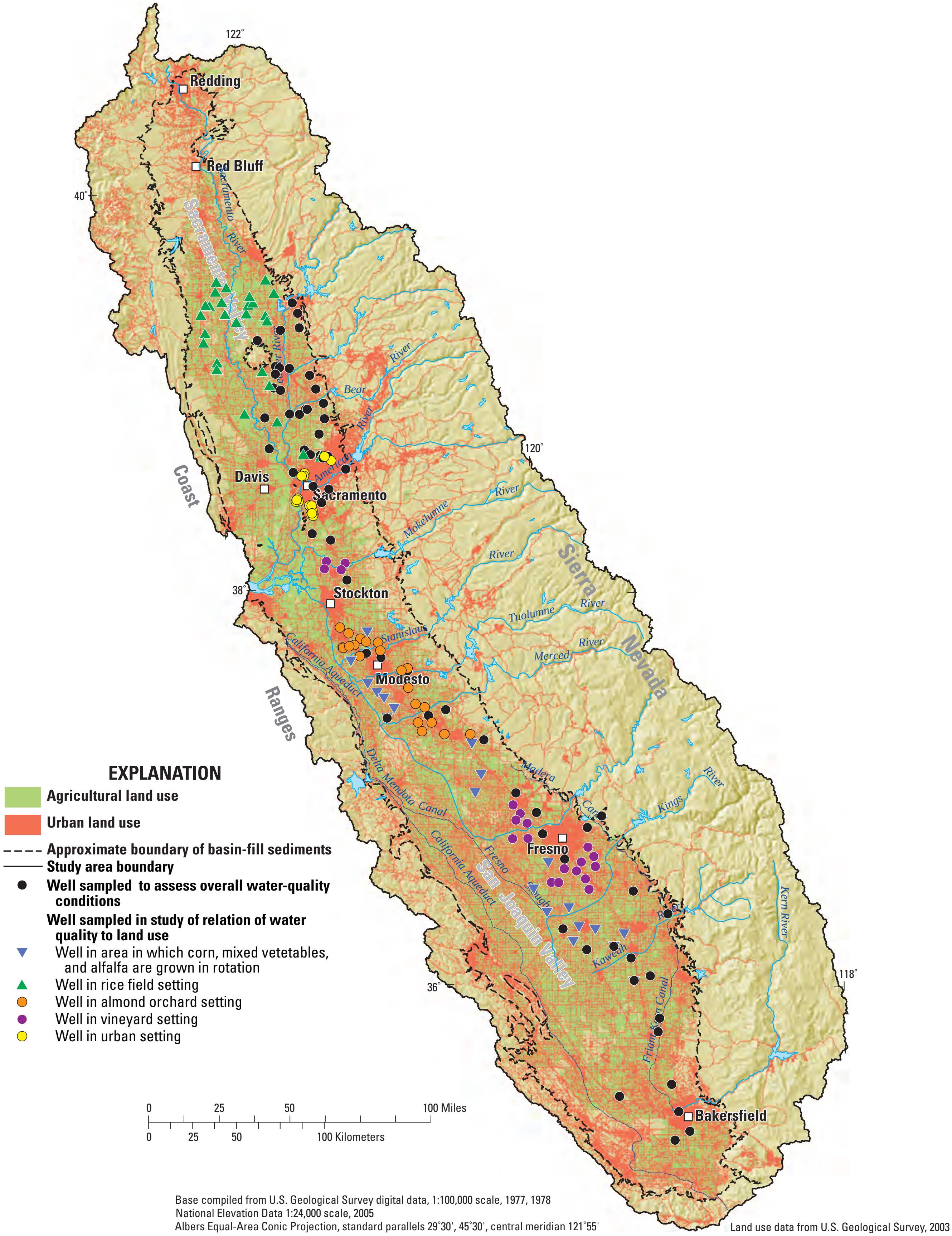


Figure 6. Location of wells sampled in the Central Valley, California, by the NAWQA Program, 1993–98.

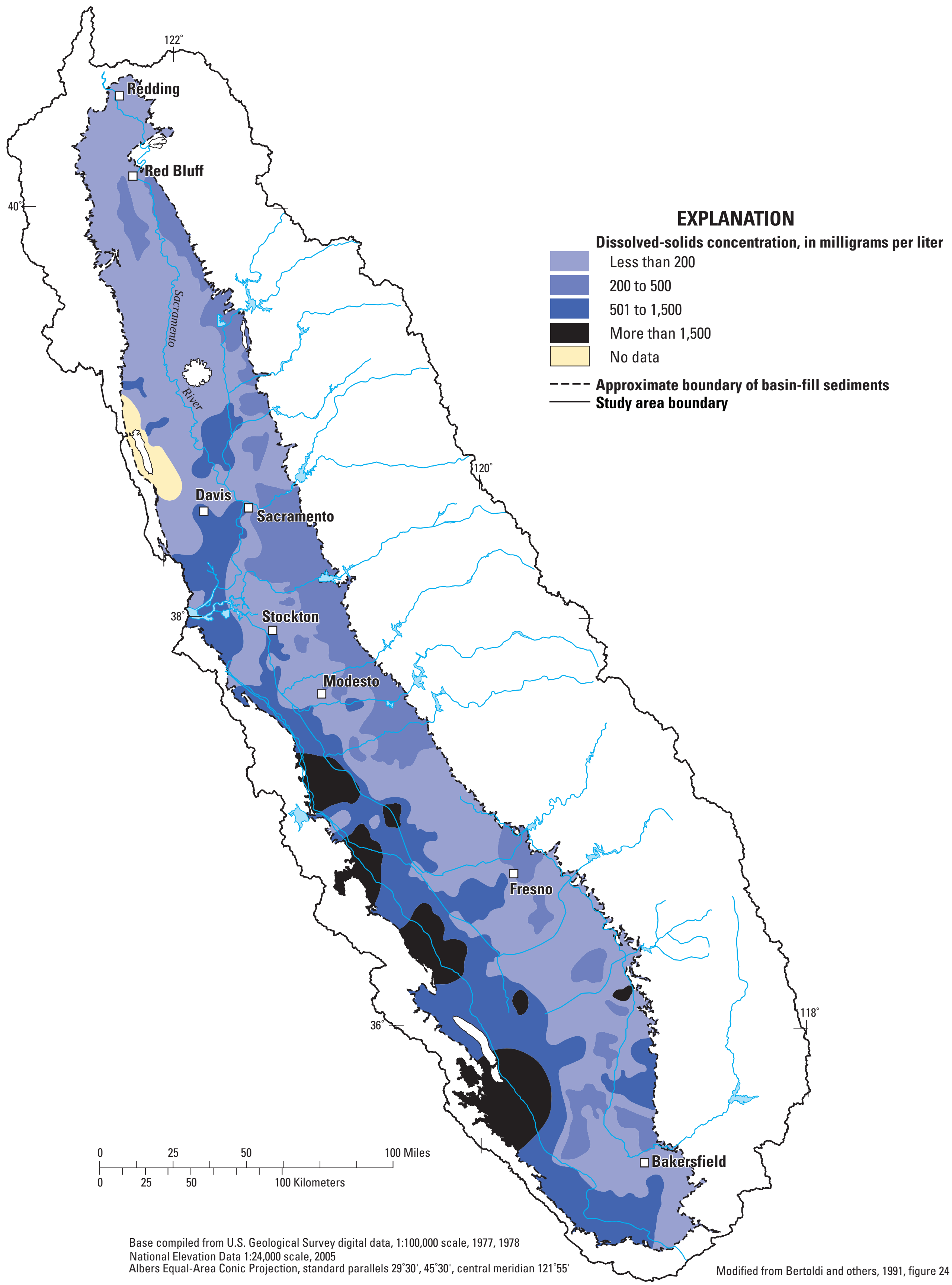


Figure 7. Distribution of dissolved-solids concentrations in groundwater of the Central Valley, California.

Groundwater chemistry varies spatially in the Central Valley. Calcium is a predominant cation and bicarbonate the predominant anion in groundwater in the northern Sacramento Valley and the eastern Sacramento and San Joaquin Valleys. Groundwater on the west side generally has higher concentrations of sulfate, chloride, and dissolved solids than groundwater on the east side (Bertoldi and others, 1991, fig. 25). Groundwater in the center of the valley is a combination of water from the east and west sides, is generally more geochemically reduced, and contains higher concentrations of dissolved solids than groundwater on the east side (Davis and others, 1959). The higher concentrations of dissolved solids result from cation exchange processes as the water moves through the sediments and from evaporative concentration in the discharge area of the aquifer system. Concentrations tend to increase from the north to south along the axis of the Sacramento Valley, but generally do not exceed 500 mg/L. On the east side of the San Joaquin Valley, concentrations of dissolved solids generally do not exceed 500 mg/L and tend to increase to the west. In general, dissolved-solids concentrations increase with depth in the Central Valley aquifer system. Therefore, typically deeper wells in the western and southern parts of the San Joaquin Valley are likely to produce water with higher concentrations of dissolved solids than the typically shallower wells in the Sacramento Valley and the eastern San Joaquin Valley (Planert and Williams, 1995, p. B20).

Groundwater in the coarse-grained alluvial-fan deposits on the east side of the valley is generally oxic (contains dissolved oxygen), while in the finer grained basin and lake deposits near the axis of the valley it is usually anoxic (contains less than 0.5 mg/L of dissolved oxygen according to a framework described by McMahon and Chapelle (2008)). Geochemically reducing conditions commonly occur in discharge areas where long flow paths terminate and residence time and content of organic matter increase (Gronberg and others, 1998, p. 29). In the northwestern San Joaquin Valley, sediments derived from the Sierra Nevada are more reduced than interbedded sediments from the Coast Range (Dubrovsky, and others, 1991, p. 24). In the NAWQA land-use study, median concentrations of dissolved oxygen in samples from shallow wells in rice field areas in the Sacramento Valley and in the corn-alfalfa-vegetable rotation fields in the San Joaquin Valley were 0.36 and 1.5 mg/L, respectively. Wells sampled as part of these studies are in the center of the valleys or near where the alluvial fans and basin bottom deposits meet, where sediments are generally fine-grained and have a relatively high organic content. Wells sampled for the other NAWQA studies are generally completed in the upper parts of the alluvial fan deposits and had median concentrations of dissolved oxygen greater than 3 mg/L. The redox environment in the aquifer strongly influences the potential for degradation or accumulation of redox sensitive constituents, such as selenium, uranium, arsenic, nitrate, and some pesticides and VOCs. The median pH for groundwater sampled as part of the Central Valley NAWQA studies ranged from 7.1 to 7.4 standard units.

Soils in the western San Joaquin Valley are derived primarily from the marine rocks that form the western boundary of the aquifer system and contain relatively large amounts of selenium. When these soils are irrigated, naturally occurring minerals containing selenium are dissolved and mobilized into the shallow groundwater (Gilliom and others, 1989, p. 1). Excess irrigation water applied to remove salts from the soil, thus preventing salt buildup, leaches selenium from the soil and the marine rocks and transports it to shallow groundwater or to surface drains. Generally, concentrations of selenium in groundwater are highest in areas of the western San Joaquin Valley where soil selenium and groundwater salinity are high, where the water table has been near the land surface and evaporative concentration has occurred, and where groundwater is oxic (Dubrovsky and others, 1993, p 543). Water that contains dissolved selenium concentrations of 1,400 ug/L is present in some of the regional surface drains and concentrations as high as 3,100 ug/L have been detected in shallow groundwater in the western San Joaquin Valley, west of the San Joaquin River flood plain (Planert and Williams, 1995, p. B20). The elevated concentrations of selenium in the western part of the valley are known only to be in the shallow groundwater and not in the deeper parts of the aquifer system from which most wells that supply municipalities obtain water (Planert and Williams, 1995, p. B20).

Arsenic is a minor constituent of minerals within the granitic rocks of the Sierra Nevada and the marine rocks of the Coast Ranges. Although arsenic concentrations in soil are consistently higher in the western and southernmost parts of the San Joaquin Valley than in the eastern part (Belitz and others, 2003, p. 58), analysis of water-quality data from public-supply wells retrieved from the California Department of Health Sevices database and from a variety of well types retrieved from the NWIS database indicate that high concentrations of arsenic in soil are not sufficient to cause high concentrations of arsenic in groundwater. The presence and concentration of arsenic in groundwater also is influenced by reducing conditions and high pH (greater than 8) (Belitz and others, 2003 p. 60). Groundwater sampled from the central San Joaquin Valley that historically was the discharge zone for regional groundwater flow generally is reduced and has higher concentrations of arsenic than groundwater sampled from the eastern and western alluvial fan areas. Arsenic concentrations in the upper semiconfined zone in the northwestern San Joaquin Valley were significantly higher in reduced groundwater from the Sierra Nevada sediments than in oxic groundwater from the Coast Range sediments (Dubrovsky, and others, 1991, p. 25). Some of the high concentrations of arsenic in shallow wells in the Tulare Lake Bed in the southern San Joaquin Valley have been attributed to evaporative concentration (Fujii and Swain, 1995), a process that also affects dissolved solids, selenium, and other trace elements in the hydrologically closed basin.

Hull (1984) proposed that reducing conditions in the fine-grained sediments in basin areas with flood-plain deposits are a major influence on groundwater chemistry of the Sacramento Valley. Concentrations of arsenic exceeded the drinking-water standard of 10 µg/L (U.S. Environmental Protection Agency, 2008) in 7 (23 percent) of the 31 domestic and monitoring wells in the southeastern Sacramento Valley and in 10 (11 percent) of the 88 domestic wells in the eastern San Joaquin Valley sampled as part of the NAWQA Program. A significant inverse correlation was found between arsenic and dissolved oxygen concentrations in water sampled from the Sacramento Valley, suggesting that the presence and concentration of arsenic is related to the redox condition of the groundwater (Dawson, 2001a, p. 17). At a local scale, concentrations of arsenic in water sampled from the TANC network of wells in the Modesto area ranged from 2.3 to 15.9 µg/L (Jurgens and others, 2008, p. 38). Water sampled from the public-supply well had a concentration of 6.2 µg/L. Some reductive dissolution of iron oxyhydroxides and the subsequent release of adsorbed arsenic is thought to be responsible for the elevated arsenic concentrations in these water samples.

Naturally-occurring uranium is commonly adsorbed to aquifer sediments derived from the Sierra Nevada. The median concentration of uranium in water from wells sampled as part of the local-scale TANC study in Modesto was 10 µg/L (Jurgens and others, 2008, p. 39), and water from two monitoring wells had concentrations of uranium above the drinking-water standard of 30 µg/L (U.S. Environmental Protection Agency, 2008). Large rates of recharge in the agricultural area surrounding the city of Modesto and large withdrawals from wells within the city have caused oxygenated, high-alkalinity groundwater near the water table to move downward and laterally toward the wells. The continued downward migration of oxygenated, high-alkalinity groundwater is likely to mobilize uranium adsorbed to deeper sediments and to increase concentrations of uranium in that part of the aquifer (Jurgens and others, 2008, p. 51). Groundwater in other areas on the east side of the Central Valley also is susceptible to increasing concentrations of uranium where concentrations of bicarbonate are elevated and water is being pumped from deeper parts of the aquifer.

Potential Effects of Human Factors

The agricultural development in the Central Valley discussed in previous parts of this section has affected groundwater quality by adding millions of pounds of nitrate and pesticides to the land surface and modifying groundwater recharge so that these compounds can be more easily transported to the subsurface. Excess irrigation water can move chemicals applied at the land surface to the upper part of the aquifer system and eventually to the deeper part that is used for public supply. Groundwater in agricultural areas also can become excessively saline and damaging to crops because evaporation of sprayed irrigation water and evapotranspiration of soil moisture and shallow groundwater leaves behind dissolved salts. Shallow irrigation wells can accelerate the process by recirculating the saline shallow groundwater. Irrigation return-water drainage systems have been used to remove some of the saline shallow groundwater (Planert and Williams, 1995, p. B20).

Elevated concentrations of nitrate have been measured in shallow groundwater in areas of the Sacramento Valley (Planert and Williams, 1995, fig. 101) and sporadically in the San Joaquin Valley. Fogelman (1983) suggested that contamination from the land surface by leaching of applied nitrate fertilizers, urban waste-treatment facilities, and septic systems are the probable sources of nitrate in the groundwater in the Sacramento Valley. The median concentration of nitrate in water sampled in 1996 as part of the NAWQA regional aquifer study in the southeastern Sacramento Valley was 1.3 mg/L. The depth to the top of the openings in the well casings ranged from 29 to 215 ft below land surface and the mostly domestic wells were completed in basin-fill deposits with no continuous confining layers or other distinct internal boundaries that might impede the movement of groundwater. Water sampled from 8 (26 percent) of the 31 wells had nitrate concentrations greater than 3 mg/L, a level that may indicate an impact from human activities. Water sampled from 10 of the wells (32 percent) had dissolved oxygen concentrations less than or equal to 1 mg/L indicating anoxic conditions (Dawson, 2001a, p. 17). The median concentration of nitrate in water from these wells was 0.09 mg/L, and all of the wells were near the center of the Sacramento Valley in areas with finer grained deposits. The median concentration of nitrate in shallow groundwater sampled from monitoring wells in rice farming areas was 0.59 mg/L and in urban areas was 2.4 mg/L. The drinking-water standard for nitrate of 10 mg/L (U.S. Environmental Protection Agency, 2008) was exceeded in only one of the NAWQA samples from the Sacramento Valley.

Concentrations of nitrate were greater than 10 mg/L in 24 percent (21 of 88) of the domestic wells sampled in 1993–95 and in 29 percent (30 of 102) of the domestic wells sampled in 2001–02 as part of the NAWQA regional aquifer and land-use studies in the eastern San Joaquin Valley. The median concentration of nitrate of 5.6 mg/L in samples collected from these wells in 1993–95 and 6.4 mg/L in samples collected in 2001–02 (Burow and others, 2008, table 2) indicates that groundwater is affected over a large part of the area because of the input of nitrate from human activities, most likely agricultural practices.

The amount of coarse-grained sediments (sand- or gravel-sized) in the subsurface is a major factor in the susceptibility of groundwater to nitrate contamination. Sediment texture influences the rates of infiltration and groundwater flow, which in turn controls how rapidly water at the surface, with high nitrate concentrations, can infiltrate the soil and move downward into the aquifer. The sediment texture in the almond orchard and vineyard settings sampled by NAWQA in the eastern San Joaquin Valley is generally coarse-grained, and in the corn, alfalfa, and vegetable setting it is generally fine-grained with abundant clay. These contrasts in sediment texture, along with the amount of nitrogen applied, help explain the range in nitrate concentrations in groundwater underlying the different land-use settings. Nitrate concentrations in groundwater were highest in the almond orchard setting where high susceptibility and large amounts of nitrogen fertilizer applied occurred together (median concentration was 10 mg/L); nitrate concentrations in groundwater were lowest in the vineyard setting, where the amount of nitrogen applied was relatively small, even though the aquifer's susceptibility to contamination was high (median concentration was 4.6 mg/L); and nitrate concentrations in groundwater were intermediate in the corn, alfalfa, and vegetable setting where the amount of nitrogen applied was large, but the susceptibility was low (median concentration was 6.2 mg/L) (Burow and others, 1998b, p. 20).

Nitrate concentrations exceeded the drinking-water standard in 1.8 percent (19 of 1,045) of public-supply wells sampled in the Sacramento Valley and in 4.3 percent (27 of 629) of public-supply wells sampled in the San Joaquin Valley from 1994–2000 as part of the California Department of Health Services Title 22 water-quality monitoring program (California Department of Water Resources, 2003, subbasin descriptions). Wells in the Central Valley used for public supply and irrigation are generally deeper than wells used for domestic supply, based on average well depths listed by subbasins (California Department of Water Resources, 2003, subbasin descriptions).

Mean groundwater ages estimated for water sampled from the NAWQA local-scale networks of monitoring wells near Fresno and Modesto indicate that groundwater age increases with depth below land surface (Burow and others, 2008, p. S253). Shallow groundwater (less than 30 ft below the water table) was generally less than 15 years old, and deeper groundwater (more than than 180 ft below the water table) was greater than 45 years old. The water table in these areas is typically about 30–50 ft below land surface. Nitrate concentrations in water sampled from these monitoring wells were generally highest near the water table, decreased with depth, and were higher in the agricultural setting than in the urban setting (Burow and others, 2008, fig. 3). The groundwater system in these areas is largely oxic; therefore denitrification is not expected to significantly reduce nitrate concentrations.

Concentrations of nitrate in both shallow and deep parts of the aquifer system in the eastern San Joaquin Valley have gradually increased during the last 50 years (Burow and others, 2008, p. S261). The amount of nitrogen fertilizer applied in the eastern San Joaquin Valley increased from 114 milliion pounds in 1950 to 745 million pounds in 1980, an increase of 554 percent (Dubrovsky and others, 1998, p. 17). Anthropogenic nitrogen inputs, and hence, elevated nitrate concentrations in groundwater are likely to continue into the future. Nitrate concentrations in deeper groundwater will likely continue to increase over time as the water moves downward from the water table, although concentrations will be influenced by increased mixing of water and dispersion of nitrate with depth.

Pesticides have been used intensively in the Central Valley for many years and are expected to be detected widely throughout the area. A study of pesticides in San Joaquin Valley groundwater found that most detections occurred on the east side of the valley (Domagalski and Dubrovsky, 1991). Factors found to affect pesticide detections include the generally more permeable coarse-grained sediments, a relatively shallow water table in many areas, and the use of water-soluble pesticides with long environmental half-lives. The fewer detections on the west side of the San Joaquin Valley are attributed to a much longer residence time in finer grained sediments of the unsaturated zone, which allows for degradation to occur (Domagalski, 1997). The most frequently detected pesticide was 1,2-dibromo-3-chloropropane (DBCP), a soil fumigant commonly used in orchards and vineyards in the San Joaquin Valley beginning in the 1950s. DBCP was detected in about 31 percent of 4,507 wells in the San Joaquin Valley sampled from 1971through 1988 (Domagalski, 1997). Agricultural use of DBCP was banned in California in 1977 in response to concern about its potential hazardous effects on human health.

Pesticides were detected in 61 of the 88 domestic wells (69 percent) sampled in the eastern San Joaquin Valley as part of the NAWQA Program during 1993–95, but concentrations of most pesticides were low, less than 0.1 µg/L. Only five pesticides were detected in more than 10 percent of the samples: simazine, DBCP, atrazine, deethylatrazine (a degradation product of atrazine), and diuron. The number of pesticide detections in groundwater in the eastern San Joaquin Valley was related to sediment texture, concentrations of dissolved oxygen, pesticide application rates, groundwater recharge rates, and groundwater residence times (Burow and others, 1998a). Concentrations of DBCP in water sampled from 18 of the 88 domestic wells (20 percent) in the eastern part of the valley during 1993–95 exceeded the U.S. Environmental Protection Agency's drinking-water standard of 0.2 µg/L (Burow and others, 1998a, p. 24 and 1998b, p. 25). The occurrence of this pesticide in groundwater near Bakersfield, Fresno, Modesto, and north of Merced and Stockton coincides with land-use patterns. DBCP was detected in 25 of the 50 domestic wells in the almond orchard and vineyard settings that were sampled in 2001–02, and concentrations in 32 percent of the samples exceeded the drinking-water standard (Burow and others, 2008, table 3). Local-scale studies indicate that DBCP detections and concentrations may increase in the deeper part of the aquifer system in the future because of the dominantly downward movement of groundwater and the lack of significant attenuation processes in the subsurface (Burow and others, 2007, p. 1004).

In the southeastern part of the Sacramento Valley, only simazine, deethylatrazine, and bentazon were detected in 10 percent or more of the NAWQA regional aquifer study samples, all at concentrations much less than drinking-water standards. The herbicide bentazon was applied on rice fields in the Sacramento Valley from 1978 until 1989, when its use was banned in California. It was detected in 20 of the 28 monitoring wells (71 percent) in the rice field setting sampled in 1997, all at concentrations much lower than the drinking-water standard (Dawson, 2001b, table 9). Rice cultivation requires that fields be flooded during the growing season, from May through September. The high detection frequency of bentazon almost 10 years since its last known use in the area suggests that it is easily transported to the water table and does not readily degrade (Domagalski and others, 2000, p. 23). Bentazon also was detected in 4 of the 24 (17 percent) domestic well samples collected in the southeastern Sacramento Valley, but was detected in only 1 of the 19 monitoring wells in the urban setting and was not detected in any of the groundwater samples collected for the NAWQA studies in the San Joaquin Valley. This confirms the association between this herbicide and the rice field setting.

Water from less than 1 percent of public-supply wells sampled from 1994–2000 in the Sacramento Valley (3 of 820 wells) had a pesticide concentration that exceeded its drinking-water standard compared to more than 8 percent (18 of 608 wells) in the San Joaquin Valley (California Department of Water Resources, 2003, subbasin descriptions). This distribution agrees with the general occurrence of pesticides detected by the NAWQA studies of shallower parts of the groundwater systems in the Sacramento and and San Joaquin Valleys.

Volatile organic compounds were infrequently detected in groundwater samples collected for the NAWQA studies in the Sacramento and San Joaquin Valleys, except for in the setting in Sacramento, where 16 of the 19 monitoring wells (84 percent) contained one or more VOCs (Shelton, 2005, table 4). Chloroform, a byproduct formed during chlorination of water for drinking purposes, was the most frequently detected VOC (16 samples) in the urban area. A likely source of chloroform in the shallow groundwater is the use of disinfected public-supply water to irrigate lawns and gardens. The presence of chloroform and tritium in water from the monitoring wells indicates a component of young water that was recharged after 1953 (Shelton, 2005, p. 28). See Section 1 of this report for a discussion of groundwater age and environmental tracers. The occurrence of VOCs in groundwater sampled from the urban Sacramento area, like those of pesticides and elevated concentrations of nitrate in other parts of the Central Valley, is generally related to the amount of coarse-grained sediments in the subsurface. Monitoring wells that penetrated finer grained sediments generally had no or only one detection of a VOC in the water sample, and pesticides and nitrate were typically not detected. This likely is a result of reducing conditions in the aquifer in discharge areas. Results of multivariate analysis of the data indicate that most of the detections of VOCs and pesticides, and elevated concentrations of nitrate in the urban area occurred in oxic groundwater that is found in coarser grained alluvial fan deposits (Shelton, 2005, p. 42).

Water from three percent of public-supply wells sampled from 1994–2000 in the Sacramento Valley (24 of 810 wells) and in the San Joaquin Valley (18 of 608 wells) had a VOC concentration that exceeded its drinking-water standard (California Department of Water Resources, 2003, subbasin descriptions). Generally, subbasins on the east side of the Central Valley had a higher percentage of concentrations that exceeded drinking-water standards than did subbasins on the west side.

Summary

The Central Valley aquifer system is contained within basin-fill deposits in the Central Valley of California. Agriculture is the predominant land use in the Central Valley, and groundwater withdrawals supplied about 10.7 million acre-ft (43 percent) of the water used for agriculture and public supply in 2000. Groundwater is especially important in dry years because it supplements the variable surface-water supplies in the valley.

Alluvial fans have formed on all sides of the Central Valley with coarse-grained material deposited closer to the valley margins and finer grained detritus transported farther toward the valley axis. Sediment in the Sacramento Valley is generally finer grained than in the San Joaquin Valley, but with no extensive layers of fine-grained sediments. The Corcoran Clay separates the basin-fill deposits into an upper unconfined to semiconfined zone and a lower confined zone in the central, western, and southern parts of the San Joaquin Valley. The conceptual model for groundwater flow in the Central Valley is that of a heterogeneous aquifer system comprised of confining units and unconfined, semiconfined, and confined aquifers. Unconfined (water table) or semiconfined conditions occur in shallower deposits and along the margins of the valley. The aquifer system becomes confined in most areas within a few hundred feet of land surface because of numerous overlapping lenses of fine-grained sediments.

Under predevelopment conditions, before surface-water diversions and irrigation began, recharge occurred naturally from the infiltration of precipitation on the valley floor and from stream losses in the upper parts of the alluvial fans where the major streams enter the valley. The natural patterns of groundwater movement and the rates of recharge and discharge have been substantially altered by groundwater development and the diversion and redistribution of surface water throughout the Central Valley for irrigation. Recharge from excess irrigation water and discharge from wells for irrigation and public supply are much larger than natural sources of recharge and discharge and have increased the amount of water flowing vertically in the aquifer system from that under predevelopment conditions. Groundwater withdrawals have lowered water levels and have caused the land to subside in some areas.

The predominant factors that influence the quality of groundwater in the Central Valley aquifer system are the bedrock geology and chemistry of soils derived from bedrock, land use, and water use. The infiltration of water pumped from deeper parts of the aquifer system and surface water applied to fields for irrigation has resulted in recharge to the shallow unconfined part of the aquifer system with water that has been exposed to agricultural chemicals and natural salts concentrated by evapotranspiration. Groundwater in the coarse-grained alluvial-fan deposits on the east side of the valley is generally oxic, while in the finer grained basin and lake deposits near the axis of the valley the water is usually anoxic. The redox environment in the aquifer strongly influences the potential for degradation or accumulation of redox sensitive constituents, such as selenium, arsenic, uranium, nitrate, and some pesticides and volatile organic compounds.

Generally, concentrations of selenium in groundwater are highest in areas of the western San Joaquin Valley where soil selenium and groundwater salinity are high, where the water table has been near the land surface and evaporative concentration has occurred, and where groundwater is oxic. Groundwater in the central San Joaquin Valley that historically was the discharge zone for regional groundwater flow generally is chemically reduced and has higher concentration of arsenic than groundwater in the eastern and western alluvial fan areas. Naturally-occurring uranium is commonly adsorbed to aquifer sediments derived from the Sierra Nevada. Groundwater recharge and discharge has caused oxygen-rich, high-alkalinity water to move downward and to likely mobilize uranium adsorbed to sediments. Groundwater in the eastern Central Valley is susceptible to increasing concentrations of uranium where concentrations of bicarbonate are elevated and water is being pumped from deeper parts of the aquifer.

Agricultural development in the Central Valley has affected groundwater quality by adding millions of pounds of nitrate and pesticides to the land surface and modifying groundwater recharge so that these compounds can be more easily transported to the subsurface. The amount of coarse-grained sediments in the subsurface is a major factor in the susceptibility of groundwater to nitrate contamination. Sediment texture influences the rates of infiltration and groundwater flow, which control how rapidly water at the surface, with high concentrations of nitrate, can infiltrate the soil and move downward into the aquifer. Concentrations of nitrate in both the shallow and deep parts of the aquifer system in the eastern San Joaquin Valley have gradually increased during the last 50 years. Pesticides have been used intensively in the Central Valley for many years and are expected to be detected widely throughout the area. Local-scale studies indicate that DBCP detections and concentrations may increase in the deeper part of the aquifer system in the future because of the dominantly downward movement of groundwater and the lack of significant attenuation processes of the compound in the subsurface. The high detection frequency of bentazon in shallow groundwater in a Sacramento Valley rice field setting almost 10 years since its last known use in the area suggests that it is easily transported to the water table and does not readily degrade.

References Cited

Belitz, Kenneth, Dubrovsky, Neil M., Burow, Karen, Jurgens, Bryant, and Johnson, Tyler, 2003, Framework for a ground-water quality monitoring and assessment program for California: U.S. Geological Survey Water-Resources Investigations Report 03–4166, 78 p. Available at http://pubs.usgs.gov/wri/wri034166/.

Bennett, G.L., V, Belitz, Kenneth, Milby Dawson, B.J., 2006, California GAMA Program: ground-water quality data in the northern San Joaquin basin study unit, 2005: U.S. Geological Survey Data Series 196, 122 p. Available at http://pubs.usgs.gov/ds/2006/196/.

Bertoldi, G.L., Johnston, R.H., and Evenson, K.D., 1991, Ground water in the Central Valley, California—A summary report: U.S. Geological Survey Professional Paper 1401–A, 44 p.

Burow, K.R., Stork, S.V., and Dubrovsky, N.M., 1998a, Nitrate and pesticides in ground water in the eastern San Joaquin Valley, California: Occurrence and trends: U.S. Geological Survey Water-Resources Investigations Report 98–4040, 33 p. Available at http://ca.water.usgs.gov/sanj/pub/usgs/wrir98-4040a/wrir98-4040a.html.

Burow, K.R., Shelton, J.L., and Dubrovsky, N.M., 1998b, Occurrence of nitrate and pesticides in ground water beneath three agricultural land-use settings in the eastern San Joaquin Valley, California, 1993–1995: U.S. Geological Survey Water-Resources Investigations Reports 97–4284, 51 p. Available at http://ca.water.usgs.gov/sanj/pub/usgs/wrir97-4284/wrir97-4284.html.

Burow, K.R., Panshin, S.Y., Dubrovsky, N.M., VanBrocklin, David, and Fogg, G.E., 1999, Evaluation of processes affecting 1,2-Dibromo-3-Chloropropane (DBCP) concentrations in ground water in the eastern San Joaquin Valley, California: Analysis of chemical data and ground-water flow and transport simulations: U.S. Geological Survey Water-Resources Investigations Report 99–4059, 57 p.

Burow, K.R., Shelton, J.L., Hevesi, J.A., and Weissmann, G.S., 2004, Hydrogeologic characterization of the Modesto area, San Joaquin Valley, California: U.S. Geological Survey Scientific Investigations Report 2004–5232, 62 p. Available at http://pubs.usgs.gov/sir/2004/5232/sir_2004-5232.pdf.

Burow, K.R., Dubrovsky, N.M., and Shelton, J.L., 2007, Temporal trends in concentrations of DBCP and nitrate in ground water in the eastern San Joaquin Valley, California, USA: Hydrogeology Journal, v. 15, no. 5, p. 991–1007.

Burow, K.R., Shelton, J.L., and Dubrovsky, N.M., 2008, Regional nitrate and pesticide trends in ground water in the eastern San Joaquin Valley, California: Journal of Environmental Quality, v. 37, September-October supplement, p. S249–S263.

Burton, C.A. and Belitz, Kenneth, 2008, Ground-water quality data in the southeast San Joaquin Valley, 2005–2006—Results from the California GAMA Program: U.S. Geological Survey Data Series 351, 103 p. Available at http://pubs.usgs.gov/ds/351/.

California Department of Water Resources, 2003, California's groundwater, Update 2003, groundwater maps and descriptions: California Department of Water Resources Bulletin 118, groundwater maps and descriptions, accessed August 7, 2009 at http://www.water.ca.gov/groundwater/bulletin118/gwbasin_maps_descriptions.cfm.

Davis, G.H., Green, J.H., Olmsted, F.H., and Brown, D.W., 1959, Ground-water conditions and storage capacity in the San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 1469, 287 p.

Dawson, B.J., 2001a, Ground-water quality in the southeastern Sacramento Valley aquifer, California, 1996: U.S. Geological Survey Water-Resources Investigations Report 2001–4125, 24 p. Available at http://pubs.usgs.gov/wri/wri014125/.

Dawson, B.J., 2001b, Shallow ground-water quality beneath rice areas in the Sacramento Valley, California, 1997: U.S. Geological Survey Water-Resources Investigations Report 2001–4000, 33 p. Available at http://ca.water.usgs.gov/archive/reports/wrir014000/.

Domagalski, J.L., 1997, Pesticides in surface and ground water of the San Joaquin-Tulare Basin, California; analysis of available data, 1966 through 1992: U.S. Geological Survey Water-Supply Paper 2468, 74 p.

Domagalski, J.L. and Dubrovsky, N.M., 1991, Regional assessment of nonpoint-source pesticide residues in ground water, San Joaquin Valley, California: U.S. Geological Survey Water-Resources Investigations Report 91–4027, 64 p.

Domagalski, J.L., Knifong, D.L., Dileanis, P.D., Brown, L.R., May, J.T., Connor, V., Alpers, C.N., 2000, Water quality in the Sacramento River Basin, California, 1994–98: U.S. Geological Survey Circular 1215, 36 p. Available at http://pubs.usgs.gov/circ/circ1215/.

Dubrovsky, N.M., Neil, J.M., Welker, M.C., and Evenson, K.D., 1991, Geochemical relations and distribution of selected trace elements in ground water of the northern part of the western San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 2380, 51 p.

Dubrovsky, N.M., Deverel, S.J., and Gilliom, R.J., 1993, Multiscale approach to regional ground-water quality assessment—Selenium in the San Joaquin Valley, California in Regional ground-water quality, W.M. Alley, ed.: Van Nostrand Reinhold, New York, p. 537–562.

Dubrovsky, N.M., Kratzer, C.R., Brown, L.R., Gronberg, J.M., and Burow, K.R., 1998, Water quality in the San Joaquin-Tulare Basins, California, 1992–95: U.S. Geological Survey Circular 1159, 38 p. Available at http://pubs.usgs.gov/circ/circ1159/.

Faunt, C.C., Belitz, Kenneth, and Hanson, R.T., 2009, Chapter B. Groundwater availability in California's Central Valley, *in* C.C. Faunt, ed., Groundwater availability of the Central Valley Aquifer, California: U.S. Geological Survey Professional Paper 1766. Available at http://pubs.usgs.gov/pp/1766/.

Faunt, C.C., Hanson, R.T., and Belitz, Kenneth, 2009, Chapter A. Introduction, overview of hydrology, and textural model of California's Central Valley, *in* C.C. Faunt, ed., Groundwater availability of the Central Valley Aquifer, California: U.S. Geological Survey Professional Paper 1766. Available at http://pubs.usgs.gov/pp/1766/.

Faunt, C.C., Hanson, R.T., Belitz, Kenneth, Schmid, Wolfgang, Predmore, S.P., Rewis, D.L., and McPherson, Kelly, 2009, Chapter C. Numerical model of the hydrologic landscape and groundwater flow in California's Central Valley, *in* C.C. Faunt, ed., Groundwater availability of the Central Valley Aquifer, California: U.S. Geological Survey Professional Paper 1766. Available at http://pubs.usgs.gov/pp/1766/.

Fogelman, R.P., 1983, Ground-water quality in the Sacramento Valley, California—Water types and potential nitrate and boron problem areas: U.S. Geological Survey Hydrologic Atlas 651, 1 map.

Fujii, R. and Swain, W.C., 1995, Areal distribution of selected trace elements, Salinity, and major ions in shallow ground water, Tulare Basin, Southern San Joaquin Valley, California: U.S. Geological Survey Water-Resources Investigations Report, 95–4048, 67 p.

Galloway, Devin and Riley, F.S., 1999, San Joaquin Valley, California, *in* Galloway, Devin, Jones, D.R., and Ingebritsen, S.E., eds., Land subsidence in the United States: U.S. Geological Survey Circular 1182, 177 p. Available at http://pubs.usgs.gov/circ/circ1182/.

Gilliom, R.J., Belitz, Kenneth, Deverel, S.J., Dubrovsky, N.M., and Fujii, Roger, 1989, Preliminary assessment of sources, distribution, and mobility of selenium in the San Joaquin Valley, California: U.S. Geological Survey Water-Resources Investigations Report 88–4186, 129 p.

Gronberg, J.M., Dubrovsky, N.M., Kratzer, C.R., Domagalski, J.L., Brown, L.R., and Burow, K.R., 1998, Environmental setting of the San Joaquin-Tulare Basins, California: U.S. Geological Survey Water-Resources Investigations Report 97–4205, 45 p.

Hall, W. H., 1889, Irrigation in California: National Geographic, v. 2, no. 4, p. 281.

Hull, L.C., 1984, Geochemistry of ground water in the Sacramento Valley, California: U.S. Geological Survey Professional Paper 1401–B, 36 p.

Ireland, R.L., Poland, F.F., and Riley, F.S., 1984, Land subsidence in the San Joaquin Valley, as of 1980: U.S. Geological Survey Professional Paper 437–I, 93 p.

Jurgens, B.C., Burow, K.R., Dalgish, B.A., and Shelton, J.L., 2008, Hydrogeology, water chemistry, and factors affecting the transport of contaminants in the zone of contribution of a public-supply well in Modesto, eastern San Joaquin Valley, California: U.S. Geological Survey Scientific Investigations Report 2008–5156, 78 p. Available at http://pubs.usgs.gov/sir/2008/5156.

Kulongoski, Justin and Belitz, Kenneth, 2006, Ground-water ambient monitoring and assessment program: U.S. Geological Survey Fact Sheet 2004–3088 version 1.1, 2 p. Available at http://pubs.usgs.gov/fs/2004/3088/.

Landon, M.K. and Belitz, Kenneth, 2008, Ground-water quality data in the Central Eastside San Joaquin Basin 2006: results from the California GAMA Program: U.S. Geological Survey Data Series 325, 88 p. Available at http://pubs.usgs.gov/ds/325/.

Maupin, M.A., and Barber, N.L., 2005, Estimated withdrawals from principal aquifers in the United States, 2000: U.S. Geological Survey Circular 1279, 46 p. Available at http://pubs.usgs.gov/circ/2005/1279/.

McKinney, T.S., and Anning, D.W., 2009, Geospatial data to support analysis of water-quality conditions in basin-fill aquifers in the southwestern United States: U.S. Geological Survey Scientific Investigations Report 2008–5239. Available at http://pubs.usgs.gov/sir/2008/5239/.

McMahon, P.B. and Chapelle, F.H., 2008, Redox processes and water quality of selected principal aquifer systems: Ground Water, vol. 46, p. 259–271.

Mendenhall, W.C., Dole, R.B., and Stabler, Herman, 1916, Ground water in San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 398, 310 p.

Milby Dawson, B.J., Bennett, G.L. V, and Belitz, Kenneth, 2008, Ground-water quality data in the southern Sacramento Valley, California, 2005—Results from the California GAMA Program: U.S. Geological Survey Data Series 285, 99 p. Available at http://pubs.usgs.gov/ds/285.

Nady, Paul, and Larragueta, L.L., 1983, Development of irrigation in the Central Valley of California: U.S. Geological Survey Hydrologic Investigations Atlas HA–649, scale 1:500,000, 2 sheets.

Page, R.W., 1986, Geology of the fresh ground-water basin of the Central Valley, California, with texture maps and sections: U.S. Geological Survey Professional Paper 1401–C, 54 p.

Phillips, S.P., Burow, K.R., Rewis, D.L., Shelton, Jennifer, and Jurgens, Bryant, 2007, Section 4. Hydrogeologic setting and ground-water flow simulations of the San Joaquin Valley regional study area, California, *in* Paschke, S.S., ed., Hydrogeologic settings and ground-water flow simulations for regional studies of the transport of anthropogenic and natural contaminants to public-supply wells—Studies begun in 2001: U.S. Geological Survey Professional Paper 1737–A, p. 4-1 to 4-31. Available at http://pubs.usgs.gov/pp/2007/1737a/.

Planert, Michael and Williams, J.S., 1995, Ground water atlas of the United States, segment 1, California, Nevada: U.S. Geological Survey Hydrologic Investigations Atlas 730–B, 28 p. Available at http://pubs.usgs.gov/ha/ha730/ch_b/index.html.

Poland, J.F., Lofgren, B.E., Ireland, R.L., and Pugh, R.G., 1975, Land subsidence in the San Joaquin Valley, California, as of 1972: U.S. Geological Survey Professional Paper 437–H, 78 p.

Saucedo, G. L., Bedford, B. R., Raines, G. L., Miller, R. J., and Wentworth, C. M., 2000, Modified from California Division of Mines and Geology, CD-ROM 2000–007, GIS data for the geologic map of California.

Shelton, J.L., 2005, Assessment of shallow ground-water quality in recently urbanized areas of Sacramento, California, 1997: U.S. Geological Survey Scientific Investigations Report 2005–5148, 51 p. Available at http://pubs.usgs.gov/sir/2005/5148/.

Shelton, J.L., Pimentel, Isabel, Fram, M.S., and Belitz, Kenneth, 2008, Ground-water quality data in the Kern County subbasin study unit, 2006—Results from the California GAMA Program: U.S. Geological Survey Data Series 337, 75 p. Available at http://pubs.usgs.gov/ds/337.

Schmitt, S.J., Fram, M.S., Milby Dawson, B.J., and Belitz, Kenneth, 2008, Ground-water quality data in the middle Sacramento Valley study unit, 2006—Results from the California GAMA Program: U.S. Geological Survey Data Series 385, 100 p. Available at http://pubs.usgs.gov/ds/385/.

U.S. Census Bureau,1995, California population of counties by decennial census 1900 to 1990, compiled and edited by R.L. Forstall: accessed August 26, 2008 at http://www.census.gov/population/cencounts/ca190090.txt.

U.S. Environmental Protection Agency, 2008, List of drinking water contaminants & their MCLs: accessed September 23, 2008 at http://www.epa.gov/safewater/contaminants/index.html.

U.S. Geological Survey, 2003, Multi-Resolution Land Characteristics (MRLC) Consortium, National Land Cover Database (NLCD 2001), accessed May 17, 2010 at http://www.mrlc.gov/.

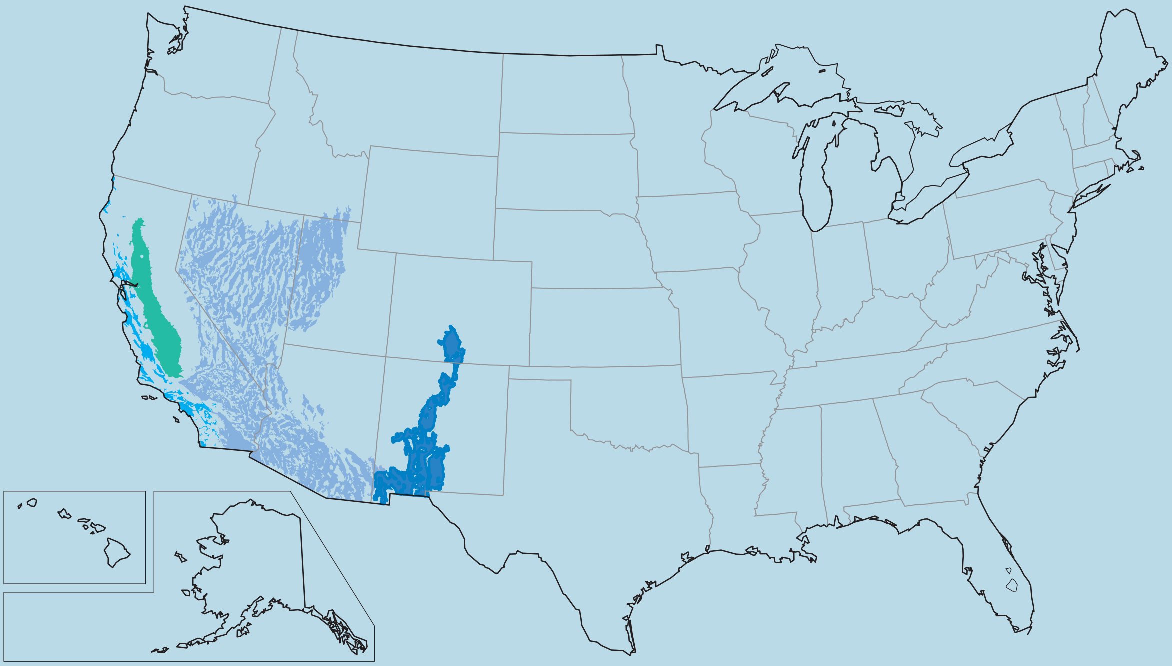
Williamson, A.K., Prudic, D.E., and Swain, L.A., 1989, Ground-water flow in the Central Valley, California: U.S. Geological Survey Professional Paper 1401–D, 127 p.

This page intentionally left blank.

Publishing support provided by the U.S. Geological Survey
Publishing Network, Sacramento and Tacoma Publishing Service Centers

For more information concerning the research in this report, contact the
Director, Utah Water Science Center
U.S. Geological Survey
2320 Orton Circle
West Valley City, Utah 84119
http://ut.water.usgs.gov

Thiros and others—**Conceptual Understanding and Groundwater Quality of Selected Basin-Fill Aquifers in the Southwestern United States**—Professional Paper 1781



Southwest Principal Aquifers—Includes (from left to right) California Coastal Basin aquifers, Central Valley aquifer system, Basin and Range basin-fill aquifers, and Rio Grande aquifer system