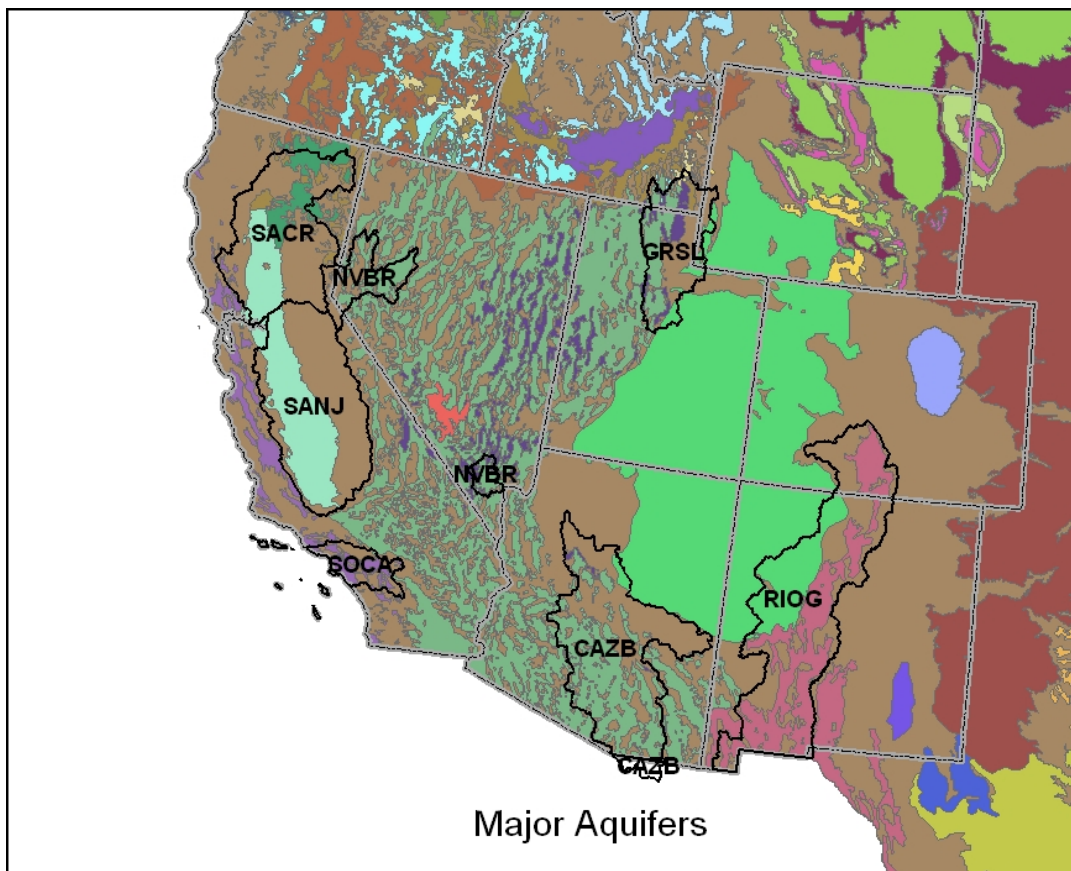


**Regional Synthesis of the Basin & Range/California
Coastal Basins/Central Valley/Rio Grande Aquifer Systems
for the SWPA team (updated April 24, 2006)**



Ground water is an important source of water to many cities and agricultural communities in the basins of the arid and semiarid southwestern United States and the quality of this resource, now and in the future, is of concern. The U.S. Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) Program is studying the water quality of the basin-fill aquifers in the Southwest and the issues that affect water quality at a regional scale. Principal aquifers included in the regional analysis are the Basin and Range basin-fill aquifers in Nevada, Utah, Arizona, and California, the Rio Grande aquifer system in New Mexico and Colorado, and the southern Coastal Basins and Central Valley aquifer systems in California. These principal aquifers consist primarily of unconsolidated sediment, have similar land- and water-use practices, and face many of the same water-quality issues.

This document contains links to selected online USGS (primarily NAWQA) publications pertaining to ground water-quality studies done in the Southwest and 'copied and pasted' sections of the reports describing the studies. This was done to provide SWPA team members with references and a synthesis of the work done during Cycle I in the region.

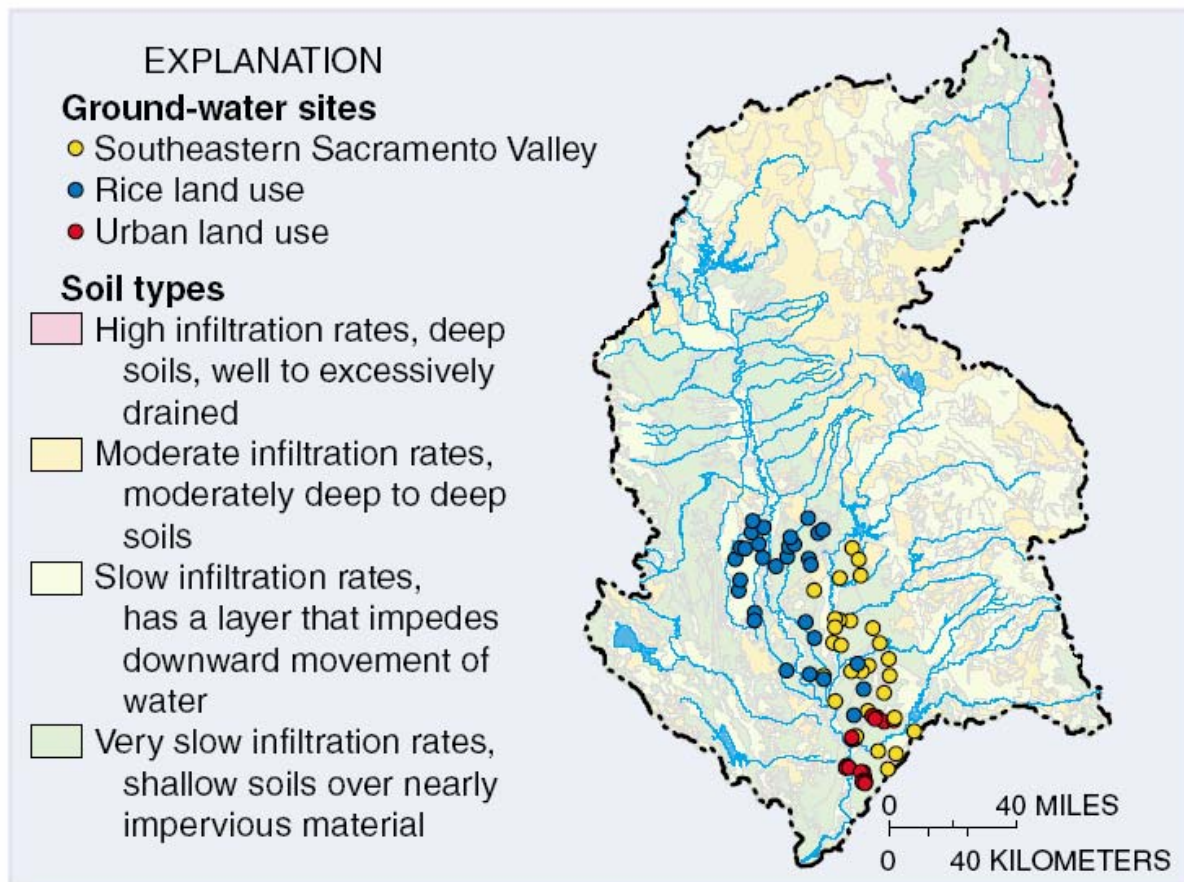
SACR

Water Quality in the Sacramento River Basin, California, 1994–98

By Joseph L. Domagalski, Donna L. Knifong, Peter D. Dileanis, Larry R. Brown, Jason T. May, Valerie Connor, and Charles N. Alpers

U.S. GEOLOGICAL SURVEY CIRCULAR 1215

<http://water.usgs.gov/pubs/circ/circ1215/>



SACR Ground-Water Chemistry

Aquifer Survey	Major ions, nutrients, pesticides, trace elements, volatile organic compounds, and radon; to describe the overall water quality and natural chemistry in a surficial aquifer.	Domestic wells in the southeastern Sacramento Valley.	31 wells	Once in 1996
Land-Use Effects— Agriculture (rice)	Major ions, nutrients, trace elements, and pesticides; to describe the water quality and natural chemistry in a surficial aquifer in an agricultural setting.	Newly drilled monitoring wells completed near the water table in a surficial aquifer beneath or near rice fields.	28 wells	Once in 1997
Land-Use Effects— Urban	Major ions, nutrients, pesticides, trace elements, volatile organic compounds, and radon; to describe the overall water quality and natural chemistry in a surficial aquifer.	Newly drilled monitoring wells completed near the water table in a surficial aquifer beneath a recently urbanized area.	19 wells	Once in 1998

The Sacramento Valley supports a diverse agricultural economy, much of which depends on the availability of irrigation water. Water is collected in reservoirs at several locations within the mountains surrounding the Sacramento Valley and is released according to allocations for agricultural, urban, and environmental needs. The reservoirs also are managed for flood control. The reservoirs provide flood protection and allow the storage of water during dry years, but the placement of dams at the reservoirs has blocked migration routes for salmonid fish.

Reservoirs within the Sacramento River Basin have been constructed in the mountainous areas just adjacent to the valley. All major rivers of the Sacramento River Basin have one or more reservoirs, which were constructed during the late 1940s to late 1960s mainly for flood control. Storage capacity of the reservoirs is managed to capture runoff from winter storms. However, stored water is not used solely in the Sacramento River Basin; it is transported to other locations in California and is a major source of supply for Los Angeles and other southern California communities.

Total water use in the Sacramento River Basin is about 18 billion cubic meters per year (m³/yr). Allocations of water for agricultural, urban, and environmental uses are made according to the California water plan (California Department of Water Resources, 1998) but are modified on the basis of the yearly conditions of reservoir storage. For example, during drought years, total allocations are decreased. Pie charts in figure 4 show the percentage of allocations during average years of rainfall for 1995 and projected for 2020. Of those allocations, surface water provides 82 percent and ground water 18 percent of the total demand. During drought years, surface water drops to about 75 percent and ground water rises to about 25 percent of total demand.

More than 8,000 km² of the Sacramento Valley are irrigated. The major crops are rice, fruits, nuts, tomatoes, sugar beets, corn, alfalfa, and wheat. Dairy products also are an important agricultural commodity. The land areas adjacent to the Sacramento Valley are mostly forested (fig. 1).

The largest cities of the basin are in the Sacramento Valley and include Chico, Red Bluff, Redding, and Sacramento. The Sacramento metropolitan area is home to more than 1 million people, which is nearly half of the total population (U.S. Department of Commerce, 1992) in the basin.

The average annual precipitation for the entire Sacramento River Basin is 914 millimeters (mm), most of which falls as rain or snow during November through March. Because little or no rain falls during the summer growing season, irrigation is required for successful agriculture.

Ground Water

Ground-water quality was investigated in the portion of the aquifer of the southeastern Sacramento Valley used for domestic purposes or irrigation and in the shallow portion of the aquifer below the valley's two major land uses—rice cultivation and land that had been urbanized from 5 to 25 years ago. The southeastern Sacramento Valley was chosen for investigation because it is in that region that domestic and irrigation usage of ground water is highest. Although ground water is used in other parts of the valley, its usage is not as great as in the southeastern Sacramento Valley.

The ground-water quality of some areas of the Sacramento Valley, such as the southwestern portion, is not entirely suitable for human or agricultural use because of the presence of elevated concentrations of boron, fluoride, chloride, nitrate, and sulfate (Hull, 1984; Davisson and Criss, 1993). The study conducted in the southeastern Sacramento Valley aquifer was designed to address the suitability of a portion of the aquifer. Specifically, only existing domestic wells were sampled.

Out of 31 wells sampled, only one sample exceeded the drinking water standard for nitrate. The median concentration of nitrate was 1.3 mg/L. Previous NAWQA investigations have found a median nitrate concentration of 0.48 mg/L in major aquifers throughout the United States (Nolan and Stoner, 2000). Therefore, the wells sampled in this study have nitrate values above the national median. One or more pesticides were detected in 9 of 31, or 29 percent, of the wells of the southeastern Sacramento Valley (fig. 19). Simazine was detected in three wells, but the concentrations were very low and were not close to any drinking-water standard. Bentazon was one of the more frequently detected pesticides. Bentazon is a herbicide that was used on rice, but its use was suspended in 1989 pending a review; then, in 1992, it was

formally banned by the California Department of Pesticide Regulation (Miller-Maes and others, 1993). All bentazon concentrations were below drinking-water standards. Volatile organic chemicals (VOCs) are not causing any apparent water-quality problems in the shallow aquifer of the southeastern Sacramento Valley. In one of the wells sampled, which is downgradient from a known point source, eight different VOCs were detected. One of those VOCs (trichloroethene, measured at 5.5 ug/L) exceeded current drinking-water standards (primary MCL is 5 ug/L).

The effects of rice cultivation—one of the most prevalent agricultural practices in the Sacramento Valley—on ground water were examined by drilling and sampling 28 new wells (fig. 20). The wells were drilled to completion near the water table so that the agricultural effects on the most recent water that entered the ground could be assessed. Rice cultivation requires that fields be flooded for the duration of the growing season, which lasts from May through September.

Nutrient concentrations tended to be low in the ground water under the rice fields. The median nitrate concentration was 2 mg/L, and no concentrations exceeded a drinking-water standard. The median nitrate concentration measured in previous NAWQA investigations in agricultural land use settings was 3.4 mg/L (Nolan and Stoner, 2000). Dissolved solids were elevated in rice fields relative to the other ground water sampled in this study. The elevated concentrations are most likely related to evaporation of irrigation water, which leaves behind salt. The effects of these increases in dissolved solids on deeper portions of the aquifer are unknown.

The effects of recent urbanization on the quality of shallow ground water were investigated as part of this NAWQA study (fig. 21). The chosen metropolitan area was that of the city and surrounding counties of Sacramento. The part of the metropolitan area developed between 5 and 25 years ago was chosen for the investigation because it was assumed that the water quality of older urban land might have degraded water quality because it was developed prior to the passage of the Clean Water Act and the period of more recent environmental awareness.

Trichloromethane was the most frequently detected volatile organic chemical (16 of 19 wells). The concentrations were always very low and did not exceed any drinking-water standards. The presence of trichloromethane can be attributed to lawn irrigation using water treated by chlorination. One or more pesticides were detected in 6 of 19 (32 percent) of the wells sampled in the urban land-use study (fig. 19). Atrazine, or its degradation product, was the most frequently detected pesticide in the shallow ground water under the recently urbanized area. The occurrence of atrazine in ground water could not be attributed to either current urban land use or past agricultural production. All atrazine concentrations were below drinking-water standards. An examination of the nitrate data revealed the highest potential contamination of ground water from recent urban development.

Although no wells had concentrations of nitrate above the drinking-water standard, one well had a concentration of 8 mg/L, and 5 of the 19 (26 percent) of the sampled wells had concentrations that exceeded 5 mg/L. The median concentration of nitrate was 2.4 mg/L. The median concentration of nitrate in ground water under urban areas measured in previous NAWQA investigations was 1.6 mg/L (Nolan and Stoner, 2000).

With the exception of arsenic, trace elements generally were not found to be a problem from the perspective of human toxicity. Some wells did have high concentrations of iron and manganese. The primary drinking-water standard for cadmium (5 ug/L) was exceeded in three wells of the rice land-use study. The source of that cadmium is unknown. Arsenic exceeded the current drinking-water standard of 50 ug/L in one urban well and approached the drinking-water standard in three other wells. Arsenic would be more problematic if the drinking-water standard were lowered to 5 ug/L, as proposed by the EPA (U.S. Environmental Protection Agency, 2000). At that level, the standard would be exceeded in 53 percent of the urban wells, 39 percent of the rice land-use wells, and 48 percent of the domestic wells sampled in the southeastern Sacramento Valley. Iron and manganese dissolve when oxygen is absent in the ground water. Although generally nontoxic, these two metals can limit the beneficial uses of the ground water because they may precipitate when the ground water is exposed to air. The precipitation can be severe enough to adversely affect household uses of water as well as plumbing.

Assessment of Shallow Ground-Water Quality in Recently Urbanized Areas of Sacramento, California, 1998

U.S. Geological Survey Scientific Investigation Report 2005-5148 (ONLINE ONLY)

By Jennifer L. Shelton

<http://pubs.usgs.gov/sir/2005/5148/>

Hydrogeology

The study area is located primarily east of the Sacramento River, in the southeast part of the Sacramento Valley. The Sacramento Valley occupies the northern third of the Central Valley, an asymmetric structural trough filled with hundreds of meters of sediment derived primarily from the Sierra Nevada to the east of the Sacramento River and from the Coast Ranges to the west (Page, 1986). Piper and others (1939), Olmsted and Davis (1961), and Page (1986) have described Sacramento Valley geology, but the smallest scale of surficial geologic units is described by Helley and Harwood (1985) and is summarized below.

There are four different types of Quaternary surficial geologic deposits in the study area (*fig. 5*). These include the lower member of the Riverbank Formation consisting of red semiconsolidated gravel, sand, and silt; undivided Basin deposits consisting of fine-grained silt and clay; alluvium consisting of unweathered gravel, sand, and silt; and the Turlock Lake Formation consisting of deeply weathered and dissected arkosic gravels with minor resistant metamorphic rock fragments and quartz pebbles, sand, and silt (Helley and Harwood, 1985). Soils are predominantly fine grained with slow to very slow infiltration rates (Tugel, 1993).

The top 60 m of the aquifer generally is unconfined (Bertoldi and others, 1991), but because of the discontinuity of clay lenses, there is not a well-defined lower boundary for the unconfined interval. Water-level information for the urban study area is minimal. Regional water-level data from 1977 to 1995 for the Sacramento County area indicate that depth to water from land surface (water table depth) varies from less than 1.5 m to greater than 69 m in areas with high pumping rates (Eric Senter, California Department of Water Resources, computer files and written commun., 1997). A map of equal elevation of ground-water levels for the spring of 1968 (California Department of Water Resources, 1974) indicates that pumping depressions have a maximum depth of only 15 m, in contrast to the data from 1977 to 1995 that indicate pumping depressions as deep as about 61 m. Thus, shallow ground-water levels have declined significantly in some areas since recent urban development has occurred.

Climate

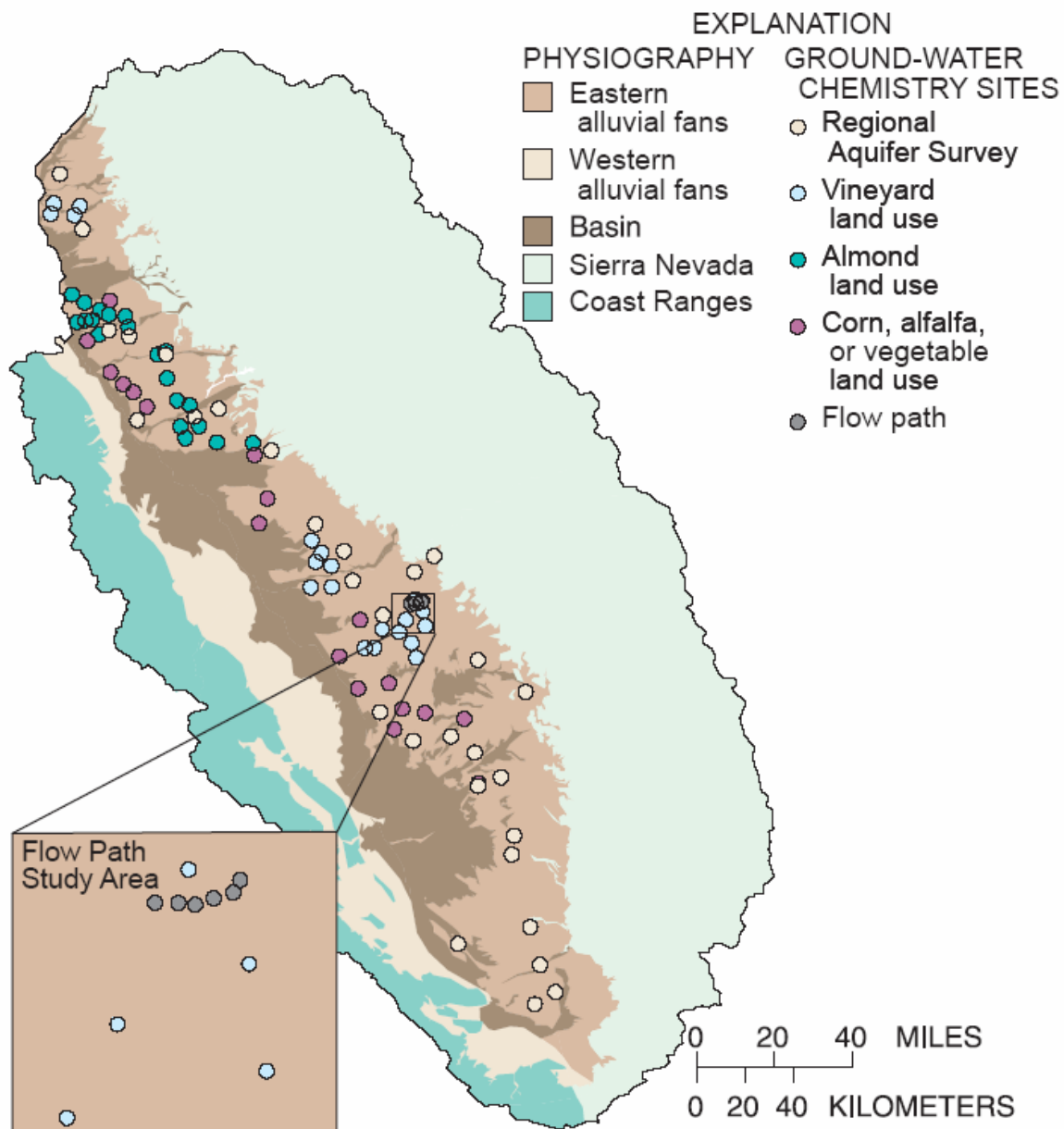
Average annual precipitation for Sacramento County is 46 cm (Earthinfo, 1996), but extremes ranged from 18 cm in 1976 to 96 cm in 1983. Eighty-eight percent of the long-term average annual precipitation falls between November and April. Temperature extremes in Sacramento County ranged from -7.8°C to 46°C between 1961 and 1990 (Western Regional Climate Center, 1997). A normal year evapotranspiration value, based on long-term records of climatic data, for Sacramento County is approximately 132 cm (California Department of Water Resources, 1992). Fluctuations in rainfall and temperature can affect the quality of ground water by influencing the amount of volatile organic compounds (VOCs) in rainfall, volatilization of VOCs to the atmosphere, seasonal loadings of pesticides in rainfall, and relative amounts of runoff and infiltration. Sorption and denitrification processes also may be affected by the climate and can affect the likelihood of detecting VOCs, pesticides, or nitrate. Precipitation, soil characteristics, and evapotranspiration data can be used to estimate a recharge rate and the traveltime that it would take for water to reach the water table.

SANJ**Water Quality in the San Joaquin-Tulare Basins, California, 1992-95**

By Neil M. Dubrovsky, Charles R. Kratzer, Larry R. Brown, JoAnn M. Gronberg, and
Karen R. Burow

U.S. GEOLOGICAL SURVEY CIRCULAR 1159

<http://pubs.usgs.gov/circ/circ1159/>



Location of ground-water chemistry sites.

SANJ Ground-Water Chemistry				
Regional Aquifer Survey—eastern alluvial fans	Major chemical constituents, nutrients, 83 pesticides, 60 volatile organic compounds, and radon to determine occurrence of these constituents in this region.	Domestic wells in the eastern alluvial fans, San Joaquin Valley.	30	Once in 1995
Land-use effects—corn, alfalfa, and vegetable row crops	Major chemical constituents, nutrients, 83 pesticides, 60 volatile organic compounds, and radon to describe the effects of agricultural land use on shallow ground water, eastern alluvial fans.	Shallow domestic wells; Shallow monitoring wells; >50 percent corn, alfalfa, and vegetables grown in rotation within a 0.25-mile radius.	20 10	Once in 1995
Land-use effects—almond orchards	Major chemical constituents, nutrients, 83 pesticides, 60 volatile organic compounds, and radon to describe the effects of agricultural land use on shallow ground water, eastern alluvial fans.	Shallow domestic wells; Shallow monitoring wells; >50 percent almond orchards within 0.25 mile radius.	20 10	Once in 1994
Land-use effects—vineyards	Major chemical constituents, nutrients, 83 pesticides, 60 volatile organic compounds, and radon to describe the effects of agricultural land use on shallow ground water, eastern alluvial fans.	Shallow domestic wells; Shallow monitoring wells; >50 percent vineyards within 0.25-mile radius.	20 10	Once in 1993
Chemical and physical processes along ground-water flow paths	Major chemical constituents, nutrients, 83 pesticides, 60 volatile organic compounds, radon, transformation products of 1,2-dibromo-3-chloropropane, and constituents used to estimate date when ground water was recharged.	20 wells at 6 sites along an approximate ground-water flow path beneath vineyard land use in eastern alluvial fan, San Joaquin Valley.	20	Once in 1994 Once in 1995

The San Joaquin–Tulare Basins NAWQA Study Unit is located in central California and includes the San Joaquin Valley, the eastern slope of the Coast Ranges, and the western slope of the Sierra Nevada. The Sierra Nevada are predominantly forested land, and the Coast Ranges and the foothills of the Sierra Nevada are rangeland. Almost the entire valley floor is used for agricultural land. In 1987, about 10.5 million acres in the San Joaquin Valley was farmland (San Joaquin Valley Drainage Program, 1990, p. 50). Abundant water, combined with the long growing season, results in an exceptionally productive agricultural economy in the San Joaquin Valley. In 1987, approximately 10.2 percent of the total value of agricultural production in the United States came from California, 49 percent of which, or \$6.82 billion, was from the San Joaquin Valley.

Thirty-eight percent of the surface water is imported from the Sacramento–San Joaquin Delta through the Delta-Mendota Canal and California Aqueduct (U.S. Bureau of Reclamation, 1990; California Department of Water Resources, 1991). Most of the rest of the surface water is from the Sierra Nevada. Surface water from the Sierra Nevada is of very high quality, but major changes in water quality occur when surface water enters the San Joaquin Valley. These changes are primarily due to the large amount of irrigated agriculture, which affects the quality of both surface and ground water in the valley. Irrigation return water may reach surface water as direct runoff (tailwater), as water from subsurface drainage systems installed to control the water table, or as ground water discharged through riverbeds. The result can be increased concentrations of dissolved solids, nutrients, pesticides, and, in some areas, trace elements. Irrigation is the largest source of recharge to the regional aquifer, and this ground-water recharge can contain higher concentrations of dissolved solids than natural recharge in the past. This recharge also may contain elevated concentrations of nutrients, pesticide residues, and trace elements.

The distribution of precipitation, and consequently runoff, in the San Joaquin–Tulare Basins is highly influenced by topography. Mean annual precipitation on the valley floor ranges from less than 5 inches in the south to 15 inches in the north. Precipitation in the Coast Ranges varies from less than 10 inches (at Panoche 2 W) to more than 20 inches. Average annual precipitation in the Sierra Nevada, mostly in the form of snow, ranges from about 20 inches in the lower foothills to more than 80 inches at some higher elevation sites (Calaveras Big Trees).

CONCEPTUAL SYSTEM: SANJ

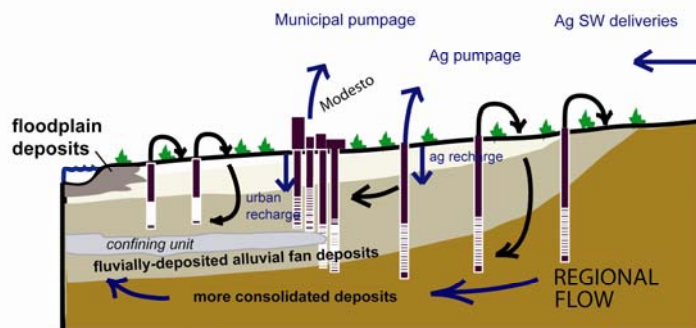
CONVERGENT FLOW TOWARD CENTER OF BASIN

LATERALLY DRIVEN

overlain with

PUMPING AND REAPPLICATION OF IRRIGATION WATER

VERTICALLY DRIVEN

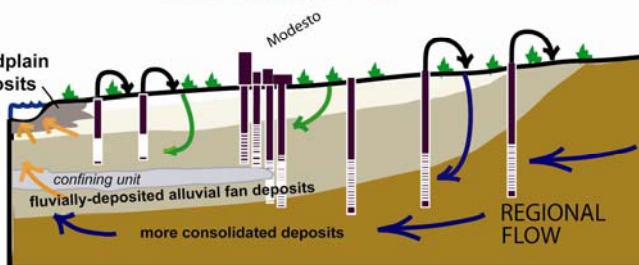


GROUND-WATER FLOW SYSTEM

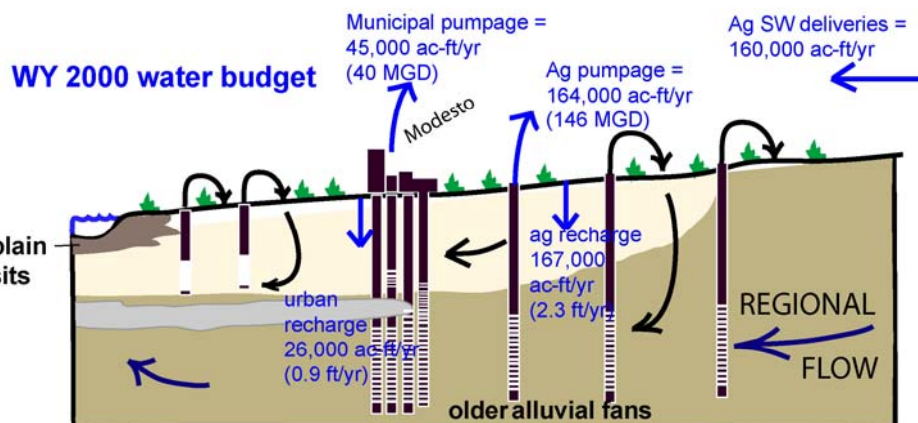
Oxic conditions prevail in shallow portion of aquifer as pumping and recharge from surrounding ag create local flow cells.

Highly reduced conditions commonly occur in discharge zones as long flowpaths terminate and residence time, organic matter, and nutrients increase.

Variably reduced conditions occur in deeper portions of the aquifer as ground water migrates downward and flows laterally.



GEOCHEMICAL CHARACTERISTICS



Nitrate concentrations in 24 percent (21 of 88) of the domestic wells sampled during 1993–95 in the regional aquifer survey and land-use studies of the eastern San Joaquin Valley exceeded the drinking-water standard of 10 mg/L established by the USEPA. Furthermore, ground-water samples from 77 percent of the wells had nitrate concentrations greater than 2 mg/L, which is believed to represent background concentrations (Mueller and Helsel, 1996). These findings indicate that ground-water quality has been degraded over a large part of this aquifer because of the input of nitrate from human activity. Ground-water samples collected in 1995 from the 30 domestic wells in the regional aquifer survey (median well depth 182 feet) had a median nitrate concentration of 4.6 mg/L; 5 of the 30 wells (17 percent) exceeded the

USEPA drinking-water standard. The median concentration of 4.6 mg/L was higher than the median of 2.4 mg/L for ground water in similar alluvial settings with agricultural land use nationwide (Mueller and others, 1995).

Nitrate concentrations in samples from shallow domestic wells were related to nitrate application, sediment texture, and potential for nitrate removal by chemical reactions

Sources of nitrate from agriculture— fertilizer and manure—were estimated for a 0.25-mile-radius circle centered at each well in each of the three agricultural land-use settings. Nitrate concentrations in ground water in the three land-use settings were related to estimates of the amount of nitrogen applied: the greatest amount of nitrogen was applied in the almond land-use setting; a slightly lower amount was applied in the corn, alfalfa, and vegetable land-use setting; and the smallest amount was applied in the vineyard land-use setting (Burow and others, in press, a). The estimated amount of nitrogen applied to individual sites was not strongly related to nitrate concentrations in ground-water samples from the wells, however. These results indicate that estimates of the amount of nitrogen applied are a fair indicator of nitrate concentrations for an area; however, the estimates are not a good predictor of concentration in an individual well. This lack of predictability probably is due to several factors.

The amount of coarse-grained sediments (sand- or gravel-sized) in the subsurface, which is referred to as sediment texture, is a major factor in the susceptibility of a site to nitrate contamination. The sediment texture influences the rates of infiltration and ground-water flow in the aquifer, which controls how rapidly water at the surface, with high nitrate concentrations, can infiltrate the soil and move downward to a well in the aquifer. The sediment textures in the almond and vineyard land-use settings were generally coarse-grained and conducive to rapid infiltration and ground-water flow. The sediment texture in the corn, alfalfa, and vegetable land-use setting was generally finegrained with abundant clay, resulting in slow rates of infiltration and groundwater flow. These contrasts in sediment texture, considered along with the contrasts in the amount of nitrogen applied, indicate that nitrate concentrations in ground water were highest where high susceptibility and high amounts of nitrogen applied occurred together (the almond land-use setting); nitrate concentrations in ground water were lowest where the amount of nitrogen applied was low, even though the susceptibility was high (the vineyard landuse setting); and nitrate concentrations in ground water were intermediate where the amount of nitrogen applied was high, but the susceptibility was low (the corn, alfalfa, and vegetable land-use setting) (Burow and others, in press, a).

Nitrate in ground water can also be removed by biochemical reactions such as nitrate reduction, in which nitrate is converted to nitrogen gas. These biochemical reactions can happen in ground water that has very low concentrations of dissolved oxygen. The chemical traits that indicate a potential for nitrate reduction, such as low dissolved-oxygen concentrations and high concentrations of iron and manganese, existed in ground-water samples from the corn, alfalfa, and vegetable land-use setting. The few samples that have these chemical traits do in fact have low or nondetectable nitrate concentrations. In contrast, there is little evidence of nitrate reduction in samples from the almond and vineyard land-use settings. Therefore, ground water in parts of the corn, alfalfa, and vegetable land-use setting is less susceptible to nitrate contamination than ground water in the other two land-use settings because nitrate may be removed by biochemical processes.

The presence of fine-grained sediment textures and evidence of nitrate reduction at some sites in the corn, alfalfa, and vegetable land-use setting are a result of its location on the lowest parts of the eastern alluvial fans, near the boundary between the alluvial fans and basin, where sediments were deposited by different sedimentary processes. As a result, sediment texture and chemical conditions in the corn, alfalfa, and vegetable land-use setting are more variable than in the almond and vineyard land-use settings. This high variability makes it difficult to generalize the conclusions from the overall data set to specific sites.

Nitrate concentrations in ground water have increased in the eastern San Joaquin Valley

Analyses of several thousand ground-water samples were compiled from USGS and USEPA data bases to evaluate the long-term changes in nitrate concentrations. Data from wells in the eastern San Joaquin Valley that were less than or equal to 200 feet deep indicate that median nitrate concentrations increased

significantly from the 1950s to the 1960s, and from the 1970s to the 1980s. From 1950 to 1980, the amount of nitrogen fertilizer applied in the eastern San Joaquin Valley counties increased from 114 to 745 million pounds per year, an increase of 554 percent. The number of dairies and other confined-animal feedlots, and hence manure production, also have increased greatly during this period. However, estimates indicate that nitrogen fertilizer is the largest source of nitrate in the eastern San Joaquin Valley (Gronberg and others, in press). Of course, this generalization may not be the case for areas where the source may be attributed to confined-animal feedlots located close together. As indicated by a much smaller but better controlled data set, nitrate concentrations increased over less than a decade. Of the 30 wells in the regional aquifer survey in the eastern San Joaquin Valley that were sampled in 1995, 23 also had been sampled during 1986–87. The median nitrate concentration of this subset of 23 domestic wells increased from 2.4 mg/L during 1986–87 to 4.8 mg/L in 1995 (Burow and others, in press, b). The increase in nitrate concentration is likely attributed to increased fertilizer use from 1950 to 1980. Analyses of samples from the 20 multilevel monitoring wells in the vineyard land-use setting indicated that nitrate concentrations have generally increased over the last four decades. In addition to analyses for nitrate and pesticides, the estimated date when the ground water sampled from these wells was recharged was determined by measuring concentrations of chlorofluorocarbons (CFC). CFC concentrations in samples from the deepest wells indicate that the deepest ground water had entered the aquifer prior to 1960. Nitrate concentrations in the pre-1960 water are generally less than 3 mg/L. In general, ground-water age increases with depth below land surface, whereas nitrate concentrations decrease with depth. The highest nitrate concentrations occur in shallow ground water that was recharged during 1977 to 1992. Because typical domestic well depths in the vicinity of the monitoring well transect are about 90 to 130 feet below land surface, these data suggest that nitrate concentrations could increase in domestic wells in the vineyard land-use setting.

Taken together, these studies offer compelling evidence that nitrate concentrations in the ground water in the eastern San Joaquin Valley have increased over the last four decades; prediction of future change is problematic because of the decrease in the amount of fertilizer application over the last decade.

Environmental Setting of the San Joaquin-Tulare Basins, California

By Jo Ann M. Gronberg, Neil M. Dubrovsky, Charles R. Kratzer, Joseph L. Domagalski, Larry R. Brown, and Karen R. Burow

U.S. Geological Survey Water-Resources Investigations Report 97-4205

<http://ca.water.usgs.gov/sanj/pub/usgs/wrir97-4205/wrir97-4205.html>

Ground-Water Hydrology

The aquifer system of the San Joaquin–Tulare Basins study unit is contained within the southern two-thirds of the Central Valley aquifer system (fig. 15A). The aquifer system is made up of Post-Eocene continental rocks and deposits, which contain most of the fresh water in the valley. Pre-Tertiary and Tertiary marine sediments underlying the continental deposits contain mostly saline water, except where fresh water may have flushed out some of the saline water. The continental deposits consist primarily of lenses of gravel, sand, silt, and clay, derived from fluvial and alluvial fan sediments and interbedded with lesser amounts of lacustrine deposits. The major aquifer in the San Joaquin Valley consists of continental deposits, which are mapped on the surface by physiography, weathering characteristics, and soil type. In the subsurface, separate units are difficult to distinguish because of the difficulty of determining these characteristics on the basis of core samples and indirect methods. Therefore, the distribution of physical properties are more important than specific lithologic formation boundaries in defining the regional and local flow system (Bertoldi and others, 1991).

Abundant lenses of fine-grained deposits (clay, sandy clay, sandy silt, and silt) make up more than 50 percent of the aquifer thickness. The lenses are distributed throughout the valley, with the exception of a few areally extensive units, such as the Corcoran Clay Member (Pleistocene) of the Tulare Formation. The Corcoran Clay is part of the modified E-clay (Page, 1986) in the San Joaquin Valley. The clay is extensive and underlies an area of approximately 5,000 mi². This diatomaceous clay is up to 900 ft in depth and up to 160 ft in thickness beneath the Tulare Lake bed with a mean thickness of 55 ft (Davis and others, 1959; Page, 1986).

The thickness of the aquifer system, based largely on the generalized thickness of continental deposits, averages 2,400 ft and increases from north to south to a maximum thickness of more than 9,000 ft near Bakersfield. In the southeastern San Joaquin Valley, sandy marine beds underlying the continental deposits contain freshwater and are hydrologically part of the aquifer system (Bertoldi and others, 1991, p. 9).

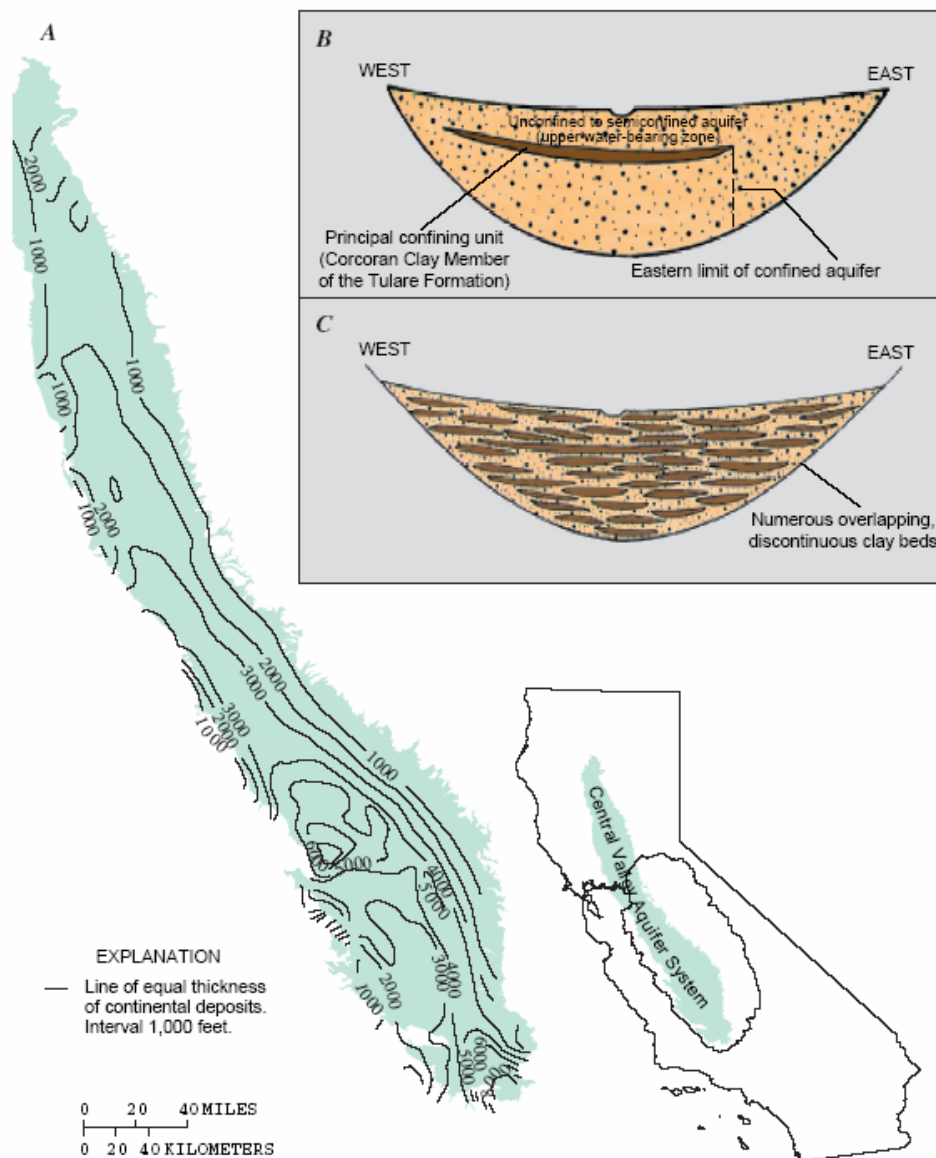


Figure 15. Central Valley, California aquifer system. (A) Thickness of continental deposits (adapted from Bertoldi and others, 1991). (B) Concept of two-layer aquifer system of the San Joaquin Valley, California (adapted from Poland and Lofgren, 1984). (C) Concept of single-layer aquifer system of the San Joaquin Valley, California (adapted from Williamson and others, 1989).

Regional Ground-Water Flow System

The regional ground-water flow system is affected by the numerous lenses of fine-grained materials distributed throughout the aquifer. Two concepts of the aquifer system have been developed, based on the role of the fine-grained lenses on regional flow (fig. 15B,C). Poland and Lofgren (1984) define the aquifer in the San Joaquin Valley as an unconfined to semiconfined upper zone separated from a lower confined zone by the Corcoran Clay Member of the Tulare Formation (fig. 15B). In parts of the valley, the Corcoran

Clay Member separates zones with distinctly different water chemistries (Davis and Poland, 1957; Davis and others, 1959; Davis and Coplen, 1989).

Differences in hydraulic head and water chemistry above and below the Corcoran Clay support the hypothesis that the clay separates the aquifer system into unconfined or semiconfined zones (above the clay) and a confined zone (below the clay). However, many fine-grained lenses throughout the valley have a combined thickness of several thousand feet. Also, many wells have been perforated above and below the Corcoran Clay Member, allowing flow through the well casings and gravel packs. In the vicinity of these wells, hydraulic head is equalized. In other areas where the Corcoran Clay Member is absent, head differences between shallow and deeper wells result from restriction of vertical movement by intervening clay layers (Williamson and others, 1989, p. D33–D36).

For these reasons, Williamson and others (1989) developed the concept of a single heterogeneous aquifer with varying vertical leakance and confinement (fig. 15C). They concluded that when the Central Valley aquifer system is examined at the regional scale, the Corcoran Clay Member is less important than the combined effect of the fine-grained lenses in controlling vertical flow. Williamson and others (1989) developed a numerical model that uses the concept of a single, mostly confined, heterogeneous aquifer system to estimate regional groundwater flow. Regardless of the role of the Corcoran Clay Member in the physical flow system, the contrasts in water chemistry above and below the clay make it an important marker in any study of ground-water quality.

Predevelopment Ground-Water Flow

Prior to development, ground water generally moved from recharge areas in the higher ground surrounding the valley floor toward lower areas in the center of the valley. Figure 16A depicts a hydrologic section across the valley floor, showing lateral groundwater flow roughly following the gradient of land surface from high to low elevation. Figure 16B is a water-table contour map; the predevelopment contours roughly correspond to land surface. Recharge was primarily from streams entering the valley from the Sierra Nevada, and to a lesser extent, directly from precipitation. Recharge through stream channels took place mostly in their upper reaches shortly after entering the valley. Most ground water was discharged as evapotranspiration in the central trough of the valley, and to a lesser extent, to streams. Potential evapotranspiration in the valley's center is about 49 in./yr, exceeding precipitation rates (Bertoldi and others, 1991, p. 17).

About 1900, flowing wells were documented along the valley trough (Mendenhall and others, 1916), indicating the upward direction of hydraulic gradients in the central part of the valley. In the Tulare Basin, most of the ground water flowed to the Tulare Lake area and evapotranspired. In the San Joaquin Basin, the Sacramento–San Joaquin Delta is the only outlet for natural discharge of surface or ground water (Bertoldi and others, 1991, p. 17). Natural recharge and discharge simulated by Williamson and others (1989) showed more recharge than discharge along the valley margins and more discharge than recharge in the low-lying central parts, corresponding to areas with flowing wells.

Ground-Water Development and Response

Ground-water development began in the Central Valley about 1880. An extensive surface-water irrigation system supplied water to the southern San Joaquin Valley, and ground water provided only a small part of the irrigation water. After 1900, well construction and ground-water withdrawal increased slowly. By 1913, annual ground-water pumpage in the Central Valley was estimated to be 360,000 acre-ft (Bertoldi and others, 1991, p. 22). Around 1930, additional groundwater development for irrigation was encouraged by the development of a deep-well turbine that pumped more efficiently from greater depths (Williamson and others, 1989). During the 1940s and 1950s, the ground-water pumpage for irrigation increased sharply (Bertoldi and others, 1991); pumpage estimates in the San Joaquin Valley were 6 million acre-ft/yr in 1948 (Williamson and others, 1989) and 7.5 million acre-ft/yr in 1952 (Davis and others, 1964).

Ground-water pumpage in the Central Valley increased to an average rate of about 11.5 million acre-ft annually in the 1960s and 1970s and provided about one-half of the irrigation water. However, the proportion between surface water and ground water used for irrigation varied substantially from wet to dry years. During wet years, inexpensive surface water was used for irrigation. During dry periods, many farms

used predominantly ground water. For example, during the drought of 1976–1977, ground-water pumpage in the Central Valley increased to a high of 15 million acre-ft in 1977.

Percolation of irrigation water past crop roots has replaced infiltration of intermittent stream water as water through wells and evapotranspiration from crops has replaced natural evapotranspiration as the primary mechanism of discharge (Belitz and Heimes, 1990). Large withdrawals of ground water from the lower zone in the west side of the San Joaquin Valley (fig. 17A) caused a decline in hydraulic heads (as much as 400 ft by 1961) and a reversal of lateral ground-water flow (Bull and Miller, 1975). Water in the lower zone of the aquifer system, which previously flowed toward the center of the valley, flowed toward the pumping center on the west side of the valley in the 1950s and 1960s (fig. 17A). In addition, a reversal in the vertical gradient caused downward flow from the upper to the lower zone. Heavy ground-water pumpage during dry years also caused land subsidence due to sediment compaction, and more than one-half of the San Joaquin Valley is affected. Land subsidence of nearly 30 ft has been documented in some areas (Bertoldi and others, 1991, p. 32).

The water table rose in the northwestern San Joaquin Valley because of recharge from surface water irrigation. On the southern and eastern sides of the valley, the water table declined because of dependence on ground water for irrigation and the use of many shallow irrigation wells (Williamson and others, 1989, p. 63). Beginning in 1950, water was diverted through the Friant–Kern Canal from below Millerton Reservoir to the east side of the San Joaquin Valley, and watertable declines were reversed in parts of the area because pumping was reduced. In 1951, surface-water deliveries along the northwest side of the San Joaquin Valley began through the Delta–Mendota Canal and in 1967, surface water deliveries to farms along the west side and near the southern end of the San Joaquin Valley began through the California Aqueduct. The availability of imported surface water following the construction of these canals resulted in a decrease in ground-water pumpage (Belitz and Heimes, 1990). Increased surface-water delivery and decreased ground-water pumpage caused water levels to rise in both the upper and lower zones in most of these areas.

Long-term water-table declines generally are less than 100 ft, except in the southern part of the San Joaquin Valley; increased irrigation recharge in some areas has caused the water table to rise as much as 40 ft (Bertoldi and others, 1991, p. 26) (fig. 17B). Ground water is generally within 20 ft of land surface in the central and western areas of the valley floor (fig. 17B); this coincides roughly with the area of predevelopment flowing wells. The extensive use of imported surface water has resulted in a water table within 5 ft of the land surface underlying 0.8 million acres in the San Joaquin Valley (San Joaquin Valley Drainage Program, 1990b). Because of continued heavy pumping, much of the ground-water system is still overdrafted. For example, in 1990 the net loss was 3.1 million acre-ft of ground water from storage in the San Joaquin Valley; loss of ground water from storage in 1991, the fifth consecutive year of below-normal precipitation, is projected to be about 8 million acre-ft (Carl Hauge, California Department of Water Resources, written commun., 1991).

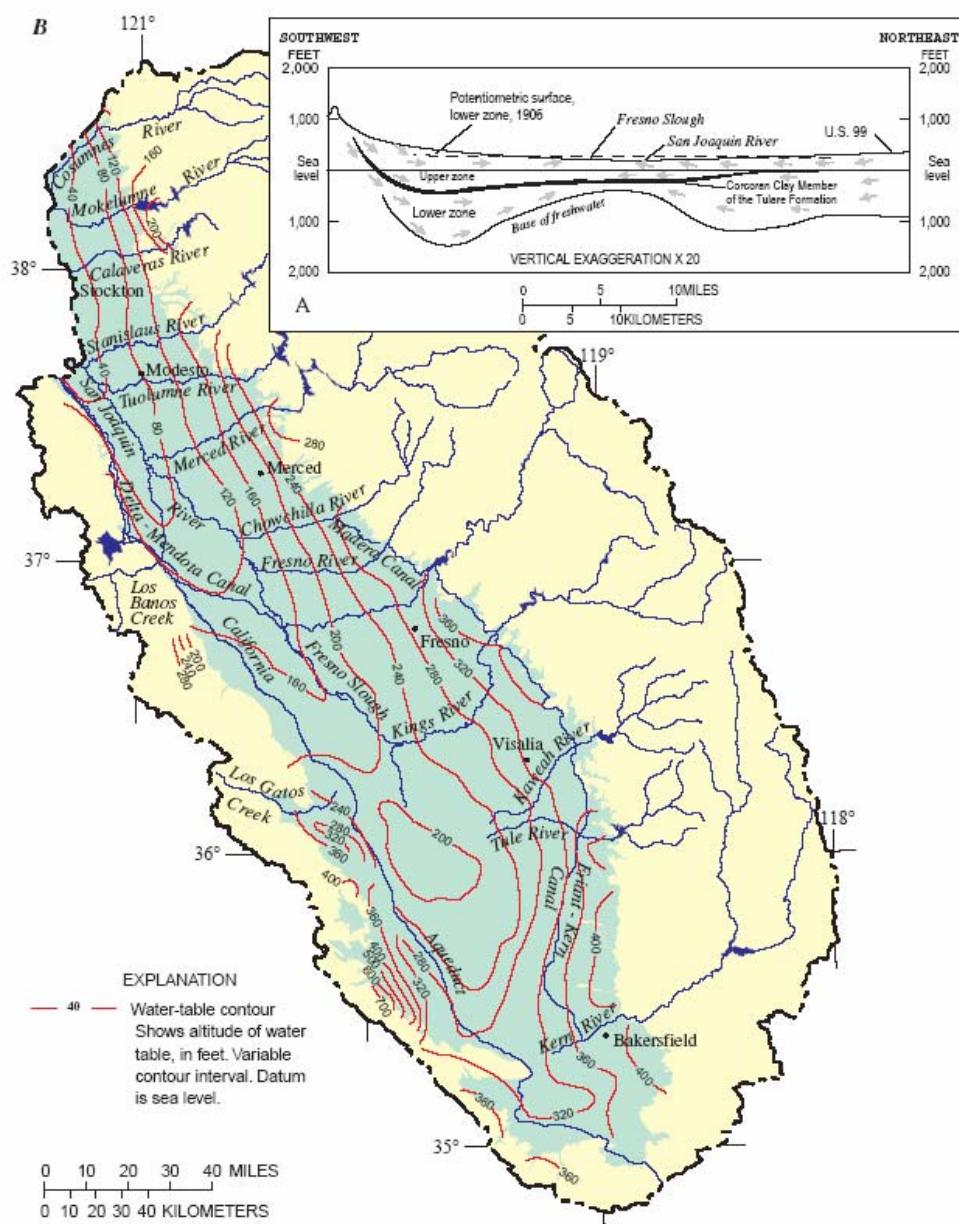


Figure 16. (A) Predevelopment ground-water flow in the San Joaquin Valley, California, about 1900 (Bertoldi and others, 1991). (B) Estimated predevelopment water table of the San Joaquin Valley, California (modified from Williamson and others, 1989).

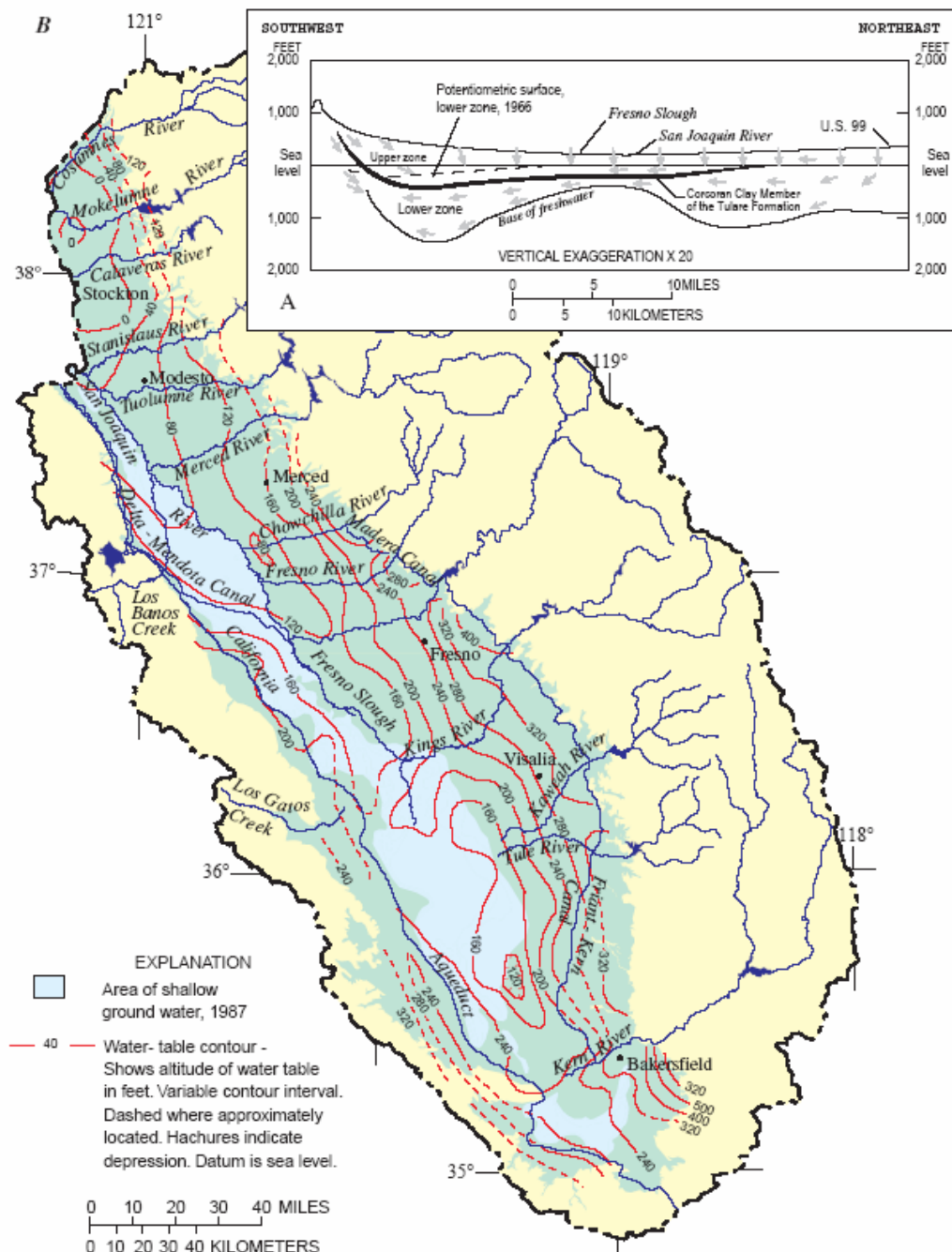


Figure 17. (A) Ground-water flow conditions in the San Joaquin Valley, California, 1966 (Bertoldi and others, 1991). (B) Water table in 1976 (modified from Williamson and others, 1989) and area of shallow groundwater in 1987, San Joaquin Valley, California (San Joaquin Valley Drainage Program, 1990b).

FACTORS AFFECTING WATER QUALITY

Many factors influence the quality of ground and surface waters in the San Joaquin–Tulare Basins study unit, but the predominant factors are climate (especially the distribution of precipitation), bedrock geology and chemistry of soils derived from bedrock, and land and water use. The western part of the study unit is

in the rain shadow of the Coast Ranges, and most precipitation falls on the eastern part of the study unit in the Sierra Nevada. The bedrock of the Sierra Nevada is primarily granitic. Because of the low solubility of the quartz and feldspars that make up the bulk of these rocks and soils, runoff and snowmelt from the Sierra Nevada have low concentrations of dissolved solids.

In contrast, rocks and sediments of the Coast Ranges in the western part of the study unit 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 compounds and trace elements. The precipitation in the Coast Ranges dissolves these constituents, and the resulting runoff has elevated concentrations of dissolved solids and other contaminants. Water quality may be further degraded by evaporative concentration because of the semiarid to arid climate. Runoff from the Coast Ranges is sparse, however; most of the surface water entering the hydrologic system in the study unit is from the Sierra Nevada and, therefore, generally low in dissolved solids.

Surface water in the Sierra Nevada can be degraded by land-use practices such as mining and logging. Mining can contribute dissolved solids, metals, and sediment to streams; logging can result in leaching of nutrients and increased sediment loads to streams. At present, runoff from the Sierra Nevada is low in dissolved solids until the water enters the San Joaquin Valley. Surface water imported to parts of the western and southern San Joaquin Valley also is generally low in dissolved solids; however, the concentration of dissolved solids in imported water is affected by hydrodynamic factors in the Sacramento–San Joaquin Delta.

Agricultural water use is the largest nonpoint source of water-quality degradation in the San Joaquin Valley. Irrigation has become the major source of recharge to the ground-water system, and can contain elevated concentrations of dissolved solids, nutrients, pesticides, and in some areas, trace elements. Surface water in the valley may be similarly impacted by direct runoff from irrigated areas (tailwater), by discharge of water from subsurface drainage systems that have been installed to control the level of the water table, and by discharge of poor-quality ground-water through river beds.

In addition to the effects of nonpoint agricultural sources, water quality in the valley is affected by the activities of the 2.7 million people who live in the San Joaquin–Tulare Basin study unit. Nonpoint sources of pathogens and nutrients from household septic systems degrade ground and surface waters. The effects of cities include point source discharge to rivers of nutrient-rich effluent from sewage treatment plants and stormwater runoff with elevated concentrations of metals. Volatile and other man-made organic compounds also can be introduced to both ground and surface water.

An additional factor in ground-water quality is the oxidation-reduction (redox) environment in the aquifers. The aquifers underlying the alluvial fans in the valley are largely oxidizing environments, whereas aquifers underlying the basin and lake deposit areas near the axis of the valley are reducing environments. In an aquifer the redox environment strongly influences the mobility of some trace elements and nitrate, and the potential for degradation of redox sensitive constituents, such as some organic compounds.

Water Use

California has the largest offstream water use in the nation, and consistently leads all states in surface and ground-water withdrawals (Templin, 1990, p. 173). Total water use in the study unit in 1990 was about 30.2 million acre-ft (W.E. Templin, U.S. Geological Survey, written commun., 1992). Hydropower production, the only instream water use studied under the USGS water-use program, accounted for 14.1 million acre-ft. Total offstream water use in the study unit was 16.1 million acre-ft, of which 91 percent was for irrigated agriculture.

Overall consumptive use of water in the study unit was about 12.1 million acre-ft. About 58 percent of this use was supplied with surface water and 42 percent with ground water. Of the surface-water use, about 38 percent (2.7 million acre-ft) was imported from the Sacramento–San Joaquin Delta through the California Aqueduct (State Water Project) and the Delta–Mendota Canal (Central Valley Project). About 22 percent of total consumptive use was supplied with imported water (California Department of Water Resources, 1994;

U.S. Bureau of Reclamation, 1990). Of this consumptive water use, 94.9 percent was for irrigated agriculture in 1990.

Domestic use (for example, drinking water) accounted for only 1.1 percent of the consumptive use in the study unit; virtually all of this was ground water. Figure 21 shows the distribution of total offstream water use and relative amounts supplied by ground water and surface water in 1990. The U.S. Geological Survey uses hydrologic units (fig. 21) to manage the National Water Data Network (U.S. Geological Survey, 1978).

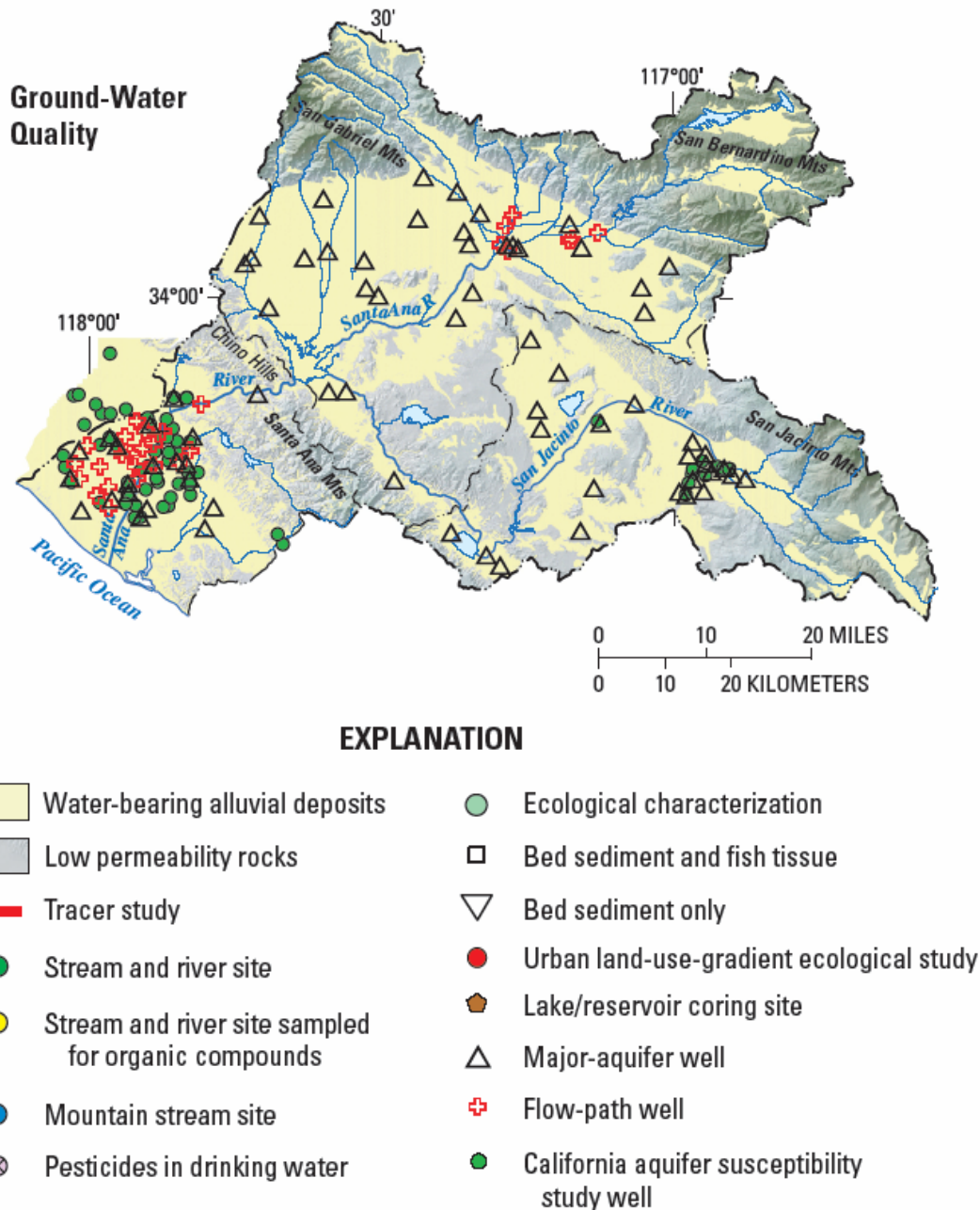
SOCA

Water Quality in the Santa Ana Basin, California, 1999–2001

By Kenneth Belitz, Scott N. Hamlin, Carmen A. Burton, Robert Kent, Ronald G. Fay, and Tyler Johnson

Circular 1238

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SOCA Ground-water quality			
Major aquifer survey	Major ions, nutrients, pesticides, volatile organic compounds, trace elements, stable isotopes, tritium, and radon analyzed to assess occurrence in the Coastal, Inland, and San Jacinto Basins.	Spatially distributed randomized deep public supply, irrigation, and domestic wells in the Coastal (20 wells), Inland (29 wells), and San Jacinto (23 wells) ground-water basins.	Coastal—spring 1999 Inland—Spring 2000 San Jacinto— — winter 2001
Urban land-use study	Same constituents as above, to assess shallow ground-water quality in the Coastal Basin.	Monitoring wells installed at 30 randomly chosen sites in commercial and residential areas.	Summer 2000
Flow-path study	Same constituents as for the major aquifer survey, to assess replacement of ground water by engineered recharge.	23 deep production wells. 20 deep monitoring wells and 1 production well.	Summer 2000

Ground-Water Quality

The quality of ground water in deep aquifers used for public supply in the Santa Ana Basin was assessed using nationally consistent approaches. Broad-scale assessments of major aquifers were conducted in the San Jacinto, Inland, and Coastal Basins. These assessments were supplemented by two flowpath studies that were designed to evaluate changes in water quality as water moves through the aquifer system. The flowpath studies, as well as additional sampling, were done in collaboration with the California State Water Resources Control Board as part of their California Aquifer Susceptibility program.

Ground-water pumping and artificial recharge have accelerated the movement of water through the aquifers of the Santa Ana Basin. To a large extent, native water in these aquifers has been replaced by water recharged since the early 1950s (p. 8). The quality of younger ground water (recharged since the early 1950s) is generally different from the quality of the older, native ground water. In the Inland and San Jacinto Basins, younger ground water tends to be shallower than older, native ground water. In the Coastal Basin, younger ground water tends to be closer to recharge facilities along the Santa Ana River and Santiago Creek.

- Ground water recharged since the 1950s has higher concentrations of nitrate and dissolved solids than older ground water (p. 8). Nitrate concentrations in younger water from aquifers that provide drinking-water supply exceed the USEPA drinking-water standard of 10 mg/L (as nitrogen) in some areas of the Inland and San Jacinto Basins. Dissolved-solids concentrations in younger water from aquifers used for public supply frequently exceed the USEPA nonenforceable drinking-water guideline (500 mg/L) in the Coastal Basin.

- o Pesticides and volatile organic compounds (VOCs) were detected more frequently in younger ground water than in older ground water (p. 14). While the concentrations of these compounds were generally low, the concentrations of three VOCs (PCE, TCE, and DBCP) exceeded USEPA drinking-water standards in four irrigation wells in the Inland Basin (p. 10).
- o Pesticides and VOCs were detected more frequently in the aquifers of the Santa Ana Basin than in aquifers assessed by NAWQA nationwide (p. 15). The higher detection frequencies are notable because the supply wells in the Santa Ana Basin are relatively deep, typically drilled to depths of 200 to 1,000 feet.

- o Nearly 80 percent of the public supply and irrigation wells sampled in the Santa Ana Basin had concentrations of radon higher than the drinking water standard of 300 picocuries per liter proposed by the USEPA (p. 16).

Trends in Ground-Water Quality

Major aquifer systems in the Santa Ana Basin are recharged by the diversion of the Santa Ana River and its tributaries to engineered recharge facilities. Water managers also use water imported from the Colorado River and northern California for recharge. Therefore, the quality of water in aquifers in the Santa Ana Basin reflects the quality of the surface water diverted during the past 50 years. Similarly, the future quality

of ground water will be affected by the quality of surface water currently being used to recharge these aquifers (p. 24).

The Santa Ana Basin, located in southern California, is characterized by prominent mountains that rise steeply from the relatively flat-lying coastal plain and inland valleys. The tallest peaks in the San Gabriel, San Bernardino, and San Jacinto Mountains exceed 10,000 feet in elevation. These mountain ranges are drained by the Santa Ana River and its tributaries. The drainage area of the Santa Ana River is about 2,700 square miles, making it the largest stream system in southern California.

The Santa Ana Basin includes parts of Orange, San Bernardino, Riverside, and Los Angeles Counties, and is home to nearly 5 million people who rely not only on water resources that originate within the basin but also on water imported from northern California and the Colorado River. The population is expected to increase to about 7 million people by 2025 and to 10 million by 2050 (Santa Ana Water Project Authority, 2003).

Land use in the basin is about 35 percent urban, 10 percent agricultural, and 55 percent undeveloped (fig. 1). Urban and agricultural land uses occur primarily in the relatively flat valleys and coastal plain. The mountains are generally steep and remain undeveloped. Population density for the entire study area is 1,500 people per square mile; excluding the land area that is steep, the population density is about 3,000 per square mile. The most densely populated part of the basin is in the city of Santa Ana, where the population density is as high as 20,000 per square mile.

The Santa Ana Basin can be subdivided into three primary subunits: the San Jacinto, the Inland, and the Coastal Basins (fig. 2). Within these subunits, water-bearing deposits lie within the alluvium-filled valleys and coastal plain, which are bounded by relatively impervious mountains and hills. Ground water pumped from the alluvial deposits is the primary source of municipal water supply. Public supply wells are typically drilled to depths ranging from 200 to 1,000 feet.

The hydrologic cycle in the Santa Ana Basin is dominated by human activities

The climate of the Santa Ana Basin is Mediterranean with hot, dry summers and cool, wet winters. Average annual precipitation ranges from 10 to 24 inches in the coastal plain and inland valleys and from 24 to 48 inches in the San Gabriel and San Bernardino Mountains. As a consequence of the semiarid climate and the water demands of the urban population, the hydrologic cycle is dominated by human activities. Streams and rivers draining the San Gabriel, San Bernardino, and San Jacinto Mountains are diverted to ground-water recharge facilities (fig. 2). In turn, pumped ground water is the major source of water supply in the watershed, providing about two-thirds of the total water demand (1.2 million acre-feet per year or about 1 billion gallons per day). Water imported from northern California and the Colorado River accounts for about one-quarter of the total consumptive demand. Imported water is delivered directly to consumers and is also used as a supplementary water source at ground-water recharge facilities.

From a mass-balance perspective, water is cycled twice in the Inland and Coastal Basins before it is discharged to the ocean. In the first cycle, tributaries exiting the San Gabriel and San Bernardino Mountains are diverted to ground-water recharge facilities in the Inland Basin. In turn, ground water is extracted for use and then discharged as treated wastewater to the Santa Ana River. In the second cycle, the Santa Ana River enters the Coastal Basin, where water managers utilize almost all of the base flow and most of the stormflow to recharge the coastal aquifer system. Base flow consists primarily of treated wastewater (Mendez and Belitz, 2002) and stormflow includes runoff from urban and agricultural land. Artificially recharged water accounts for about three-quarters of the ground water pumped from the coastal aquifer system (about 300,000 acre feet per year, or about 270 million gallons per day). Urban water use exceeds agricultural water use in the Study Unit (75 percent as compared to 25 percent). After delivery to consumers, the water is typically used for landscape irrigation and for indoor purposes. Water used for landscape irrigation can be used by plants, run off into storm drains and then streams, or recharge ground-water aquifers. Water used indoors is routed to wastewater-treatment facilities and then discharged, primarily to surface-water bodies: in the San Jacinto Basin, to constructed ponds; in the Inland Basin, to the Santa Ana River, its tributaries, and ponds; and in the Coastal Basin, to the ocean. In the Coastal Basin, reclaimed wastewater is also injected into aquifers along the coast as a barrier to seawater intrusion. During the period of this study (October 1998 to September 2001), precipitation was less than the 30-year average.

Streamflow in Warm Creek, a small urban watershed in the Inland Basin, was generally near historical lows during base-flow conditions; high flows occurred only during storms (fig. 3). In contrast, streamflow in the Santa Ana River was higher than the 30-year average (fig. 4). The high streamflow resulted from discharges of treated wastewater to the Santa Ana River (Burton and others, 1998). During the past 30 years, increasing urbanization has led to a steady increase in the volume of these discharges.

Water quality in the Santa Ana Basin reflects the influence of urbanization

The best quality water in the Santa Ana watershed occurs in streams flowing from surrounding mountains and in ground water recharged by those streams. As the water flows away from the mountains, either in the surface or subsurface, its quality is affected by mineral dissolution, urban runoff, discharge of treated wastewater, dairy operations in the Chino Dairy preserve (fig. 1), landscape irrigation, and the use of high-salinity imported surface water. Water quality can also be affected by the legacy of previous activities including spills and leaks of industrial solvents, and by agricultural production. The purpose of the NAWQA study in the Santa Ana Basin was to assess water quality in streams and aquifers. In this report, particular emphasis is placed on understanding the influence of urbanization on water quality. The Santa Ana Basin is the most densely populated of the NAWQA study areas and is among the more arid. The large population places a high demand on the limited water resources. Therefore, the Santa Ana Basin provides an ideal opportunity for assessing the effects of human activities on water quality.

Nitrate and dissolved- solids concentrations are higher in ground water recharged since the 1950s than in older ground water

In the Santa Ana Basin, ground water recharged since the early 1950s typically has higher concentrations of nitrate and dissolved solids than ground water recharged before the early 1950s. The water quality of younger ground water has generally been affected by activities such as agriculture, landscape irrigation, and wastewater generation and disposal. These activities typically increase the concentrations of nitrate and dissolved solids. As aquifers are recharged with water associated with these activities, the concentrations of nitrate and dissolved solids increase in ground water.

In the Santa Ana Basin, the concentration of nitrate in younger ground water typically ranges from 2 to 10 mg/L (fig. 7). In general, nitrate concentrations greater than 2 mg/L in ground water are indicative of contamination by human activities (Mueller and Helsel, 1996). In the San Jacinto and Coastal Basins, concentrations of nitrate in older ground water typically range from less than 0.05 mg/L to about 3 mg/L, with most less than 0.1 mg/L (fig. 7). In the Inland Basin, only three wells tap older ground water; the nitrate concentrations range from 0.8 to 8 mg/L. Of the 52 public supply and irrigation wells sampled in the San Jacinto and Inland Basins, 6 public supply wells and 3 irrigation wells produced water with nitrate concentrations that exceeded the USEPA drinking-water standard (10 mg/L as nitrogen). Ground water from eight of those nine wells is young. Water from the public supply wells with high nitrate concentrations is blended with other water sources with lower nitrate concentrations to comply with drinking-water standards. Concentrations of dissolved solids are generally higher in ground water recharged since the early 1950s than in older ground water in the Coastal and Inland Basins (fig. 8). In the Coastal Basin, the median concentration of dissolved solids in younger ground water exceeded 500 mg/L (the USEPA nonenforceable drinking-water guideline).

In contrast, the median concentration in older ground water was less than 300 mg/L. In the Inland Basin, the concentrations of dissolved solids in younger ground water ranged from 200 to 800 mg/L, whereas the samples of older water had concentrations less than 300 mg/L. The higher concentrations in younger ground water in both basins are primarily due to human activity, including the importation of high-salinity water from the Colorado River. The Colorado River has been an important source of engineered recharge in the Coastal Basin since the 1950s. In the San Jacinto Basin, the difference in the concentrations of dissolved solids between younger and older ground water is not as apparent as that observed in the two other ground-water basins. This difference is due to several factors including less human activity, higher concentrations of dissolved solids in the native ground water, and greater variability in concentrations in the native ground water. In many parts of the San Jacinto Basin, the native ground water has dissolved-solids concentrations exceeding 1,000 mg/L (Burton and others, 1996; Kaehler and others, 1998; Kaehler and Belitz, 2003); there are relatively few water-supply wells in these areas.

Pesticides and volatile organic compounds are more frequently detected in ground water recharged since the 1950s than in older ground water

In the Santa Ana Basin, pesticides and VOCs are detected more frequently in ground water recharged during the past 50 years than in older ground water (fig. 15). In the San Jacinto Basin, pesticides and VOCs were detected in more than 70 percent of the younger samples but were detected in only 20 percent of the older samples. In the Inland Basin, these compounds were detected in more than 80 percent of the younger samples and in only one of the three older samples. In the Coastal Basin, pesticides were detected in almost 40 percent of the younger samples but in none of the older samples; VOCs were detected in more than 90 percent of the younger samples but in only 50 percent of the older samples.

The age of ground water reflects proximity to sources of recharge. In the San Jacinto and Inland Basins, shallower ground water is younger than deeper ground water, indicating the predominance of recharge from the overlying land surface rather than by lateral flow from more distant areas. In the Coastal Basin, ground water in the unconfined area, located closer to the Santa Ana Mountains, is younger than ground water in the confined area, located farther from the mountains. The age distribution is consistent with the dynamics of the flow system: recharge in the unconfined area and lateral flow toward the confined areas. The detection frequencies of pesticides and VOCs reflect these age distributions. In the San Jacinto Basin, pesticides and VOCs were detected more frequently in water from shallower wells than in water from deeper wells (fig. 16A). In this basin, wells are typically drilled to depths ranging from about 300 to more than 1,500 feet. The top one-third of a typical well is solid casing, and the bottom two-thirds is usually screened (open to the aquifer). Pesticides and VOCs were detected more frequently in wells in which the top of the well screen is within 250 feet of land surface than in deeper wells (fig. 16A). The differences in detection frequency based on depth are comparable to the differences based on age. In the Inland Basin, as in the San Jacinto Basin, pesticides and VOCs were detected more frequently in the shallower wells (fig. 16B). These compounds were detected in all wells (100 percent) in which the top of the screen is within 150 feet of land surface, but in less than two-thirds of the wells in which the top of the screen is more than 350 feet deep. As in the San Jacinto Basin, the differences in detection frequency are consistent with recharge from the overlying land surface. The detection frequencies for pesticides and VOCs were higher in the Inland Basin than in the San Jacinto Basin and reflect more intensive land use, more pumping of ground water, and higher rates of recharge in the Inland Basin. In addition, the Inland Basin has been developed for a longer period of time.

In the Coastal Basin, pesticides and VOCs were detected more frequently in ground water in the unconfined area than in the confined area (fig. 16C). The differences in detection frequencies based on location are similar to the differences based on age, and reflect the fact that ground water is recharged in the unconfined area and moves laterally into the confined area.

Today's surface water is tomorrow's ground water

In many parts of the Nation, an important source of surface water is ground water. In the Santa Ana Basin and elsewhere in the Western United States, the converse is often true: surface water is the primary source of ground-water recharge. For example, in the Santa Ana Basin, ground-water recharge facilities are located along rivers and streams draining the San Gabriel, San Bernardino, and San Jacinto Mountains. In addition, large-scale recharge facilities are located along and adjacent to the Santa Ana River in the Coastal Basin. Recharge facilities throughout the Santa Ana Basin are used to enhance replenishment of aquifers used for public supply. At the large-scale recharge facilities in the Coastal Basin, water managers capture almost all of the base flow and most of the stormflow of the Santa Ana River. This engineered or artificial recharge, averaging about 225,000 acre-feet per year (about 200 million gallons per day), helps balance the total amount of ground water pumped from the aquifer system and used for public supply. These facilities have been used since the late 1940s. The distribution of anthropogenic compounds in ground water in the Coastal Basin illustrates the extent to which native ground water has been replaced by artificial recharge (fig. 26).

Tritium, an isotope of hydrogen that is incorporated into the water molecule, is an indicator of ground water recharged since the early 1950s (see sidebar, p. 9). Tritium is widespread in the aquifer system, indicating widespread replacement of older, native ground water with water recharged during the past 50 years.

Chloroform, a byproduct of water disinfection, is also widely distributed in the aquifer system, although it is not as widespread as is tritium. MTBE, a compound added to gasoline to reduce air pollution, has been used extensively since the early 1990s (Squillace and Moran, 2000). MTBE is not as widespread in the aquifer system as is chloroform, which has been generated for a longer period of time. The concentrations of chloroform and MTBE are below USEPA drinking-water standards. These results indicate that yesterday's surface water is today's ground water and therefore remind us of the importance of protecting surface-water quality. Protection of today's surface water will ensure the future quality of the ground-water resource.

Coastal Basin illustrates the extent to which native ground water has been replaced by artificial recharge (fig. 26). Tritium, an isotope of hydrogen that is incorporated into the water molecule, is an indicator of ground water recharged since the early 1950s (see sidebar, p. 9). Tritium is widespread in the aquifer system, indicating widespread replacement of older, native ground water with water recharged during the past 50 years. Chloroform, a byproduct of water disinfection, is also widely distributed in the aquifer system, although it is not as widespread as is tritium. MTBE, a compound added to gasoline to reduce air pollution, has been used extensively since the early 1990s (Squillace and Moran, 2000). MTBE is not as widespread in the aquifer system as is chloroform, which has been generated for a longer period of time. The concentrations of chloroform and MTBE are below USEPA drinking-water standards. These results indicate that yesterday's surface water is today's ground water and therefore remind us of the importance of protecting surface-water quality. Protection of today's surface water will ensure the future quality of the ground-water resource.

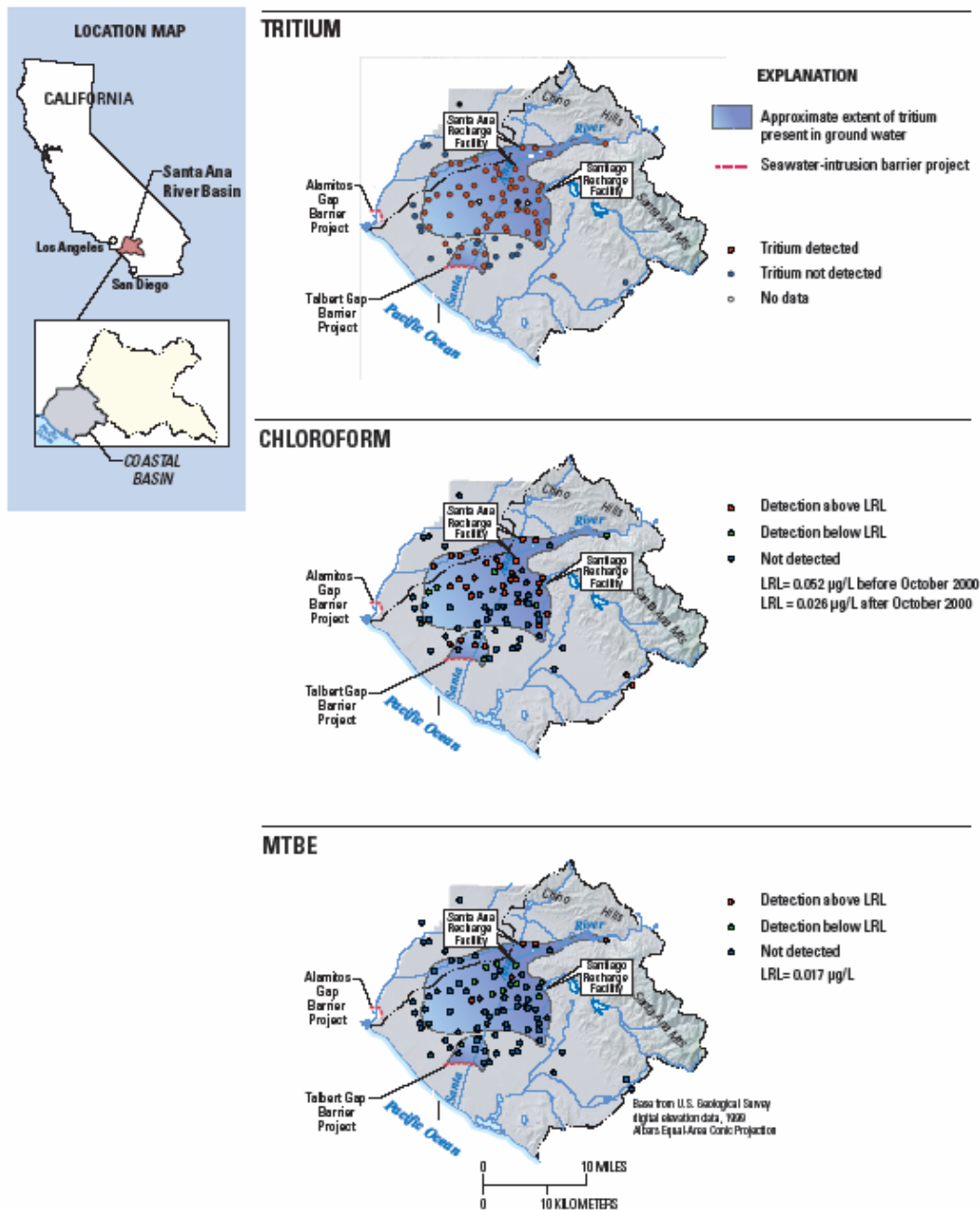


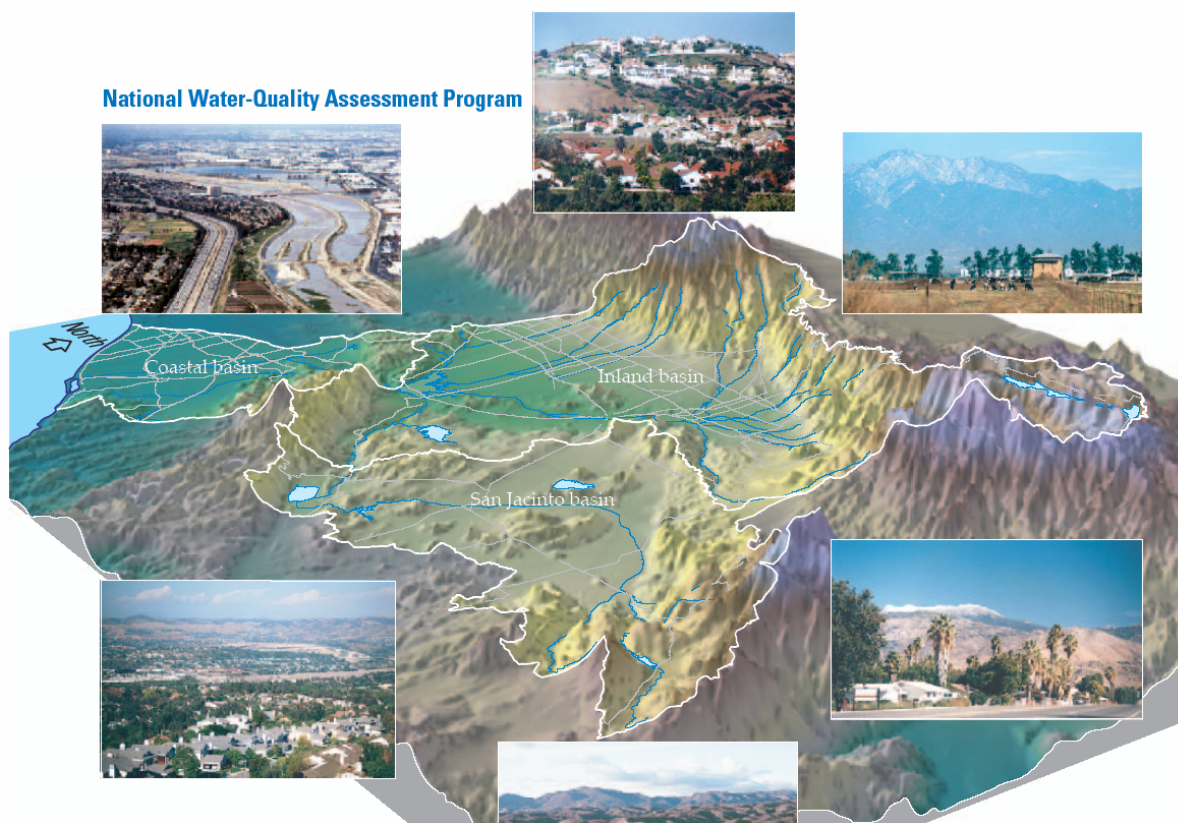
Figure 26. The areal distribution of anthropogenic compounds in the Coastal ground-water basin illustrates the displacement of native ground water by artificial recharge. (LRL, laboratory reporting level)

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

By Scott N. Hamlin, Kenneth Belitz, and Tyler Johnson

<http://water.usgs.gov/pubs/sir/2005/5032/>

Scientific Investigations Report 2005-5032



Description of Study Area

The Santa Ana Basin is located in southern California between Los Angeles and San Diego (*fig. 1*). The 2,700-mi² watershed is home to nearly 5 million people, and the population is expected to increase by more than 50 percent by the year 2020. During the same period, water demand is expected to increase by somewhat less than 50 percent (Santa Ana Watershed Project Authority [SAWPA], 1998). The Santa Ana Basin includes parts of Orange, San Bernardino, Riverside, and Los Angeles Counties. Population density for the entire study area is 1,500 people per mi²; excluding areas too steep to develop, the population density is about 3,000 people per mi². In the city of Santa Ana, the population density is as high as 20,000 people per mi² (California Department of Finance, accessed March 3, 2005). The Santa Ana River is the largest stream system in southern California, beginning in the San Bernardino Mountains (which reach altitudes exceeding 10,000 ft above sea level [NAVD, 1988]) and flowing more than 100 mi to the Pacific Ocean near Huntington Beach. The climate is Mediterranean with hot, dry summers and cool, wet winters.

Average annual rainfall ranges from 12 in. in the coastal plain and 18 in. in the inland valley to 40 in. in the San Bernardino Mountains. Ground water is the main source of supply in the watershed, providing about two-thirds of the total water used (about 1.2 million acre-ft/yr). Imported water from northern California and the Colorado River accounts for about one-quarter of the total consumptive demand. Local surface water provides the remaining supply. Urban water use (63 percent) exceeds agricultural water use (28 percent of total use) in the study area (Hamlin and others, 1999). The Santa Ana Basin can be subdivided into three primary ground-water basins: the San Jacinto Basin, the Inland Basin, and the Coastal Basin (*fig. 1*). Relatively impervious hills and mountains bound water-bearing deposits in the alluvium-filled basins. Urban and agricultural land uses occur primarily in the alluvium-filled valleys and the coastal plain. Land use in the watershed is about 35 percent urban; 10 percent agricultural; and 55 percent open space, primarily steep mountain slopes (Belitz and others, 2004).

Description and Comparison of Explanatory Factors in the Ground-Water Basins

The Santa Ana Basin includes three ground-water basins: the San Jacinto, Inland, and Coastal Basins. In this section, the hydrogeology of each of the basins is briefly described. In addition, the basins are compared with one another with respect to selected potential explanatory factors. Selected explanatory factors are listed by ground-water basin and compiled in the appendix.

Figure 2 shows the location of the San Jacinto Basin. The aquifers in this basin are generally unconfined and consist of a series of interconnected alluvium-filled valleys bounded by steep-sided bedrock mountains and hills. The thickness of deposits in these valleys typically ranges from 200 to 1,000 ft (Eastern Municipal Water District, 2002). Collectively, alluvium covers about one-half of the total area in the subunit. Prior to development, recharge to the local aquifers was from infiltration of mountain streams, primarily the San Jacinto River. Presently, sources of recharge include irrigation return flows, engineered (artificial) recharge facilities (*fig. 2*), and infiltration from streams. Water levels in sampled wells ranged from 90 to 540 ft below land surface, with an average depth of 230 ft. The basin is virtually closed; ground-water discharge occurs primarily by ground-water pumping.

Figure 3 shows the location of the Inland Basin. The aquifers of this basin generally are unconfined and comprise several subbasins filled with alluvial deposits eroded from the surrounding mountains. The thickness of these deposits ranges from less than 200 to more than 1,000 ft (Dutcher and Garrett, 1963). Recharge to the basin varies seasonally and is largely from infiltration of runoff from the San Gabriel and San Bernardino Mountains. Much of the runoff is diverted into storm detention basins, which also operate as ground-water recharge facilities (*fig. 3*).

Surface water imported from northern California and the Colorado River is also used to recharge the ground-water basin. Incidental recharge occurs from landscape irrigation. Water levels in sampled wells ranged from 20 to 560 ft below land surface, with an average depth of 200 ft. Ground-water discharge occurs primarily by pumping for public supply (*fig. 3*). Several VOC plumes in the basin define ground-water flow paths.

The Coastal Basin is subdivided into the Main and Irvine Basins (*fig. 4*; Singer, 1973). The Main Basin was divided, on the basis of relative abundance of shallow clay layers, into unconfined (forebay) and confined

(pressure) areas (California Department of Public Works, 1934). The unconfined area is small (28 percent) compared with the confined area (72 percent) (*fig. 4*). Within this basin, the thickness of freshwater-bearing deposits is as great as 4,000 ft (California Department of Water Resources, 1967).

Almost all recharge in the basin is from engineered facilities located within and along the Santa Ana River and along Santiago Creek (Herndon and others, 1997). Ground-water discharge occurs primarily by pumping for public supply. Water levels in sampled wells ranged from 20 to 230 ft below land surface, with an average depth of 100 ft. VOCs and pesticides were detected more frequently in younger ground water (recharged since the early 1950s) than in older ground water. In the three ground-water basins, VOCs were detected in 88 percent of the younger samples and in 36 percent of the older samples. Pesticides were detected in 72 percent of the younger samples and in 14 percent of the older samples.

Data from wells sampled for the three subunit surveys were used to compare one basin with another. Depth to the top of the well screen can be an important explanatory factor for the occurrence of VOCs and pesticides, particularly in unconfined aquifers. The median depth to the top of the well screen is 270 ft in the San Jacinto Basin, 240 ft in the Inland Basin, and 388 ft in the Coastal Basin (*fig. 5*). The depths to the tops of the well screens are significantly deeper in the Coastal Basin than in either the San Jacinto Basin or Inland Basin (*table 1; fig. 5*). Total well depths range from 328 to 1720 ft in the San Jacinto Basin, from 225 to 1,180 ft in the Inland Basin, and from 98 to 1,550 ft in the Coastal Basin. Median total depths of sampled wells are 696 ft in the San Jacinto Basin, 585 ft in the Inland Basin, and 910 ft in the Coastal Basin.

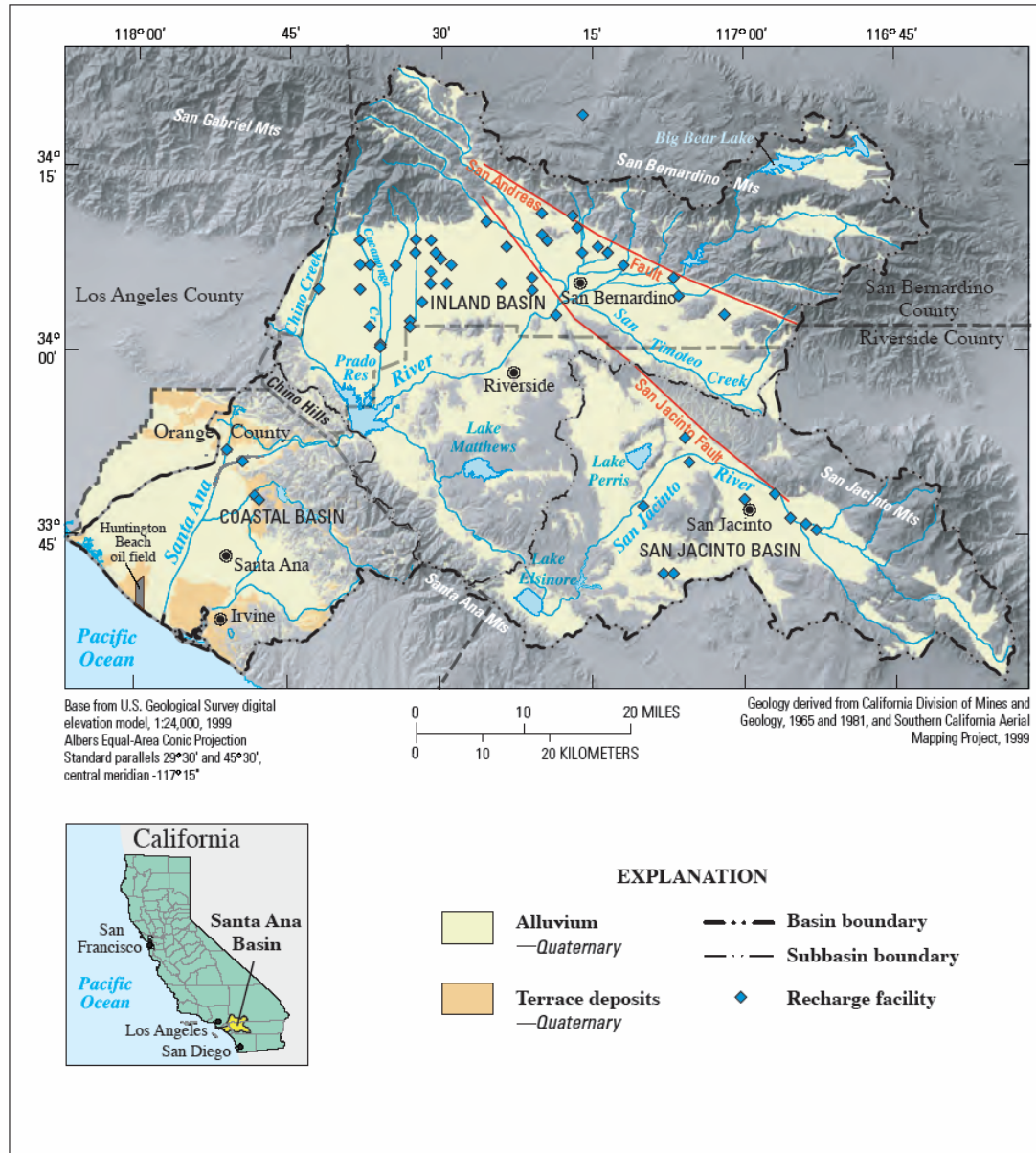


Figure 1. Location of the study area and the distribution of alluvial deposits in the Santa Ana Basin, southern California.

Summary and Conclusions

The Santa Ana Basin can be subdivided into three primary ground-water basins: the San Jacinto, Inland, and Coastal Basins. Aquifers in the San Jacinto Basin are unconfined and land use in the basin is primarily agricultural (37 percent), with lesser amounts of urban (33 percent) and undeveloped (25 percent) land use. Population density in the basin increased from 188 people per km² in 1970 to 558 people per km² in 2000. Aquifers in the Inland Basin are also unconfined and land use in the basin is primarily urban (58 percent), with lesser amounts of agricultural (13 percent) and undeveloped land (28 percent). Population density in the basin increased from 925 people per km² in 1970 to 1,620 people per km² in 2000. The Coastal Basin includes a relatively small unconfined area (28 percent) and a relatively large confined area (72 percent).

Recharge is primarily from engineered facilities in the unconfined area. The Coastal Basin is the most urbanized (80 percent) of the three ground-water basins. Population density in the basin increased from 1,800 people per km² in 1970 to 3,160 people per km² in 2000. Data from two different types of studies

were used in this report. Subunit surveys (SUS) were designed to obtain a statistical characterization of the basins as a whole. Data from special studies (flow path and aquifer susceptibility) were used, along with SUS data, to evaluate explanatory factors affecting water quality within individual basins. The wells sampled in these studies were primarily deep production wells, mostly used for public supply. The total depths of these wells ranged from 98 to 1,720 ft. The depths to the tops of the well screens ranged from 26 to 818 ft. Land use near the wells (within 500 m) was more urban than land use in the basins as a whole; public-supply wells tend to be located in urban areas in proximity to highest demand.

The most frequently detected VOCs in the San Jacinto, Inland, and Coastal Basins (occurring in more than 20 percent of the wells in any one basin) were disinfection byproducts (chloroform and bromodichloromethane), solvents (PCE, TCE, DCE, *cis*-1,2-dichloroethene, TCA, and 1,1-dichloroethane), refrigerants (CFC-113 and CFC-11), and the gasoline additive MTBE. The most frequently detected pesticide compounds were the herbicides atrazine and simazine and the atrazine degradation products deethylatrazine, deisopropylatrazine, and deethyldeisopropylatrazine. The occurrence and distribution of VOCs and pesticides in the Santa Ana ground-water basins were evaluated in relation to two types of explanatory factors: hydrogeologic characteristics and land use. Hydrogeologic characteristics evaluated were setting (unconfined or confined), ground-water age, depth to the top of well perforations, and proximity to engineered recharge. Land-use factors evaluated were the percentage of urban and agricultural land and population density. Population density can be considered a surrogate for the relative degree of urbanization.

Anthropogenic compounds were detected more frequently in unconfined aquifers than in confined aquifers in the Santa Ana ground-water basins. In the mostly unconfined Inland and San Jacinto Basins and in the unconfined part of the Coastal Basin, tritium was detected in 90, 78, and 97 percent of the wells sampled, respectively. In contrast, tritium was detected in 59 percent of the wells sampled in the confined area of the Coastal Basin. Similarly, VOCs were detected in more than 70 percent of the wells sampled in the unconfined areas, but in only 59 percent of the wells sampled in the confined area. Pesticides were detected in 83 and 48 percent of the wells sampled in the unconfined Inland and San Jacinto Basins, respectively. In the unconfined part of the Coastal Basin, pesticides were detected in 57 percent of the wells sampled, but in only 26 percent of the wells sampled in the confined area.

VOCs and pesticides were detected more frequently in younger ground water (recharged since the early 1950s) than older ground water. In the three ground-water basins, VOCs were detected in 88 percent of the younger samples and in 36 percent of the older samples. Pesticides were detected in 72 percent of the younger samples and in 14 percent of the older samples. In addition, more compounds (both VOCs and pesticides) were detected in younger samples. Water from shallow wells and wells proximal to recharge facilities is generally younger than water from deep wells and wells distal from recharge facilities.

VOCs and pesticides were detected more frequently in shallower wells than in deeper wells in the unconfined San Jacinto and Inland Basins, and in the unconfined area of the Coastal Basin. In addition, the number of compounds detected per well was inversely correlated with depth to the top of the well screen. These correlations reflect the presence of younger ground water in shallower wells as compared to deeper wells, and proximity to sources of contamination from the overlying landscape. VOC and pesticide occurrence were not correlated with depth to the top of the well screen in confined aquifers of the Coastal Basin; the lack of correlation reflects the dominance of lateral flow and insulation from the overlying landscape.

Proximity to engineered recharge facilities is an important explanatory factor for the distribution of anthropogenic compounds in the Coastal Basin and in the Hemet area of the San Jacinto Basin. VOCs and pesticides were detected more frequently in wells near engineered recharge facilities than in more distal wells. In addition, the number of VOCs and pesticides detected per well was significantly higher in proximal wells than distal wells in the Coastal Basin. In the Hemet area, the number of pesticides detected per well was significantly higher in proximal wells. In the Coastal Basin, anthropogenic compounds (tritium, VOCs, and pesticides) are present in ground water as much as 16 mi (25 km) from the recharge facilities, illustrating the large-scale displacement of native water by water recharged since the early 1950s.

On a national basis, land use within 500 m of shallow wells was an explanatory factor for the occurrence of VOCs and pesticides. A national evaluation of NAWQA data found that urban land use was positively

correlated with VOC and pesticide occurrence. Similarly, on a national basis, population density showed a positive correlation with VOCs and pesticides, and agricultural land use was positively correlated with pesticide occurrence. In contrast, land use within 500 m of wells sampled in the Santa Ana ground-water basins was not an explanatory factor for the occurrence of VOCs and pesticides.

The absence of correlation between land use near wells (urban, agricultural, and population density) and the occurrence of VOCs and pesticides in the Santa Ana ground-water basins is due to a number of factors. In the Coastal Basin, most of the wells sampled are relatively deep and insulated from the overlying landscape by a thick confining layer. In the unconfined Inland and San Jacinto Basins, the absence of correlation is likely due to the large depth of the wells sampled; the capture zones for these deep wells likely extend beyond 500 m. As evidence, there are solvent plumes in the Inland Basin that extend up to 10 km from the points of entry into the system. Another explanation for the lack of correlation between land use and the occurrence of VOCs and pesticides is that the current land use commonly differs from land use at the time of recharge; in the unconfined San Jacinto and Inland basins, population density has increased substantially since 1970.

Qualitatively, there was a relationship between basinscale land use and basin-scale detection frequencies of VOCs and pesticides in the unconfined aquifers of the Santa Ana ground-water basins. Urban land uses and population density are higher in the Inland Basin than in the San Jacinto Basin, and VOCs and pesticides were detected more frequently in the Inland Basin. In addition, urbanization in the Inland Basin has led to more intense use of ground water and increased transport rates of these compounds.

The distribution of anthropogenic compounds in the Santa Ana ground-water basins illustrates the extensive influence of human activities on water quality. These compounds are widely distributed in ground water in the Santa Ana Basin, both laterally and vertically. Engineered recharge, irrigation return flow (both agricultural and landscape), and intensive municipal pumping have accelerated the movement of ground water, and have widely distributed anthropogenic compounds in the aquifer systems. The distribution of anthropogenic compounds, generally in trace amounts, refines our understanding of the aquifer systems; this knowledge can be used to better manage the ground-water resource.

Low-Level Volatile Organic Compounds in Active Public Supply Wells as Ground-Water Tracers in the Los Angeles Physiographic Basin, California, 2000

By Jennifer L. Shelton, Karen R. Burow, Kenneth Belitz, Neil M. Dubrovsky, Michael Land, and JoAnn Gronberg

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HYDROGEOLOGIC SETTING

The study area covers the Los Angeles physiographic basin (fig. 1A), a 2,700 km² orthwest- trending alluviated lowland plain, which is bounded on the north, east, and southeast by mountains and hills, and is sometimes referred to as the coastal plain (Yerkes and others, 1965). Ground-water flow is largely controlled by engineered recharge along the San Gabriel, the Rio Hondo, and the Santa Ana Rivers, and by ground-water pumping from the many hundreds of wells distributed across the area. The direction of ground-water flow is primarily lateral and radial from the recharge facilities toward the coast (fig. 1B). Along the coast near the seawater intrusion barriers, the direction of ground-water flow is inland owing to pumping. The Los Angeles physiographic basin is subdivided into two basins determined by the sources of recharge water—the Coastal Los Angeles Basin and the Coastal Santa Ana Basin. These basins have historically been delineated by the Los Angeles County–Orange County line (fig. 1A) (Poland and others, 1956; California Department of Water Resources, 1961). Although the hydrogeologic setting is similar between the Coastal Los Angeles Basin and the Coastal Santa Ana Basin, each of these basins has distinct radial flow paths originating from the engineered recharge facilities.

Each basin can be further subdivided into two areas—forebay and pressure areas—based on the texture and permeability of aquifer materials (Eckis, 1934; California Department of Water Resources, 1961). The forebay areas are in the northeastern part of the Los Angeles physiographic basin along and adjacent to the rivers (fig. 1A). The sediments in the forebay areas consist of unconsolidated, interbedded sands and gravels with occasional lenses of silt and clay material. The relatively small amount of fine-grained sediments in the forebay areas are laterally discontinuous and generally do not impede the vertical movement of ground water. The forebay areas are the primary areas in which groundwater recharge occurs. The pressure area extends from the southwestern edge of the forebay to the Pacific Ocean; it occupies most of the land area in the basin and is characterized by sand, gravel, and thick lenses of silt and clay. The laterally extensive and thick lenses of fine-grained sediments in the pressure area can impede vertical movement of ground water.

Coastal Los Angeles Basin

The hydrogeology of the Coastal Los Angeles Basin was characterized using the extensive amount of hydrologic and geologic data collected as part of an ongoing study between the USGS and the Water Replenishment District of Southern California (WRD). The simplified cross section shown in figure 1B illustrates the regional flow path in the study area. The Coastal Los Angeles Basin occupies the northern half of the study area (fig. 1A). The fresh-water-bearing deposits are up to 900-m thick and consist of gravel, sand, silt, and clay of fluvial and marine origin. Depth to water ranges from about 100 m to near land surface adjacent to the ground-water recharge facilities (Water Replenishment District of Southern California, 2001). The Coastal Los Angeles Basin is composed of several smaller basins, the two largest of which are the Central Basin and West Coast Basin. The Central and West Coast Basins are delineated by the Newport–Inglewood Uplift (fig. 1A), the Pacific Ocean, and by surrounding hills. Ground water flows primarily laterally and radially from the recharge facilities in the Central Basin and eastward from the Pacific Ocean in the West Coast Basin (Water Replenishment District of Southern California, 2001). The primary sources of engineered recharge water differs in these basins: the Central Basin receives recharge water from the groundwater recharge facilities in the forebay areas, and the West Coast Basin receives recharge water from injection wells that form the seawater intrusion barriers (fig. 1A) (Water Replenishment District of Southern California, accessed June 13, 2001).

The forebay areas in the Coastal Los Angeles Basin are in the Central Basin and can be subdivided into two areas: the Los Angeles forebay and the Montebello forebay (fig. 1A). The Los Angeles forebay is located along and adjacent to the Los Angeles River as it flows from the San Fernando Valley. Intense urban development and lining of the Los Angeles River has greatly limited the amount of natural ground-water recharge in this western forebay area. The Montebello forebay is located along and adjacent to the Rio Hondo and the San Gabriel Rivers downgradient from the Whittier Narrows Flood Basin in the San Gabriel Valley (fig. 1A). Recharge facilities in the Montebello forebay receive a combination of water from the San Gabriel Valley, treated wastewater, and imported water, and serve as the primary source of recharge to the Coastal Los Angeles Basin. The large Whittier Narrows Flood Basin just north of the study area boundary in the San Gabriel Valley is not actively managed for ground-water recharge purposes. However, the flood basin receives storm-water runoff from the San Gabriel Valley, which can recharge the aquifer. Additional recharge occurs at a series of injection wells operated to restrict seawater intrusion along parts of the coast. Injection wells receive a combination of highly treated wastewater and imported water. Replenishment and management of the groundwater resources necessary to offset historic overdraft began during the early 1960s by the WRD, although artificial recharge of ground water has taken place in the Los Angeles area since about 1920 (Peterson, 1961; Water Replenishment District of Southern California, 2001). The primary sources of recharge are from engineered ground-water recharge, which includes water that is spread at the recharge facilities and injected into wells that form seawater intrusion barriers. For the 2000 water year (October 1999 through September 2000), about 0.227 km³ (184,300 acre-ft) of water was recharged. About 77 percent (0.174 km³ or 141,700 acre-ft) of the total was from engineered ground-water recharge (Water Replenishment District of Southern California, 2001): about 0.139 km³ (113,000 acre-ft) at recharge facilities in the forebay area, and about 0.035 km³ (28,700 acre-ft) at injection wells that form the seawater intrusion barriers.

Other sources of recharge include underflow from adjacent basins, recharge along the margins of mountains and hills, and precipitation. Outflow or ground-water production is from ground-water pumping primarily from the pressure area. About 0.310 km³ (251, 525 acre-ft) of ground water was pumped during

the 2000 water year, most of which was from the Central Basin (79 percent), and the remainder was from the West Coast Basin. The main production zones generally range from 120 to 370 m (400 to 1,200 ft) below land surface (Michael Land, U.S. Geological Survey, unpublished data, 2000).

Coastal Santa Ana Basin

The hydrogeology of the Coastal Santa Ana Basin was characterized using data collected by the Orange County Water District (OCWD) and the USGS's Santa Ana River Basin study team of the NAWQA Program. Similar to the flow path in the Coastal Los Angeles Basin, ground-water studies indicate that regional ground-water flow in the Santa Ana River Basin extends laterally and radially outward from the ground-water recharge facilities toward the coast (Orange County Water District, 1997). The Coastal Santa Ana Basin occupies the southern half of the study area.

The thickness of freshwater-bearing deposits is as much as 1,200 m (Orange County Water District, 1996). These deposits consist of alluvium derived from the mountains to the east and southeast and from marine deposits. Similar to the Coastal Los Angeles Basin, depth to water ranges from about 100 m below land surface to near land surface along the rivers, creeks, and ground-water recharge facilities (Orange County Water District, 1996). The forebay area occupies about 130 km² along and adjacent to the Santa Ana River after it leaves the Santa Ana Mountains (fig. 1A). Recharge facilities in the forebay area receive a combination of treated wastewater, Santa Ana River water, and imported water, providing the primary source of recharge to the ground-water basin.

Since the formation of OCWD in 1933, groundwater recharge and pumpage from public supply wells have been intensively managed to offset historic ground-water overdraft (Orange County Water District, accessed June 13, 2001). In 1999, 0.342 km³ (278,000 acre-ft) was recharged primarily at facilities along the Santa Ana River. Lesser amounts of recharge to the aquifer system occur along Santiago Creek from natural infiltration along the margins of the hills that surround the coastal plain and from vertical leakage through the confining layers. In 1999, about 0.438 km³ (356,000 acre-ft) was pumped from the aquifer system primarily in the confined pressure area of the basin.

The upper aquifer zone is generally from 90 to 450 m (300 to 1,500 ft) below land surface, with most of the pumpage from 150 to 300 m (500 to 1,000 ft) below land surface. In the confined pressure area, the upper aquifer zone is overlain by 90 to 150 m (300 to 500 ft) of deposits that consist primarily of silts and clays, which typically impede vertical movement of groundwater (Orange County Water District, 1996).

This lack of distinction between hydrogeologically vulnerable and other areas is evidence that the acceleration of the lateral rate of ground-water flow produced by the engineered recharge and ground-water pumping in the Coastal Los Angeles Basin and the Coastal Santa Ana Basin is a more important factor in the occurrence of VOCs in active public supply wells than the surficial features used for mapping vulnerability. This engineered acceleration implies that areas downgradient of the recharge facilities are susceptible to contamination by any contaminant that will persist for decades.

The forebay areas represent a more consistent designation of susceptibility in the study area, possibly because they more closely reflect the zones where ground water is more likely to migrate vertically compared with the pressure areas where clay layers are more extensive and vertical migration of ground water is impeded. Sixty-one wells sampled in this study are located within the forebay, and 79 percent have one or more detections above the LRL. The number of VOC detections above the LRL is significantly different in wells in the forebay areas and in close proximity to the recharge facilities than in wells in the pressure areas at greater distances from the recharge facilities ($p = 0.0008$, Kruskal-Wallis test), and the median number of VOC detections above the LRL is higher in samples from wells in the forebay areas (3) than in samples from wells in the pressure areas (1). This relation is slightly stronger when the West Coast Basin wells, which are hydrogeologically isolated from the forebay areas or recharge facilities, are removed from the data set ($p = 0.0005$, Kruskal-Wallis test). Only one of the wells outside the forebay area had a concentration of MTBE above the LRL, whereas 23 wells inside the forebay area had a MTBE concentration above the LRL (fig. 5A). Unlike MTBE, the concentrations of VOCs in use for decades, such as CFC-11, CFC-113, chloroform, PCE, and TCE, are not different between the forebay and the pressure areas, and the detection rates are similar.

The overall lack of correlation between population and LUFT density on the land surface and the occurrence and distribution of VOCs in active public supply wells in this study suggests that vertical migration of contaminants is not a dominant process outside the susceptible forebay areas, as evidenced by the higher number of VOC detections and the higher concentrations of MTBE in the forebay area.

Although VOCs are detected in wells outside the forebay areas, these wells are further from the source of recharge and reflect a longer time of travel, as evidenced by lower MTBE concentrations. While the forebay areas may be the most susceptible to current contamination, the presence of compounds in use for decades at similar concentrations both inside and outside the forebay areas indicate that the pressure areas are susceptible to downgradient migration of persistent contaminants recharged upgradient in the forebay areas.

VOC DISTRIBUTION INDICATES ENHANCED RECHARGE AND GROUND-WATER PUMPING ARE PRIMARY CONTROLS ON GROUND-WATER FLOW SYSTEM

As a simple approximation, the ground-water flow system can be conceptualized as a radial wedge or pie-slice with the apex at the recharge facilities in the forebay and flow paths extending outward toward the coast. Within the wedge is water bearing the fingerprint of urbanization—that is, water containing VOCs. Outside the wedge is native ground water. VOCs might be introduced in the coarse-grained forebay areas either at the recharge facilities or in other sources of recharge that encounter point or nonpoint contaminant sources. This radial pattern of flow is reflected in the distribution of VOCs (fig. 3), which act as tracers of postindustrial-aged water (water recharged since the onset of intense urban development). Inspection of water-level contour maps (Orange County Water District, 1996) also indicates flow originating at the engineered recharge facilities and radiating outward. However, because of changes in the source and quality of recharge water over time, ground-water quality in the flow system is not the same as the quality of water presently recharged at the forebay areas and the recharge facilities. This radial wedge concept of the ground-water flow system, however, does not describe the flow system west of the Newport–Inglewood Uplift in the West Coast Basin of the Coastal Los Angeles Basin. The Newport–Inglewood Uplift separates the Central Basin from the West Coast Basin and impedes lateral ground-water flow, thus the West Coast Basin receives more water from the seawater intrusion barriers than from recharge in the forebay areas (Water Replenishment District of Southern California, 2001).

The spatial distribution of the VOCs detected above the LRL can be quantified in terms of the distance between the recharge facility and the location of the well. This distance represents distance along the radial flow path and is a surrogate for the time of travel. The distance from the recharge facilities is negatively correlated with the number of VOCs detected above the LRL in the Central Basin of the Coastal Los Angeles Basin and in the Coastal Santa Ana Basin ($p < 0.001$ and $p = 0.002$, respectively; Spearman's rank correlation) (fig. 4). This analysis shows samples with the most VOC detections are from wells located within 15 km of the recharge facilities, with a few exceptions. The wells with VOCs detected above the LRL almost form a line at approximately 10–15 km of the recharge facilities beyond which few or no VOCs were detected (fig. 3). Thus, the low-level concentrations of VOCs seem to act as a reasonable groundwater tracer for postindustrial-aged water and indicate the distance water has traveled from where it was recharged in the forebay areas. The most notable exceptions to this generalization are the four wells in which more than five VOCs were detected outside the wedge of postindustrial-aged water: one is in the southern part of the Coastal Santa Ana Basin, one is in the West Coast Basin of the Coastal Los Angeles Basin, and two are in the extreme northwestern corner of the Coastal Los Angeles Basin (fig. 3). The VOCs in wells located in the older unconsolidated deposits and outside the radial flow paths are probably not derived from the water recharged in the forebays.

The injection wells that make up the seawater intrusion barriers contribute to water recharged in areas along the coast and may be a potential low-level source of VOCs to public supply wells in those areas. Although the recharge facilities along the Rio Hondo, the San Gabriel, and the Santa Ana Rivers are the major sources of recharge, they are not the only sources. The Los Angeles River, prior to being lined with concrete, and Santiago Creek are also sources of recharge water. In addition, there are a variety of potential point and nonpoint sources of VOCs in urban areas. The coarse-grained sediments and discontinuous silt and clay lenses in the forebay areas also allow vertical migration of precipitation, infiltration from excess landscape watering, storm-water runoff, chemical spills, leaky underground fuel tanks, and leaky sewer and water distribution lines to recharge the aquifer. Age dating from the tritium/helium-3 analyses may assist in the interpretation of potential sources of recharge water and VOCs. The use of VOCs as tracers of

postindustrial-aged ground water may be further examined by evaluating the most frequently detected VOCs with reference to when these VOCs were introduced into the environment. Figure 5 shows the distribution of sample concentrations of four of the most frequently detected VOC in each of four primary use categories (table 4). The spatial distribution of the fuel oxygenate MTBE, which has the most recent history of use, is shown in figure 5A. MTBE was used to enhance octane levels in gasoline during the late 1970s (Zogorski and others, 1996), and the use of MTBE increased significantly during the early 1990s after fuel oxygenates were mandated by the Clean Air Act amendments to reduce atmospheric concentrations of carbon monoxide. In 1996, about 10.6 billion liters (2.8 billion gallons) of MTBE were produced in the United States, over 95 percent of the gasoline used in California was blended with MTBE, and the annual production capacity of MTBE in California was about 719 million liters (190 million gallons) (Gomez and others, 1998). MTBE has the most localized distribution of detections with all but two of the detections at concentrations above the LRL in wells close to the recharge facilities in the forebay (fig. 5A). This nearly exclusive occurrence of MTBE close to the area of recharge is consistent with the relatively recent introduction of MTBE as a fuel additive, the concept of the radial-flow system, and the use of VOCs as ground-water tracers for postindustrial water.

Most of the MTBE detections and the highest MTBE concentrations are found in samples from wells close to the recharge facilities and the Rio Hondo or the San Gabriel Rivers in the Montebello forebay area in the Coastal Los Angeles Basin. The reason for this higher frequency of occurrence and concentration in the Coastal Los Angeles Basin than in the Coastal Santa Ana Basin is unknown. Perhaps it is due to inflow from adjacent basins or the greater amount of surface area of recharge facilities or unlined reaches of rivers in the Montebello forebay than in the Coastal Santa Ana Basin. Approximately 3.4 km² (850 acres) of recharge facilities or unlined reaches of the Rio Hondo and the San Gabriel Rivers are in the Montebello forebay area. The rapid infiltration of large volumes of surface water potentially could provide a more rapid way for MTBE to be transported to the aquifer.

TCE has been in use longer than MTBE and is more widely distributed. TCE has been produced in large quantities since the 1960s (Stackelberg and others, 2000) and is considered to be a high production volume chemical in that over 450,000 kg (1 million pounds) are produced annually in the United States (Environmental Defense, 2001a). In contrast with MTBE, TCE is detected farther from the recharge facilities and at about the same frequency in the forebay areas as it is detected in the pressure areas (fig. 5B). Because TCE has been in production longer than MTBE, TCE has traveled with recharge water a greater lateral distance from the forebay. Similar to MTBE, TCE is detected more frequently and occurs at higher concentrations in the Coastal Los Angeles Basin than in the Coastal Santa Ana Basin. However, unlike MTBE, some of the highest concentrations of TCE are isolated and are not in proximity to the recharge facilities (fig. 5B); these detections may be due to rapid vertical migration from a localized source of contamination.

The spatial distribution of the most frequently detected refrigerant, CFC-11, is shown in figure 5C. CFC-11 was frequently used in air conditioners and coolants during the 1970s and is also considered a high production volume chemical (Environmental Defense, 2001b). In July 1992, producers of CFC-11 were required to gradually reduce production, and production was to be completely phased out by January 1, 2000 (U.S. Environmental Protection Agency, 1999). This reduction may have decreased the amount of chlorofluorocarbon that enters the ground-water system and explain the lack of detections in the forebay areas and the higher frequency of detections in the pressure areas. Most of the detections of CFC-11 above the LRL (0.09 μ g/L) are from wells in the pressure areas.

Unlike the spatial distribution of MTBE and TCE, CFC-11 is more frequently detected and occurs at higher concentrations in the Coastal Santa Ana Basin than in the Coastal Los Angeles Basin (fig. 5C). The second most frequently detected refrigerant, CFC-113, has a similar pattern and spatial distribution. The reasons for the increased frequency of refrigerant detections in the Coastal Santa Ana Basin are unknown, but could potentially result from localized sources.

The distribution of chloroform (fig. 5D) is similar to the overall distribution of VOCs. Chloroform, often produced as a byproduct of the disinfection of drinking water, has been produced since the 1920s for many uses, including manufacturing of pharmaceuticals, dry cleaning, fire extinguishers, fumigants, and as an anesthetic (Stackelberg and others, 2000; U.S. Department of Health and Human Services, 2001). Most of

the chloroform detections are above the LRL ($0.024 \text{ } \mu\text{g/L}$) and are found in almost equal numbers in both the Coastal Los Angeles and the Coastal Santa Ana Basins. Sources of chloroform that would result in a wide distribution of this disinfection byproduct include chlorination of drinking water, leaks in drinking water distribution systems, lawn irrigation, and recharge of large volumes of treated wastewater and State Water Project water at recharge facilities. Chloroform detections close to the coast may be related to injection of treated wastewater into the wells that form the seawater intrusion barriers.

Degradation along ground-water flow paths could also result in lower concentrations of VOCs at greater distances from where ground water is recharged. Assuming the spatial distribution of VOCs is dominated by lateral flow from the forebay, then the current extent of any specific VOC is minimal because loss of VOC mass from sorption and degradation along the ground-water flow path have not been accounted for in this study.

An alternative to the radial flow concept of the flow system is that the VOC distribution could be controlled by vertical movement of ground water in proximity to a well. Given that VOCs are likely to enter the ground-water system at the water table, a reasonable expectation would be that the frequency and magnitude of VOC concentrations would decrease with increased depth of a public supply well. However, this relation was not observed in the data for the entire study area. The number of VOC detections above the LRL was not significantly different ($p = 0.72$; Kruskal-Wallis test) among samples from wells grouped into three depth categories—shallow, composite, and deep—and located throughout the entire study area. Conversely, the number of VOC detections above the LRL is significantly different ($p = 0.03$; Kruskal-Wallis test) among samples from wells grouped by the three depth categories for the 61 wells in the forebay areas (fig. 6), and the median of the number of VOC detections above the LRL is higher in the shallow wells (4) than the deep wells (1). The number of VOC detections above the LRL is not significantly different ($p = 0.27$; Kruskal-Wallis test) among samples from wells grouped by the three depth categories for the 117 wells in the pressure area. There was no difference in well depth between the forebay and pressure area. These results indicate that the greater thickness of clay layers and other fine-grained sediment in the pressure area possibly impede the downward migration of VOCs.

The differences in the number of detections of VOCs above the LRL among the three depth categories in the forebay areas is due solely to contrasts in VOC detections among the 31 wells in the forebay of the Coastal Santa Ana Basin ($p = 0.01$, Kruskal-Wallis test); there are no significant differences in the number of detections of VOCs above the LRL among depth categories in the forebay in the Coastal Los Angeles Basin ($p = 0.88$; Kruskal-Wallis test). The contrast in the number of VOC detections with depth in the Coastal Santa Ana Basin forebay indicates that there could be a vertical component of transport affecting the distribution in this area. Rapid vertical transport could result from several mechanisms; for example, a leaking well casing, inter-borehole flow, or local high vertical gradients from pumping. More data is necessary to determine the significance of these mechanisms and to distinguish the effects of lateral or vertical movement of ground water in the forebay.

The data indicate that VOC detections above the LRL are not uniformly distributed over the study area (fig. 3). Most of the wells with multiple VOC detections, which also have the highest concentrations, are in the forebay areas and clustered proximal to the recharge areas. In addition, there appears to be a distance of about 10–15 km from the recharge facilities beyond which the number of VOC detections above the LRL (figs. 3 and 4) and VOC concentrations (fig. 5) decrease. The pattern of VOC distribution is related to the history of use. For example, chloroform has the longest history of use and is the most widely distributed when compared with MTBE, which has the most recent history of use and is detected at concentrations above the LRL almost exclusively in the forebay areas.

This distribution and the history of introduction of the most frequently detected VOCs into the urban landscape indicate that engineered recharge and groundwater pumping are the dominant controls on the flow system; VOCs travel along radial flow paths to a distance that is indicative of the age of the recharge water or the time in which the VOC was introduced into the environment.

The data also suggests that downward vertical migration of VOCs from the water table may be a viable process for contaminating aquifers in the forebay areas but there is little evidence that the same process is active and affecting the occurrence of VOCs in many wells in the pressure areas. The significant difference

in the number of VOC detections above the LRL between different depth categories in the forebay areas suggests that vertical downward migration of VOCs is a potential pathway to the capture zone of public supply wells in the Coastal Santa Ana forebay.

The lack of contrast in the number of VOC detections between well depth categories basinwide and the nonuniform distribution of VOCs basinwide suggests that vertical migration is not the dominant pathway. The data are insufficient, however, to exclude the possibility that (1) VOCs traveling vertically from near-surface have not yet reached the wells or (2) VOCs are present but not detectable in the uppermost fraction of the well screens because of the mixing over the large lengths of the screened intervals. Additional data along the ground-water flow paths collected from discrete zones in the aquifer at various depths may aid in assessing the transport of VOCs to public supply wells.

STABLE ISOTOPE DATA RELATE SOURCE OF RECHARGE WATER TO VOC OCCURRENCE

The presence of recharge from precipitation; urban runoff; Santiago Creek; and the Santa Ana, the San Gabriel, and the Los Angeles Rivers can be partly discerned from analysis of stable isotope ratios in ground- and surface-water samples. Oxygen-18 (^{18}O) and deuterium (D) are naturally occurring stable isotopes of oxygen and hydrogen, respectively. The abundance of these isotopes in water is generally expressed as a ratio in delta ($^{\text{TM}}$) notation, and on a global scale the relation between $^{\text{TM}}18\text{O}$ and $^{\text{TM}}\text{D}$ is linear, plotting along the global meteoric water line (Craig, 1961).

Deviations from the global meteoric water line can occur as a consequence of several hydrologic processes, including evaporation and mixing. Figure 7 shows plots of the isotopic composition of water samples from active public supply wells in the Los Angeles physiographic basin. The samples are differentiated on the basis of the occurrence of VOCs: samples with no detections, samples with one or more detections above the LRL, samples with detections only below the LRL. Also shown in both figures is the global meteoric water line. In general, the groundwater samples plot along and to the right of (below) the global meteoric water line.

Figure 7A illustrates hypothetical mixing lines for mixtures of imported Colorado River water and southern California surface water. The isotopic composition of Colorado River water plots to the right of (below) the global meteoric water line owing to evaporation. The isotopic composition of southern California surface water plots over a range of values, but is consistently heavier than Colorado River water. For the purposes of illustration, three hypothetical southern California surface water end-members are considered that may represent the source of the ground-water sampled in the active public supply wells: one that is isotopically light (more negative), one that is isotopically heavy (less negative), and one that is intermediate. Each of the mixing lines illustrates the plotting positions of water samples that represent varying amounts of the Colorado River end-member and one of the southern California end-members. A sample with a relatively high proportion of Colorado River water would plot relatively close to the Colorado River endmember.

A sample with a relatively low proportion of Colorado River water would plot relatively close to the global meteoric water line. Overall, mixing of imported Colorado River water with southern California surface water results in a shift in the isotopic composition away from the global meteoric water line. Along any particular mixing line, the shift is downward and to the left towards the Colorado River endmember.

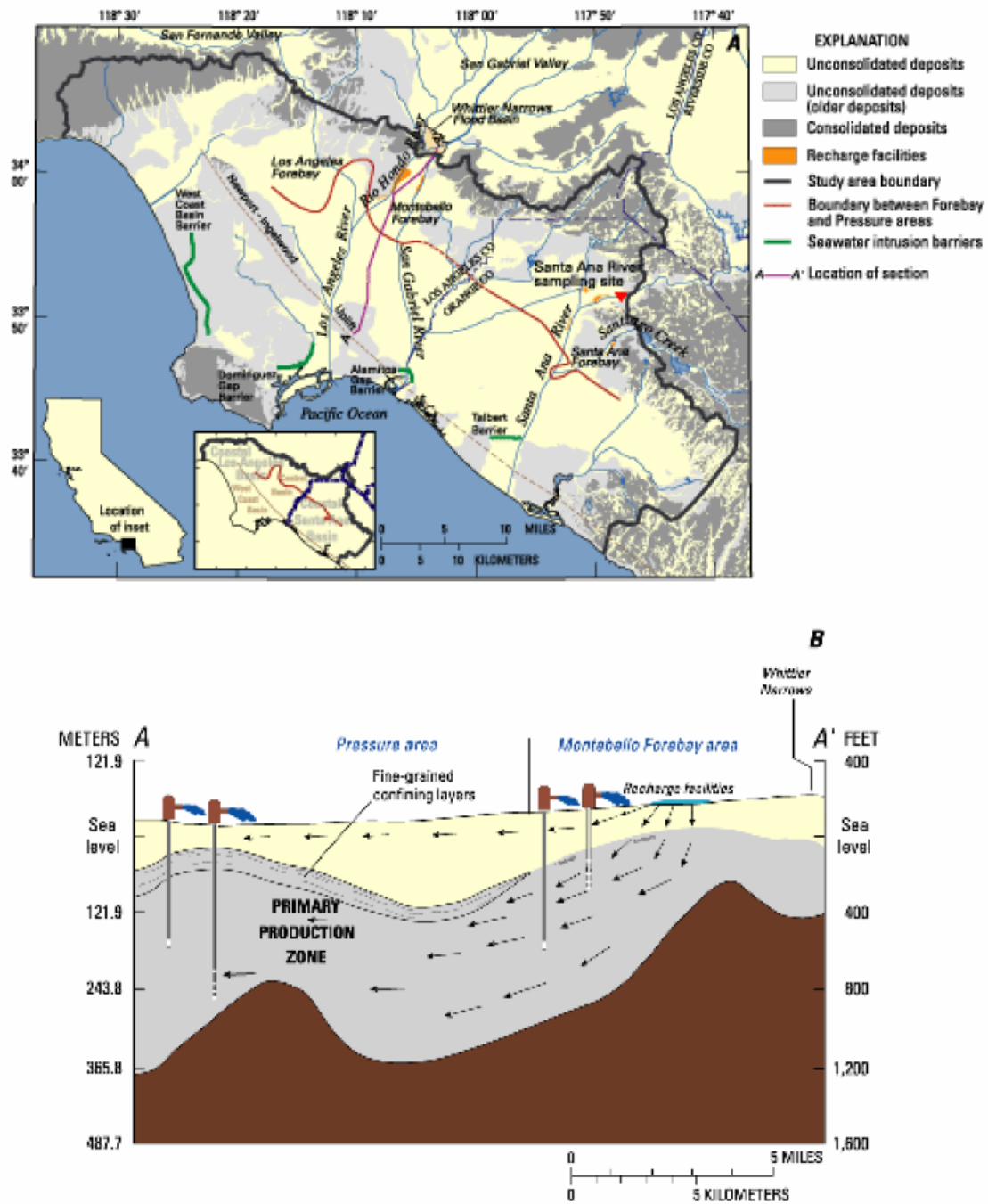


Figure 1. Major hydrogeologic features of the Los Angeles physiographic basin, California, (A) Study area and (B) Simplified cross-section showing the direction of ground-water flow.

SUMMARY AND CONCLUSIONS

Results of sampling untreated ground water from 178 active public supply wells in the Coastal Los Angeles and Coastal Santa Ana Basins show that volatile organic compounds (VOC) were detected in 61 percent of the ground-water samples at concentrations above the laboratory reporting limits (LRL). Thirty-nine of the 86 VOCs analyzed were detected: 29 VOCs were detected at concentrations above the LRL, and 10 VOCs were detected only at concentrations below the LRL. The most frequently detected VOC was trichloromethane (chloroform), detected at concentrations above the LRL in 82 (46 percent) of the samples.

Trichloroethene (TCE) and tetrachloroethene (PCE) were detected at concentrations above the LRL in 50 (28 percent) and 34 (19 percent) of the samples, respectively. Methyl *tert*-butyl ether (MTBE) was detected at concentrations above the LRL in 25 (14 percent) of the samples. VOCs detected in 7 percent or more of the samples also include 1,1,1-trichloroethane (TCA); 1,1-dichloroethane; *cis*-1,2-dichloroethene; bromodichloromethane; trichlorofluoromethane (CFC-11 or Freon 11); 1,1,2-trichloro-1,2,2-trifluoroethane (CFC-113 or Freon 113); and dichloromethane.

The six most frequently detected VOCs—chloroform, TCE, PCE, MTBE, TCA, and 1,1-dichloroethane—were also the most frequently detected VOCs in a wide variety of hydrogeologic conditions in ground water from urban areas across the United States. Most VOCs were detected at low-level concentrations, with only 29 percent of the detections above 1 $\mu\text{g/L}$ (microgram per liter). Two VOCs exceeded the state and federal primary maximum contaminant level (MCL) for drinking water. TCE concentrations exceeded the MCL (5 $\mu\text{g/L}$) in four samples with a maximum concentration of 69 $\mu\text{g/L}$, and PCE concentrations exceeded the MCL (5 $\mu\text{g/L}$) in five samples with a maximum concentration of 11 $\mu\text{g/L}$. More than half of the remaining samples with TCE and PCE detections contained concentrations between 0.5 and 5 $\mu\text{g/L}$. Because samples were collected prior to water treatment, sample concentrations do not represent the concentrations entering the drinking-water system.

VOC detections above the LRL are not uniformly distributed over the study area. Samples from wells in the forebay areas contained significantly more VOCs than samples from wells in the pressure area downgradient of the forebay. Samples with the most VOC detections generally are from wells located within 10–15 kilometers of the recharge facilities.

Ground water containing VOCs may be considered to be a tracer of postindustrial-aged water that recharged after the onset of intense urban development. The overall distribution of VOC detections is related to the hydrogeology and the recharge facilities in the forebay, and the distribution of individual VOCs is related to their history of use. For example, chloroform has been used since the 1920s, is the most widely distributed, and is detected at the greatest distances from the recharge facilities. In contrast, MTBE has the most recent history of use and was detected almost exclusively in the forebay areas at concentrations above the LRL.

Downward migration of VOCs from the land surface may be a viable process for contaminating aquifers in parts of the forebay areas, but there is little indication that the same process is active in the pressure area. The number of VOC detections above the LRL is significantly higher in samples from shallow wells than in samples from deep wells in the forebay areas; however, effects of the processes of lateral movement and downward vertical movement of ground water in the forebay areas cannot be separated.

Detections are not significantly different when wells are grouped by depth categories in the pressure area. The lack of contrast in the number of VOC detections between wells of different depths in the pressure area suggests that downward migration from the land surface is not the dominant pathway for VOCs to travel to the capture zones of public supply wells; however, the data are not sufficient to evaluate other processes that could affect the occurrence of VOCs in public supply wells, such as mixing over the long screened intervals. Isolated occurrences of multiple VOC detections and high concentrations of some VOCs, such as TCE, in individual wells may be explained by rapid vertical transport from localized sources.

Stable isotope composition indicates that ground water containing VOCs is a mixture of local precipitation and runoff and of water that is isotopically lighter (more negative) than the local sources. The isotopically

lighter water could either be Colorado River water or State Water Project water, both of which are imported to the basin and used as a source of groundwater recharge. Nearly all of the wells with no VOC detections at concentrations above the LRL are located in the pressure area and likely represent predominately older ground water than samples from wells in the forebay areas. The stable isotope data support the interpretation that VOCs in ground water are associated with the recharge facilities in the forebay areas.

Two of the most frequently detected VOCs in ground water, MTBE and chloroform, were detected more frequently and at higher concentrations in the Santa Ana River upstream from a recharge facility than in ground-water samples from active public supply wells, suggesting that the Santa Ana River may be a source of these VOCs in ground water. In contrast, the maximum concentrations and frequency of TCE and PCE detections were much higher in the ground water samples than in the Santa Ana River. However, the Santa Ana River water is only one component of the water used in the engineered recharge facilities and these data are not necessarily representative of past surface-water quality. In addition, engineered recharge water is likely not the sole source of VOCs in the forebay or the only process by which VOCs can enter the ground-water flow system. Point and nonpoint sources may enter the flow system in the forebay areas and spread radially outward.

The location of VOC sources on the land surface, such as population density and leaking underground fuel tank (LUFT) sites, or mapped susceptibility features, do not explain VOC occurrence in the study area. Population density was only weakly related to the occurrence of VOCs. The density of LUFT sites was not related to the occurrence or the frequency of MTBE detection, or the number of detections of VOCs above the LRL. The mapped surficial areas designated as hydrogeologically vulnerable areas did not encompass the area in which most of the VOCs were detected. Rather, the delineation of the forebay areas provides a reasonable approximation of areas susceptible to current contamination.

Overall, the data show that active public supply wells in the forebay areas are more vulnerable to contamination than those in the pressure area but areas downgradient of the recharge facilities are susceptible to contamination by any contaminant that will persist for decades. The lack of correlation between population density and LUFT density on the land surface and the occurrence and distribution of VOCs in active public supply wells in this study indicates that vertical migration of contaminants is not a dominant process outside the susceptible forebay areas, and the occurrence of VOCs is controlled primarily by the direction and rate of movement of water and the hydrogeology.

The overall occurrence of VOCs is not a function of localized contaminant sources or well characteristics, and rapid vertical transport, while possible, is not dominant. The acceleration of the lateral rate of groundwater flow produced by engineered recharge and ground-water pumping in the Los Angeles physiographic basin appears to be the dominant factor in controlling the distribution of VOCs in active public supply wells; thus, low-level VOCs can act as tracers of postindustrial-aged water recharged in these southern California Coastal Basins.

Stable Isotopes and Volatile Organic Compounds Along Seven Ground-Water Flow Paths in Divergent and Convergent Flow Systems, Southern California, 2000

By Barbara J. Milby Dawson, Kenneth Belitz, Michael Land, and Wesley R. Danskin
Water-Resources Investigations Report 03-4059

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Study Area

Three southern California ground-water basins were selected to assess VOC occurrence along ground-water flow paths: the Central Basin in Los Angeles County, the Main Basin in Orange County, and the Bunker Hill Basin in San Bernardino County (fig. 1). The Central and Main Basins are located in the southern California coastal plain, and Bunker Hill Basin is located in the inland Santa Ana Valley.

The aquifers in the southern California coastal plain consist of Recent to Pleistocene unconsolidated alluvial and marine sediments. Ground-water flow in the coastal plain moves from the forebay (also referred to in this report as proximal) areas into the pressure (also referred to as distal) areas. The ground-water flow systems in the coastal plain are divergent; ground water moves radially away from a focused point of recharge (proximal) into the rest of the basin (distal). The change from forebay to pressure area is based on the relative abundance of finer-grained deposits and is not well defined. Aquifers in the forebay areas are generally unconfined and more connected to overlying land use and each other than in the pressure areas; clay and silt lenses generally do not significantly impede vertical ground-water flow. The aquifers in the pressure area consist of sands and gravels separated by relatively continuous layers of silt and clay that act as confining layers. The pressure area covers roughly 60 percent of the area in the Central Basin and in the Main Basin. Ground-water pumping is generally from about 50 to 300 m (meter) below land surface (California Department of Water Resources, 1961; Herndon and others, 1997).

The valley-fill aquifers in the inland Santa Ana Basin consist of alluvial fan and stream deposits eroded from the surrounding mountains. These deposits consist of sands and gravels with layers of silt and clay. Faults in the sediment act as barriers to ground-water flow in places, and divide the valley into several ground-water basins. The flow paths selected for this study are located in the Bunker Hill Basin, which is bounded by faults in the northeastern part of the Santa Ana Valley. The ground-water flow system in the Bunker Hill Basin is convergent; ground water moves radially away from dispersed recharge areas (proximal) toward an area of focused discharge (distal). The aquifer system in the Bunker Hill Basin is divided into upper, middle, and lower aquifers (Dutcher and Garrett, 1963). According to Dutcher and Garrett (1963, p. 63), the upper aquifer is confined in the distal area of the flow paths by a near-surface confining member, which "is discontinuous; it may be absent, thinner, or locally semipermeable in the vicinity of Warm Creek," and is mapped by Duell and Schroeder (1989, fig. 3) as cut by river channel deposits that are highly permeable. In the distal area, the confining units in the aquifer system are described by Duell and Schroeder (1989, p. 9) as "clay layers that act as leaky confining beds." The confined (pressure) area covers roughly 20 percent of Bunker Hill Basin (Dutcher and Garrett, 1963; Duell and Schroeder, 1989).

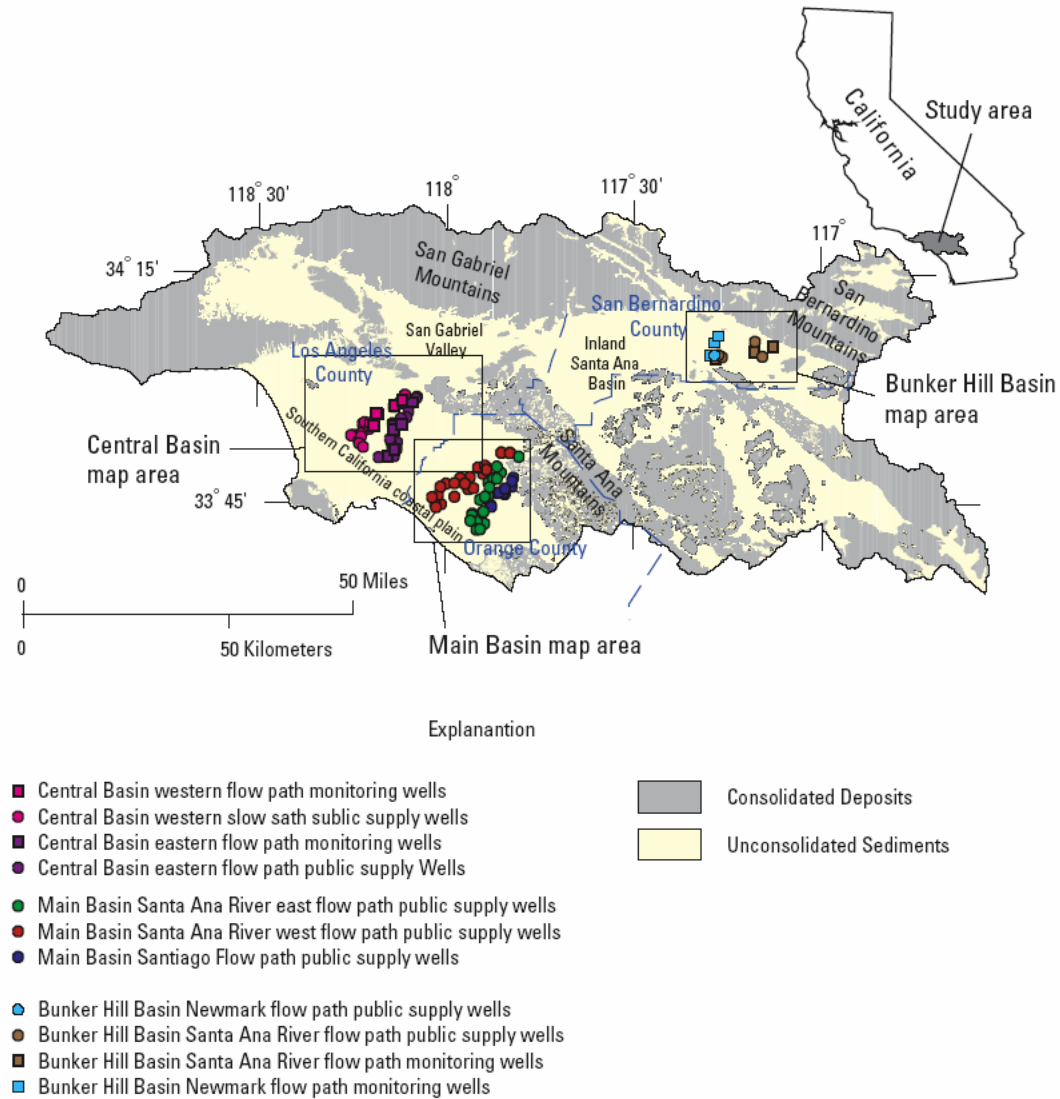


Figure 1. Study area, basin, and well locations in Southern California.

CENTRAL BASIN, LOS ANGELES COUNTY

Hydrogeology

Two ground-water flow paths were selected in the Central Ground Water Basin of the Los Angeles Coastal Plain (California Department of Water Resources, 1961), hereinafter referred to as the Central Basin; both flow paths begin in the Montebello forebay area upgradient of the Rio Hondo and San Gabriel recharge facilities and end in the Central Basin pressure area upgradient of the Newport–Inglewood fault zone (fig. 2). The western flow path is 21 km (kilometer) long and is located near and west of the Rio Hondo and the Los Angeles Rivers. The eastern flow path is 22 km long and is located near and west of the San Gabriel River. Both flow paths were selected to follow ground-water gradients based on water-level contours (fig. 3) (Water Replenishment District of Southern California, 2000).

Historically, about two thirds of ground-water recharge to the Central Basin was from the Rio Hondo and the San Gabriel Rivers as they entered the basin at the Whittier Narrows from the inland San Gabriel Valley to the northeast; minor recharge resulted from ground-water flow through the Whittier Narrows

(California Department of Water Resources, 1961). Other sources of recharge include irrigation, local precipitation, and runoff from the surrounding hills. Ground-water recharge has been engineered since the 1930s at recharge facilities on the Rio Hondo and San Gabriel Rivers southwest of the Whittier Narrows (Water Replenishment District of Southern California, 2001). A variety of water sources have been used historically for engineered recharge, including storm flow and treated wastewater from the San Gabriel Valley and imported water from northern California and the Colorado River (Water Replenishment District of Southern California, 2001). Nearly all ground-water discharge from the basin is by pumping from the forebay and pressure areas (Water Replenishment District of Southern California, 2001).

Ground-water levels measured in wells tapping the main pumping zone range from more than 40 m above sea level near the recharge facilities, to more than 20 m below sea level near the Newport–Inglewood fault zone (fig. 3) (Water Replenishment District of Southern California, 2000). The water-level contour pattern in most of the Central Basin is semicircular and centered on the Whittier Narrows. The ground-water flow system in the Central Basin is divergent; ground water moves radially away from a common area of recharge below the Whittier Narrows into the rest of the basin. The two flow paths for the 68 wells sampled in the Central Basin (table 1) begin at a well upgradient of the Rio Hondo and the San Gabriel recharge facilities (fig. 2). Twenty-five sites—23 are public supply wells and 2 are USGS monitoring sites with 4 to 6 individual wells of varying depths—were on the western flow path; 33 wells were sampled (fig. 2). Twenty-two sites—19 are public supply wells and 3 are USGS monitoring sites with 5 to 6 individual wells of varying depths—were on the eastern flow path; 36 wells were sampled. Geologic, hydrologic, and additional water-quality data from the USGS monitoring wells are described in Land and others (2002). The wells on the western flow path ranged from 49 to 409 m deep, and wells on the eastern flow path ranged from 27 to 366 m deep. There was no significant difference between well depths or distances on the western and eastern flow paths (table 2) and no correlation between distance and well depth on either Central Basin flow path (table 3).

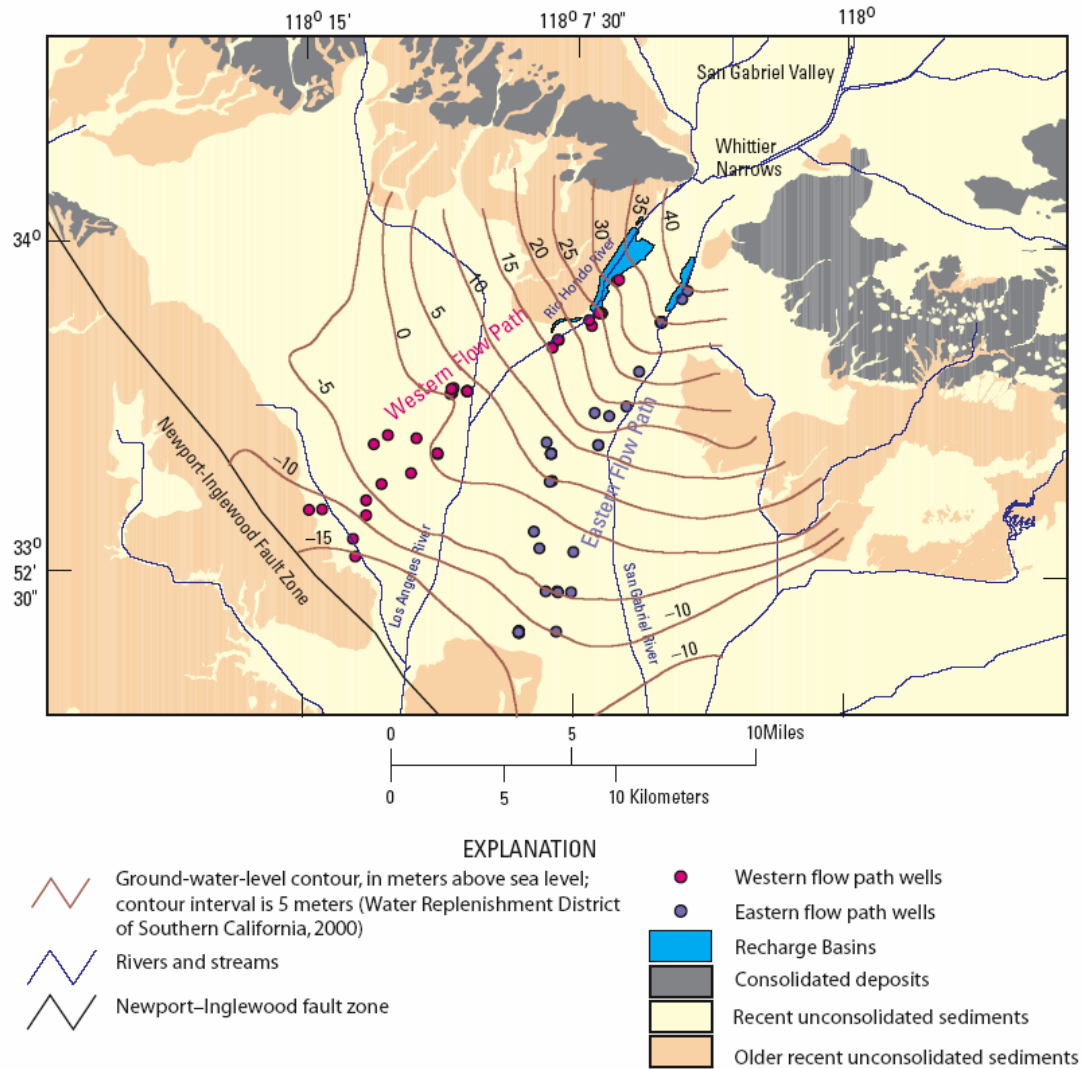
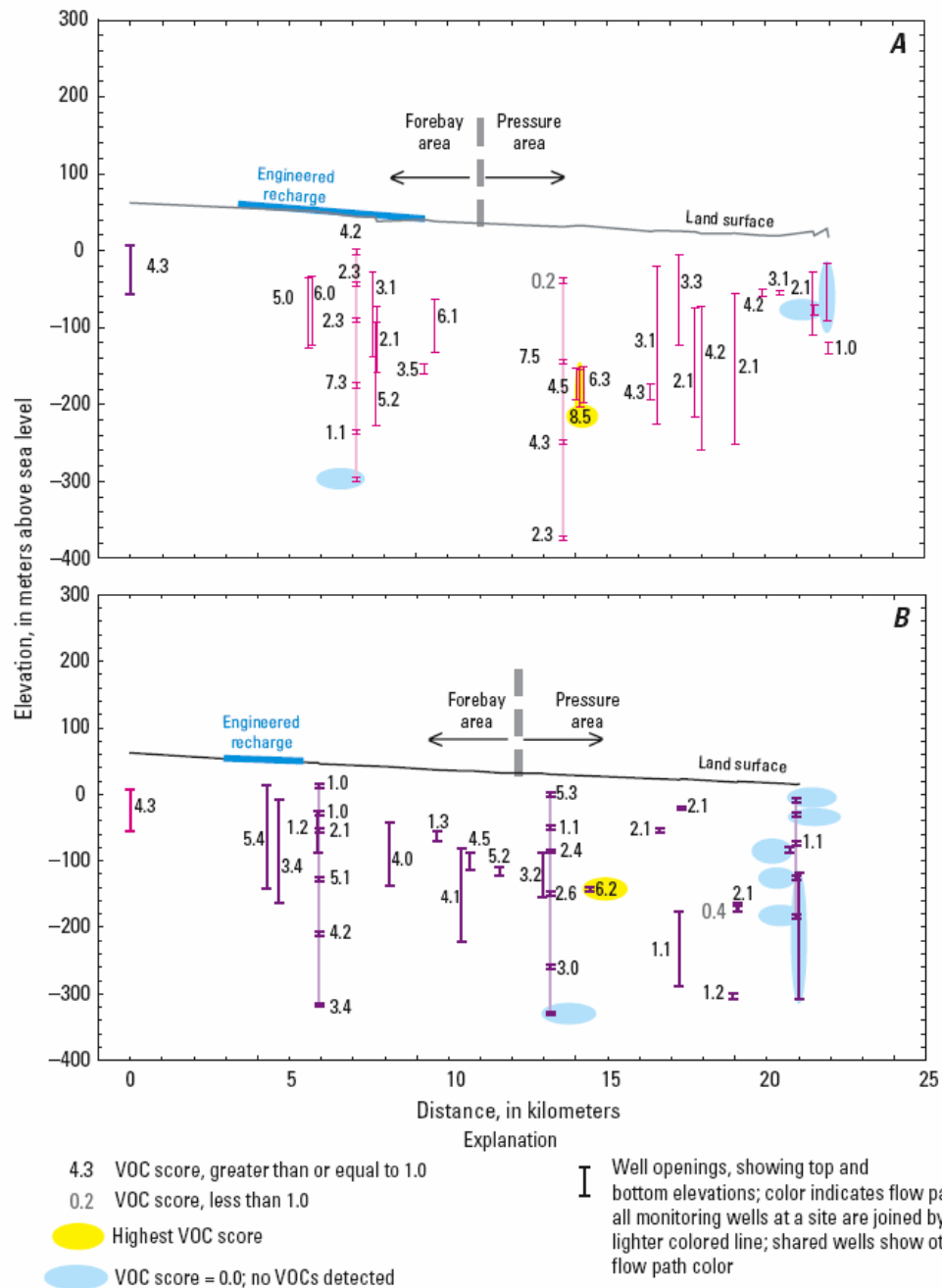


Figure 3. Ground-water level contours in the Central Los Angeles Basin, California, spring 2000.



MAIN BASIN, LOS ANGELES COUNTY

Hydrogeology

Three ground-water flow paths were selected in the Main Basin of the Coastal Santa Ana ground-water basin. Two flow paths, the Santa Ana River (SAR) west and SAR east, begin near the recharge facilities along the Santa Ana River in the forebay area and end in the pressure area, upgradient of the Newport–Inglewood fault zone (fig. 17). The SAR west flow path is west of the Santa Ana River, and the

SAR east flow path is near and east of the Santa Ana River. The third flow path, Santiago, begins near the recharge facilities on Santiago Creek, is near and south of Santiago Creek, and ends on the SAR east flow path.

The ground-water flow system in the Main Basin includes two flow systems, one associated with the Santa Ana River and the other with Santiago Creek. As in the Central Basin, both are divergent flow systems; ground water moves radially away from a focused point of recharge (proximal) into the rest of the basin (distal). In contrast to the Central Basin, there are two main points of recharge, one along the Santa Ana River and a smaller one along Santiago Creek.

Three-fourths of ground-water recharge takes place at engineered recharge facilities in and along the Santa Ana River and at the smaller facility on Santiago Creek. Other sources of recharge include irrigation return, local precipitation, and ground-water inflow from other basins (Herndon and others, 1997). Engineered recharge facilities, in operation along the Santa Ana River since the 1940s, have used stormflow and base flow from the inland Santa Ana Basin, imported Colorado River water, and imported northern California water (Moreland and Singer, 1969; Herndon and others, 1997). The Santiago Creek recharge facility, in operation for about 10 years, uses local mountain runoff from the Santa Ana Mountains and excess stormflow diverted from the Santa Ana River recharge facilities. Additional engineered recharge is from injection wells at the Talbert and Alamitos seawater barriers ([fig. 17](#)). The Talbert barrier has been in operation since 1975 and uses a mixture of treated wastewater and deep ground water (Herndon and others, 1997; Orange County Water District, 2001). The Alamitos barrier has been in operation since 1965 and uses imported water (Herndon and others, 1997; Orange County Water District, 2001). Nearly all ground-water discharge in the Main Basin is from ground-water pumping in the pressure area.

The Main Basin flow paths were selected to approximately follow ground-water gradients based on water-level contours ([fig. 18](#)) (Wildermuth Environmental, Inc., 2000). Well information for the 51 wells sampled in the Main Basin are given in table 6. Wells were selected that tap the upper production zone (from 60 to 120 m below land surface), the lower production zone (from 150 to 300 m below land surface), and both zones.

The SAR west flow path (25 km long) follows the middle of the radial flow from the Santa Ana River recharge facilities from proximal to distal areas. Twenty-three public supply wells were sampled on the SAR west flow path ([fig. 17](#) and [table 6](#)).

The Santiago flow path (11 km long) follows the middle of the radial flow from the Santiago Creek recharge facilities from proximal to distal areas. Although the Santiago flow path is shorter in length, it covers the same parts of that flow system as the SAR west flow path (proximal to distal). The shorter length reflects the shorter time of operation and lower recharge rates of the Santiago Creek recharge facility. Ten public supply wells were sampled on the Santiago flow path ([fig. 17](#) and [table 6](#)).

The SAR east flow path (22 km long), which follows the eastern edge of the radial flow from the Santa Ana River recharge facilities from proximal to distal areas, is also distal to the Santiago flow path. Twenty public supply wells were sampled on the SAR east flow path ([fig. 17](#) and [table 6](#)).

Well distances and depths are statistically similar along the SAR west and east flow paths ([table 7](#)). The wells on the Santiago flow path are more evenly distributed along the entire flow path than the wells on the SAR flow paths, which have more wells located in the downgradient part of the flow paths ([fig. 17](#)). Well depths are similar between the SAR east and Santiago flow paths ([table 7](#)). The bottoms of well openings are statistically deeper on the SAR west than on the Santiago flow path ([table 7](#)). There was no correlation between well depths and distance downgradient along either SAR flow path. The midpoints and bottoms of well openings were significantly deeper with distance downgradient along the Santiago flow path.

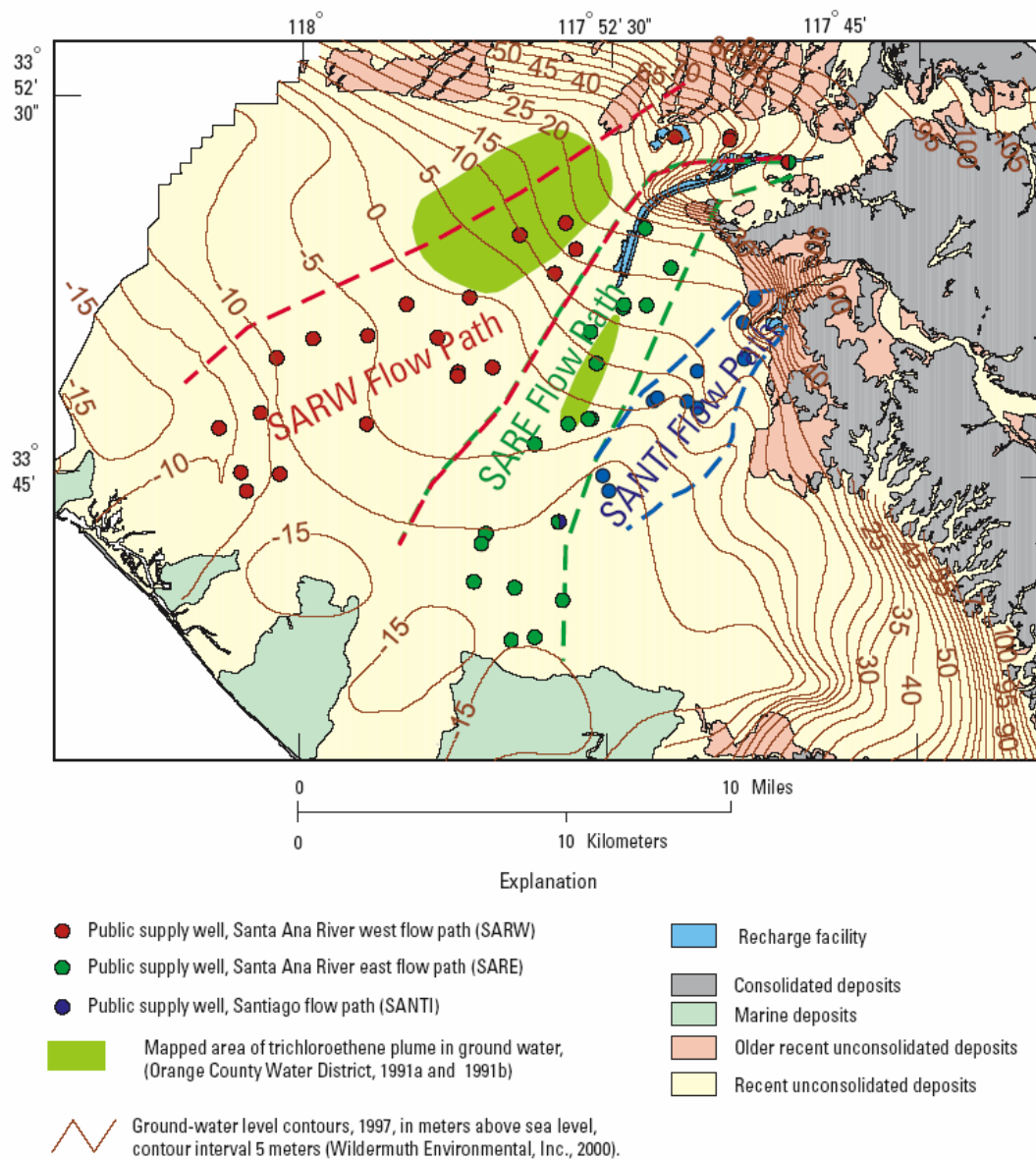
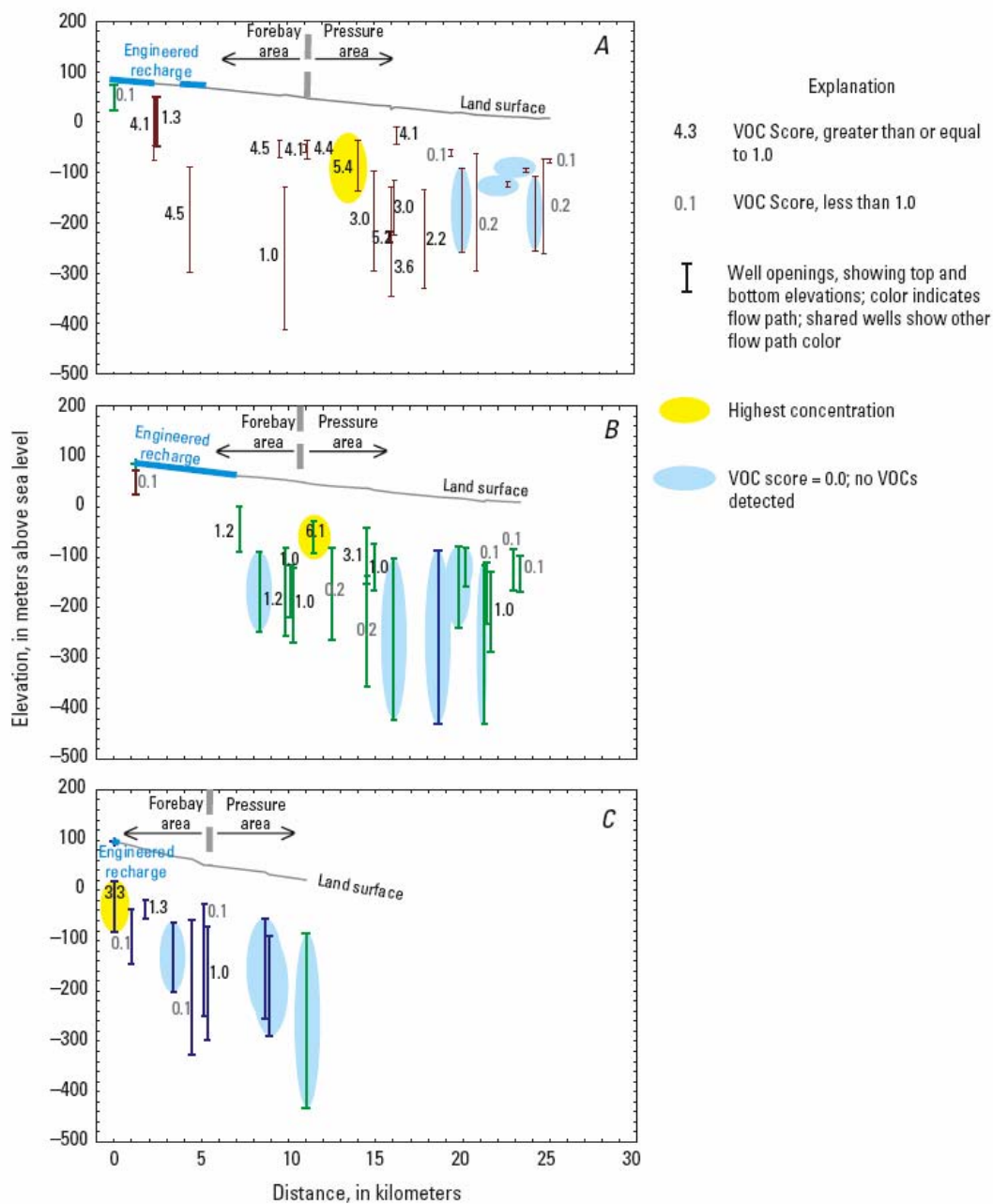


Figure 18. Ground-water level contours in the Central Los Angeles Basin, California, spring 2000.



BUNKER HILL BASIN, SAN BERNARDINO COUNTY

Hydrogeology

Two ground-water flow paths were selected in the Bunker Hill Basin of the inland Santa Ana ground-water basin (fig. 32). Both paths begin near engineered recharge facilities located along the San Bernardino Mountain front, and end near the San Jacinto Fault. The SAR flow path is located along of the Santa Ana River, and the Newmark flow path is located along the Warm and the East Twin creeks. The ground-water flow system in the Bunker Hill Basin is radially convergent. Ground water recharges at several points along the upgradient edges of the basin (proximal area) and moves to a focused area of discharge near the San Jacinto Fault (distal area). The distal parts of the flow paths are confined (fig. 32). However, Dutcher and Garrett (1963, p. 28 and 63) describe the uppermost confining unit near land surface as discontinuous and ranging from semiconfining to confining. Duell and Schroeder (1989, p. 9) describe the confining units at depth in the aquifer system as poorly permeable and leaky.

Recharge to the flow system varies seasonally and is largely from infiltration of runoff from streams exiting the San Gabriel and San Bernardino Mountains (fig. 32). Much of the runoff is diverted to engineered recharge facilities along the mountain front, in or adjacent to stream channels. These facilities have been in operation since the early 1900s using runoff from the San Bernardino Mountains to the north and the San Gabriel Mountains to the west (Moreland, 1972), and imported water from northern California in the 1970s and 1980s (Hardt and Freckleton, 1987). The flow paths are influenced by recharge facilities on the East Twin Creek, Santa Ana River, Mill Creek, and, to a lesser extent, Warm Creek and City Creek. Ground-water discharge is primarily by pumping.

The Bunker Hill Basin flow paths were selected to approximately follow ground-water gradients based on water-level contours (fig. 33) (Wildermuth Environmental, Inc., 2000). Depths to water range from tens of meters near the flanks of mountains to near land surface along rivers and in wetland areas in the distal area of the flow paths. Well information for wells sampled in the Bunker Hill Basin is given in table 11. Wells on two different flow paths were selected in the Bunker Hill Basin. The Newmark flow path (6 km long) follows the radial flow that originates near the East Twin Creek recharge facilities (fig. 32). Ten monitoring wells and 1 public supply well were sampled on the Newmark flow path. The SAR flow path (16 km long) follows the radial flow that originates near the Santa Ana River and Mill Creek recharge facilities. Ten monitoring wells and 5 public supply wells were sampled on the SAR flow path.

Distance downgradient along the Bunker Hill flow paths was measured relative to a point of convergence on the San Jacinto Fault (fig. 34). The convergence point is roughly 16 km from the recharge area on the SAR flow path, but is only about 11 km from the recharge area on the Newmark flow path. For consistency with distance measurements in the coastal plain flow paths, well distances increase in the downgradient direction. Consequently, in Bunker Hill, a well at a distance of 10 km on the SAR flow path is 10 km from the recharge area, but on the Newmark flow path a well at 10 km is only about 5 km from the recharge area.

The Newmark path has flow lines running north to south, and the Santa Ana River path has flow lines running east to west. The distributions of wells on the Bunker Hill flow paths differ. Wells on the Newmark path are located in the middle and downgradient parts of that flow path, with no wells in the upgradient area. Wells on the SAR flow path are located upgradient and downgradient, with no wells in the middle part of that flow path. For this investigation, these two flow paths were analyzed as one, increasing the coverage of wells along the flow paths and the number of wells available for analysis. To justify the combining of the two flow paths, statistical comparisons were made along the lower parts of both flow paths (distances greater than 11 km) where the distribution of wells was the same. No statistical difference was found between the flow paths, except in stable isotopes and benzene occurrence (table 12). Depths to the tops and midpoints of the wells are similar between both flow paths; the bottoms of well openings are significantly deeper on the SAR flow path (table 12). There was no correlation between well depths and distance downgradient along the Bunker Hill flow paths (table 13).

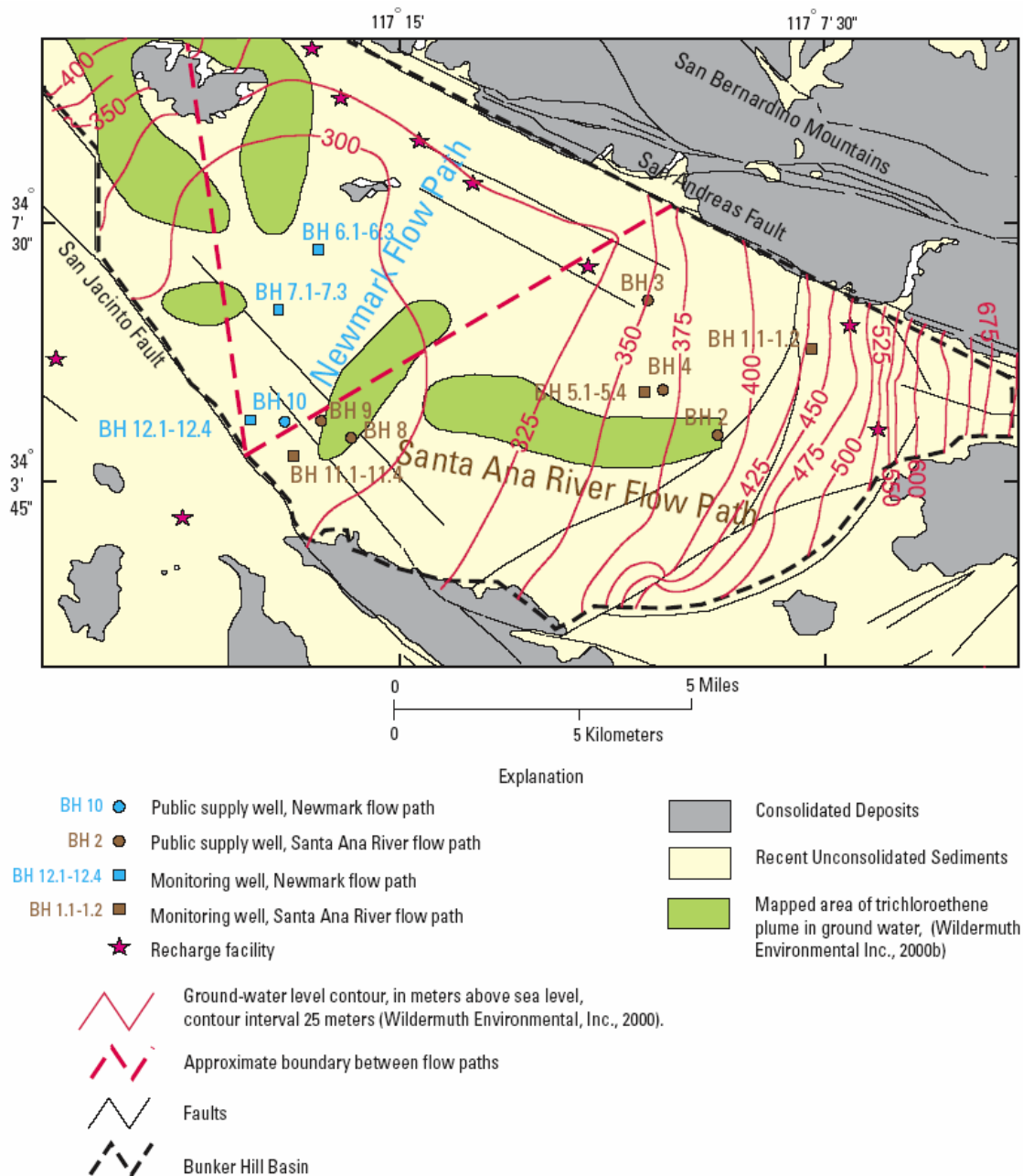


Figure 33. Ground-water level contours and well locations in the Bunker Hill Basin, San Bernardino County, California.

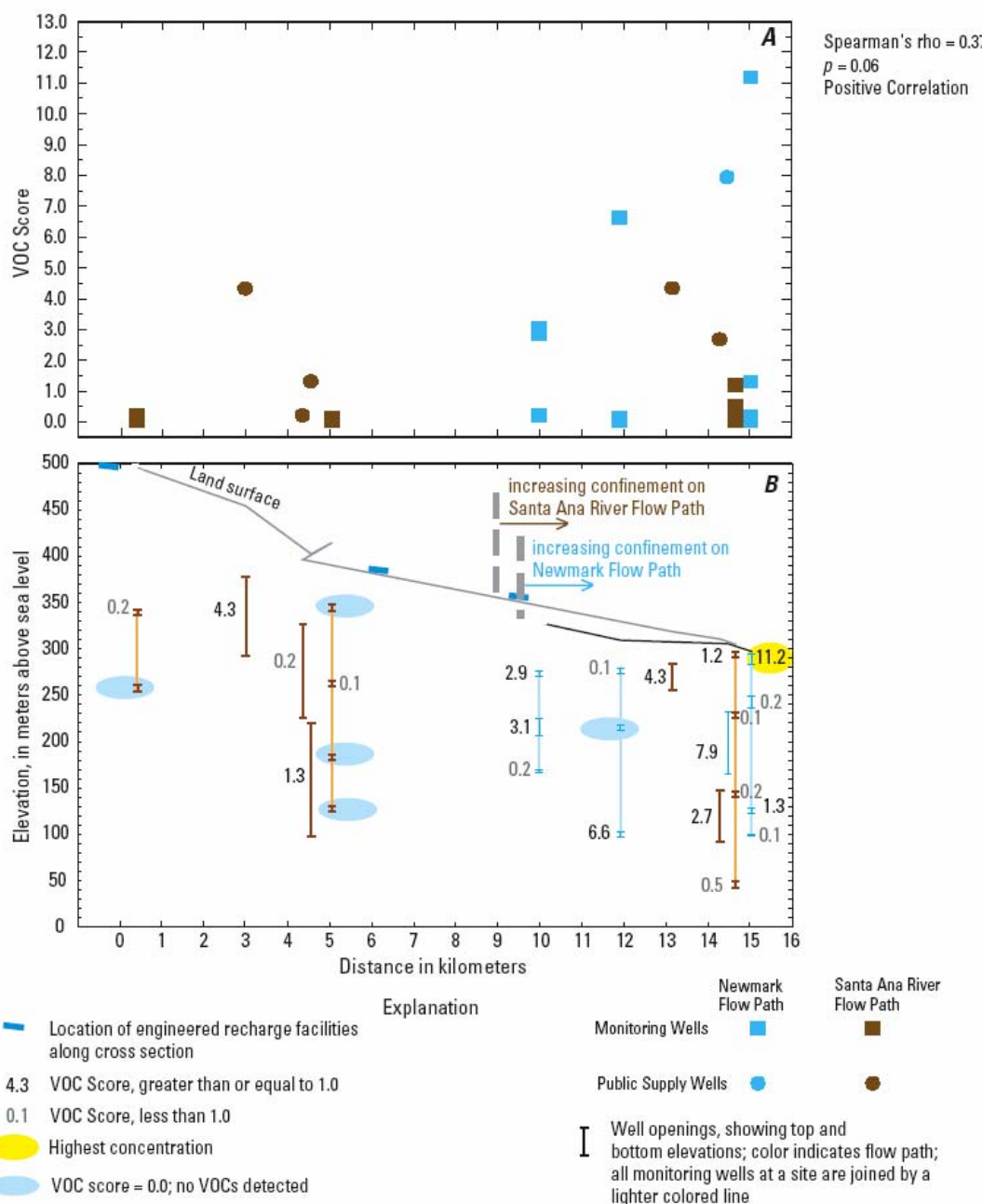


Figure 36. VOC score (A) versus distance and (B) in wells along flow paths, Bunker Hill Basin, San Bernardino County, California, 2000.

SUMMARY AND CONCLUSIONS

Stable isotopes and volatile organic compounds (VOC) were examined along ground-water flow paths in three southern California basins. Five flow paths are located in the southern California coastal plain. Ground-water flow systems in the Coastal Plain are characterized by areas of focused ground-water recharge; ground-water flows radially outward into the basins toward dispersed areas of ground-water pumping (discharge). Two flow paths are located in the Inland Santa Ana Basin, which is characterized by dispersed areas of ground-water recharge; ground-water flow converges radially toward an area of focused discharge.

As part of this study, a new way of counting VOC detections in a sample, the VOC score, was developed to include detections of VOCs below the laboratory reporting levels (LRL) in data analyses. About half of the VOC detections in this study were at concentrations below the LRL. The VOC score allows all the VOC detections to be used for graphical and statistical analyses, and takes into account the analytical certainty based on the LRL. The VOC score gives more weight to detections above LRL, which have less than 1 percent chance of false positive or false negative detections, and less weight (one tenth) to detections below the LRL, which may have a greater than 1 percent chance of false negative detections.

In the Central Basin, Los Angeles County, the two flow paths sampled originate at the San Gabriel and Rio Hondo recharge facilities in the Montebello forebay area. Stable isotopes indicate that ground water along both flow paths is from similar sources. The patterns of VOC score and concentrations indicate that a volume of water that contains VOCs and is presumably young (has recharged since the 1930s) extends about 20 km into the Central Basin from the Montebello forebay and to depths of at least 400 m below land surface. In this volume of water, VOC score and concentration are variable and may reflect the history of VOC inputs and fate in the subsurface environment. This water is mixing with presumably older ground water that contains no VOCs. The patterns also suggest that some VOCs were introduced to ground water at higher concentrations in the past than at present and that VOCs continue to enter the Central Basin through the Whittier Narrows. The pattern of MTBE concentrations is different from the other VOCs, and suggests that MTBE is currently being introduced into ground water in the upgradient areas of the Central Basin at higher concentrations than in the past.

An important exception to the general pattern is the very high concentrations of TCE and PCE in shallow wells located in the pressure area. These wells are downgradient from wells with much lower concentrations. This indicates that even in the pressure area, ground-water is susceptible to contamination from VOC sources. In the Main Basin, Orange County, three flow paths were sampled. The Santa Ana River (SAR) west flow path is located in the radial flow system that originates at the SAR recharge facilities. The Santiago flow path is located in the radial flow system from the Santiago Creek recharge facilities. The SAR east flow path originates in the SAR flow system, but also is along the convergence of the Santa Ana River and Santiago Creek flow systems. Ground water is isotopically the same on SAR flow paths, indicating similar sources of water. Ground water along the Santiago flow path is isotopically different from the other two flow paths, indicating a different source of ground water on the Santiago flow path.

Although stable isotopes show that the SAR west and east flow paths are similar and the Santiago flow path is different, the VOCs show a contrasting relation. VOC score and VOC concentrations on the SAR west flow path are different from both the SAR east and Santiago flow paths. VOC occurrences were statistically similar between the SAR east and Santiago flow paths, suggesting that recharge from Santiago Creek is affecting VOC occurrences on the SAR east flow path. The patterns of VOC scores, chloroform, and TCE along the Santiago flow path are different than on the SAR flow paths and suggest that more VOCs are currently reaching ground water than in the past. These patterns may reflect that the Santiago Creek recharge facilities have been operational for a shorter period of time than the SAR recharge facilities. VOC score and concentrations on the SAR west flow path show a pattern similar to the Central Basin flow paths. A volume of presumably young water containing VOCs extends to a distance around 18 km from the SAR recharge facilities and a depth of at least 300 m, and is mixing with presumably older water containing no VOCs.

In the Inland Bunker Hill Basin, two flow paths were sampled. In contrast to the Coastal Plain basins, ground-water recharge in the Bunker Hill Basin is dispersed along the mountain front, and ground-water flow converges radially to an area of focused discharge along the San Jacinto Fault. The two flow paths originate in different sections of the mountain front and converge near the San Jacinto Fault. The stable isotopes indicate different sources of water for the two flow paths; however, VOC occurrence is similar between them. In contrast to the patterns found in the Central and Main Basins, higher VOC scores and concentrations of MTBE, TCE, and PCE in the Bunker Hill Basin were detected at the distal end of the flow paths in the confined part of the flow system. This suggests that either (1) high concentrations of these VOCs reached ground water at some time in the past in the unconfined, upgradient area of the flow paths and have migrated downgradient or (2) VOCs are introduced all along the flow paths, even in the confined part of the aquifer. The absence of MTBE in the proximal part of the flow paths and presence in the distal

part suggest that the confining units present in the distal part of the Bunker Hill Basin do not prevent VOCs from reaching ground water. The patterns of VOC score and concentration in the Bunker Hill Basin indicate a different situation than in the coastal plain. In Bunker Hill, a volume of presumably young water containing relatively few VOCs is moving outward from the recharge areas and mixing with a volume of possibly intermediate to older water that contains many VOCs.

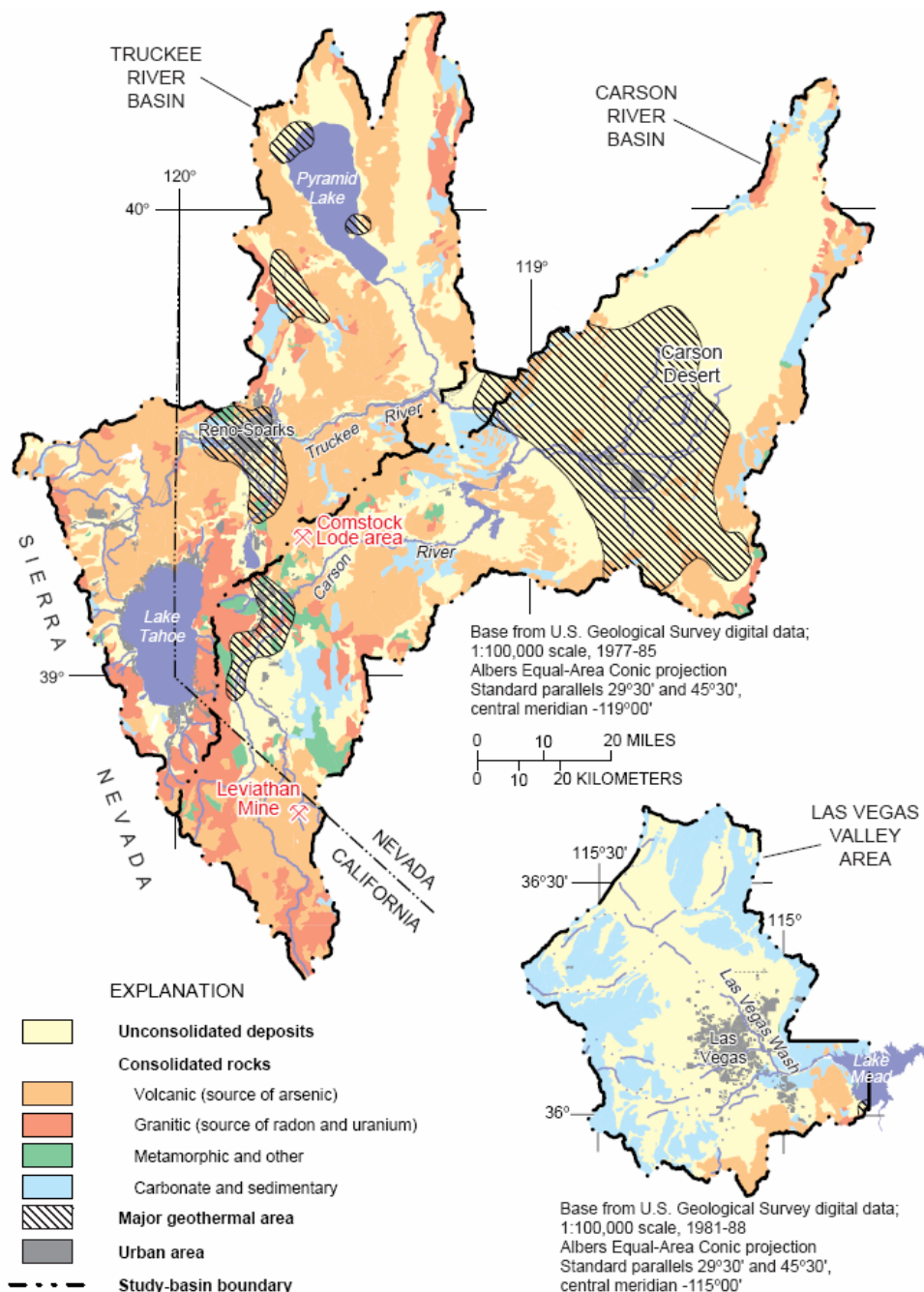
The susceptibility of public supply wells to land surface contamination can be understood, in part, by developing an understanding of the regional ground-water flow system. In the southern California coastal plain, and perhaps other radially divergent confined flow systems, VOC score and concentrations tend to decrease with increasing distance. In the inland Santa Ana Basin, and perhaps other radially convergent, confined flow systems, VOC score and concentrations tend to increase with increasing distance. These results suggest that distal wells tapping older ground water are less susceptible to land surface contamination than proximal wells tapping younger ground water in radially divergent confined systems. Conversely, distal wells tapping older ground-water probably are more susceptible than proximal wells tapping younger ground water in radially convergent, confined systems. These differences in susceptibility are partly controlled by the pattern of ground-water flow (radially divergent or radially convergent) and perhaps by the degree of confinement.

NVBR

Water Quality in the Las Vegas Valley Area and the Carson and Truckee River Basins, Nevada and California, 1992-96

By Hugh E. Bevans, Michael S. Lico, and Stephen J. Lawrence
U.S. GEOLOGICAL SURVEY CIRCULAR 1170

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NVBR Ground-water chemistry				
Study-Unit Survey	Major chemical constituents, nutrients, trace elements, radon-222, volatile organic compounds, dissolved organic carbon, pesticides, phenols, and MBAS to assess principal aquifer systems.	Existing supply wells in principal aquifers in Las Vegas area, Reno-Sparks area, and Carson Valley.	57	Once in 1995.
Land-Use Studies	Study-Unit Survey data to assess effects of urban and agricultural land use on shallow ground water.	Shallow observation wells in urban (Las Vegas and Reno-Sparks) and agricultural (Carson Valley and Carson Desert) areas.	90b	Once in Las Vegas (1993), and once in Reno-Sparks, Carson Valley, and Carson Desert (1994).
Special studies				
Irrigation Drainage	Streamflow, major chemical constituents, nutrients, trace elements, and pesticides in water and trace elements in bottom sediment to determine major sources and loads.	Irrigation drains in Carson Desert.	22	Water chemistry (3 times), pesticides and bottom sediment (once); Feb.-Sept. 1995.
Endocrine Disruption	Organochlorine pesticides, PCBs, dioxins, furans, and semivolatile organic compounds in water, bottom sediment, semi-permeable membrane devices, and carp to determine occurrence. Sex-steroid hormones in carp to assess endocrine systems.	Las Vegas Wash and Las Vegas Bay, Callville Bay, Colorado River inlet, Virgin River inlet, and Muddy River inlet of Lake Mead.	16	Once, 6 sites in 1995 and 10 sites in 1996.

The NVBR Study Unit (fig. 1), which includes the Las Vegas Valley area in southeastern Nevada and the Carson and Truckee River Basins in northwestern Nevada and northeastern California, is typical of Basin and Range physiography (see Glossary).

High mountains, exceeding 10,000 ft in some areas, surround valleys underlain by thick, unconsolidated deposits [1]. Consolidated rocks in high mountains are primarily carbonate rocks in the Las Vegas Valley area and granitic and volcanic rocks in the Carson and Truckee River Basins (fig. 2). Unconsolidated deposits beneath valleys (fig. 2), which range from clay to boulders, are present in thicknesses as great as 5,000 ft [4]. **High mountains are headwater areas that can receive more than 20 inches of annual precipitation (fig. 3). Basin areas are low mountains and valleys that can receive less than 4 inches of annual precipitation (fig. 3).** Most of the precipitation falls during the winter (fig. 3). Precipitation in headwater areas provides runoff to recharge principal aquifers [4] and to sustain streamflow. Principal aquifers (fig. 3) are present in unconsolidated deposits beneath some valleys.

Streamflow in Las Vegas Wash, which drains Las Vegas Valley, is sustained by urban drainage and discharge of treated sewage from the Las Vegas urban area. Although mean seasonal streamflow is higher in winter and spring, seasonal differences are overshadowed by the nearly fourfold increase in streamflow during October 1964 through September 1995 (water years 1965-95). This increase in streamflow (fig. 4A) was caused by increasing rates of urban drainage and discharge of treated sewage [4].

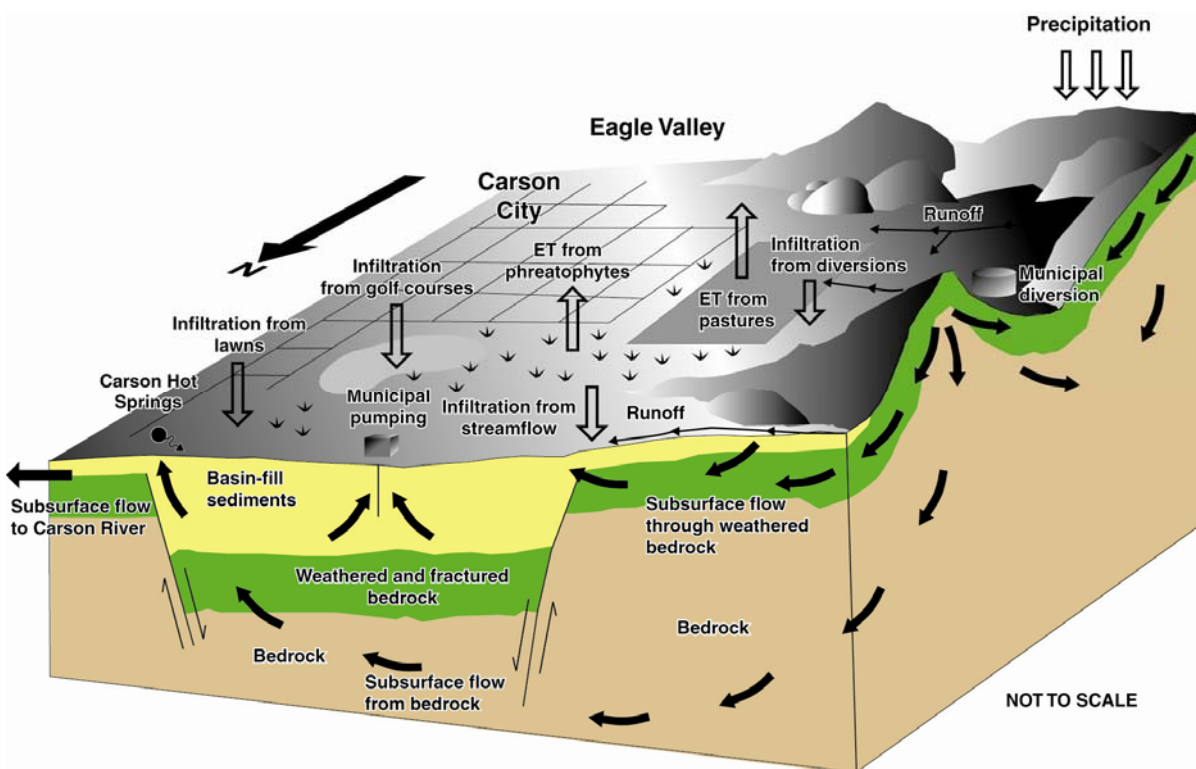
Environmental Subareas Were Identified for Investigation

The NVBR Study Unit was divided into subareas representing important combinations of natural and human factors. A conceptual model of basin and range hydrology was used to develop a framework of primary subareas (headwater and basin areas). Headwater areas (fig. 3) were defined as mountains and adjacent areas where precipitation is adequate to provide ground-water recharge and streamflow (yielding 0.2 inch or more of annual runoff [59]). Basin areas do not receive adequate precipitation for appreciable ground-water recharge and streamflow, but can receive recharge and streamflow from upstream headwater

areas. Natural discharge in the Study Unit is by stream outflow from and evapotranspiration in the Las Vegas Valley and by only evapotranspiration in the Carson and Truckee River Basins (hydrologically closed basins characterized by internal drainage). Secondary subareas were developed by subdividing the two primary subareas (headwater and basin areas) into land uses that represent more natural conditions (forest and range) or human activities (urban and agricultural land use; fig. 5). Water resources in the environmental subareas were then identified for study. Principal surface-water resources included in the investigation were Las Vegas Wash and Lake Mead, the Carson River, and the Truckee River (fig. 3). Principal aquifers (fig. 3) were defined as unconsolidated deposits beneath valleys with 100 ft or more of saturated thickness [60,61,62] and containing ground water with less than 1,000 mg/L of dissolved solids [63,64]. Headwater areas are primarily forest and basin areas are primarily range. The principal headwater urban area is Reno-Sparks and the principal headwater agricultural area is Carson Valley. The principal basin urban area is the Las Vegas urban area and the principal basin agricultural area is Carson Desert.

Ground-water-chemistry activities included Land-Use Studies and Study-Unit Surveys (fig. 13).

Land-Use Studies of shallow ground water were completed in the Las Vegas basin urban area, the Carson Valley headwater agricultural area, the Carson Desert basin agricultural area, and the Reno-Sparks headwater urban area. The Study-Unit Surveys sampled principal aquifers in the Las Vegas urban area, Carson Valley, and Reno-Sparks urban area.



Minerals in Rocks and Sediment Are Sources of Chemical Constituents

Certain rock types contain elements that may be harmful to humans or animals if they are released and concentrated in the water. In the NVBR Study Unit, arsenic, radon, and uranium were detected in some water samples t concentrations that exceeded drinking-water standards. The geology of the NVBR NAWQA Study Unit is quite diverse (fig. 2), including carbonate, metamorphic, sedimentary, granitic, and volcanic rocks, and unconsolidated deposits derived from the rocks [4]. Water in contact with each of these rock types can derive a chemistry specific to that particular rock type. In headwater areas, such as Carson Valley and the Reno-Sparks area, ground and surface water tend to be dilute (less than 1,000 mg/L of dissolved solids) and contain calcium and bicarbonate ions as the major dissolved constituents. In basin areas, such as Carson Desert and Las Vegas Valley, ground water tends to be dilute to

briny (more than 35,000 mg/L of dissolved solids) and contain sodium, calcium, bicarbonate, and sulfate as the major dissolved constituents.

Dissolved Solids Are Contributed by Evaporite Minerals

In headwater areas (fig. 3), contact time between the water and rock, or sediment derived from the rock, is generally short, allowing minimal time for reaction with, or dissolution of, the minerals in the rock. In basin settings, Las Vegas Valley and Carson Desert, water tends to have higher concentrations of dissolved solids than in headwater areas owing to the longer time the water has to react with the rock and sediment. Sediments in the basin parts of the NVBR Study Unit also tend to be finer grained than those in headwater areas; thus, they have more surface area upon which the reactions can take place. Evaporite minerals, characteristically quite soluble, are present in parts of Las Vegas Valley and Carson Desert. If these minerals come into contact with water, they may dissolve, adding constituents including sodium, calcium, bicarbonate, and sulfate. In Las Vegas Valley and Carson Desert, the USEPA secondary maximum contaminant level (SMCL) of 500 mg/L for dissolved solids [7] is commonly exceeded.

Arsenic is Derived from Coatings on Sediment Grains

Surface-water samples from Carson Desert, mostly irrigation return flow, and Steamboat Creek, in the Reno-Sparks area, had high arsenic concentrations. Agricultural drainwater and lake samples from Carson Desert had arsenic concentrations as great as 380 and 1,400 µg/L, respectively [8,9]. Steamboat Creek (fig. 3) had arsenic concentrations ranging from 44 to 120 µg/L during 1994 [10].

Ground water in several general areas of the Study Unit had arsenic concentrations that exceeded drinking-water standards. Arsenic concentrations in Carson Desert ground-water samples commonly were high with about 57 percent of the samples exceeding the USEPA maximum contaminant level (MCL) of 50 µg/L [7,11,12]. Previous reports have explained that the high arsenic concentrations are a result of dissolution of arsenic-bearing iron oxyhydroxide coatings on sediment grains [8,11,13] caused by a rise in the water table associated with large-scale irrigation. The ultimate source of arsenic probably is volcanic rocks and sediment derived from volcanic rocks.

Uranium is Dissolved from Granitic Rocks and Sediment

In surface waters within the NVBR Study Unit, uranium concentrations were high in Las Vegas Valley and Carson Desert. Las Vegas Wash (fig. 2), which carries urban runoff and drainage from the Las Vegas urban area, had a median uranium concentration of 24 µg/L during 1993-95 [10,14]. Shallow ground-water seepage into Las Vegas Wash is the most likely source. The proposed MCL for uranium is 20 µg/L [7]. Agricultural drains in Carson Desert had median uranium concentrations that ranged from 8 to 157 µg/L [9]. Water samples collected from five drain systems had uranium concentrations greater than 100 µg/L. Lico [8] noted uranium concentrations in drain-water samples ranging from 2.2 to 470 µg/L with a median concentration of 83 µg/L.

Ground water in Las Vegas Valley, Carson Desert, and Lake Tahoe Basin had uranium concentrations greater than the proposed MCL for drinking water. Five shallow monitoring wells in the water table aquifer in the Las Vegas urban area had uranium concentrations that ranged from 7 to 56 µg/L [12]; the source of this uranium is not known. In the Sierra Nevada, uranium is dissolved from granitic rocks by water rich in carbon dioxide and oxygen [15]. Most of this released uranium is quickly removed from the water by attachment to fracture surfaces and fine-grained sediment and organic material. The resulting uranium-rich sediment grains and organic material have been transported to downstream valleys. Ground-water samples from deep wells on the southern side of Lake Tahoe (fig. 2) had high uranium concentrations ranging from less than 1 to 60 µg/L with a median concentration of 7.4 µg/L. Samples from shallow monitoring wells in the Lake Tahoe area, which had lower uranium concentrations than those from the deep wells, ranged from 0.5 to 16 µg/L with a median concentration of 1.4 µg/L. Shallow (less than 50 ft) groundwater samples from irrigated areas in Carson Desert, collected during the NVBR study, had a median concentration of 40 µg/L [12], which is double the proposed MCL of 20 µg/L. Previous authors have suggested that uranium in Carson Desert ground water is due to the dissolution of

uranium-rich sedimentary organic material and iron and manganese-oxide coatings and was caused by a rise in the water table from large-scale irrigation in the area [11,15]. Ground water near the distal end of the flow system, in Carson Desert wetlands, had uranium concentrations ranging from 1.9 to 1,500 µg/L with a median value of 200 µg/L [8].

Radon is Derived from Granitic and Volcanic Rocks

The Sierra Nevada (fig. 2) has rock types that typically have high uranium content and high radon generating potential. Radon is a radioactive gas derived from the decay of uranium-238 [16]. Granitic rocks have uranium concentrations that are high relative to most rocks. Radon has been reported at high concentrations in ground-water samples from western Nevada and eastern California [10,12,14,16,17,18], mostly where these rock types (fig. 2) and sediment from them are present. Ground-water samples from most of the study area, with the exception of Las Vegas Valley, had median values that exceeded the former proposed MCL [7] (currently under review for possible revision) of 300 picocuries per liter (pCi/L; see table below).

Geothermal Water Has High Concentrations of Arsenic, Boron, and Mercury

Geothermal systems in Carson Desert and Reno-Sparks area add arsenic and boron to ground- and surface-water systems. Active geothermal areas are present in the NVBR Study Unit (fig. 2). A representative water sample from a geothermal aquifer in Carson Desert was reported [22] as having high concentrations of arsenic (130 µg/L) and boron (18,000 µg/L) compared to other ground water in the area. In this area, deeper geothermal water moving upward may affect the quality of shallow ground water [23].

Geothermal systems in the Reno-Sparks area have an effect on the surrounding water quality—arsenic, boron, and mercury concentrations in ground-water samples commonly are high in comparison to other ground water in the area. Arsenic concentrations in water samples from wells have been reported as high as 640 µg/L and from springs as high as 4,000 µg/L [24]. The reported median arsenic concentration for all wells and springs was 970 µg/L [24]. Boron concentrations in water samples from wells and springs, ranged up to 127,000 µg/L with a median concentration of 22,000 µg/L. Mercury is actively being deposited [24,25] as cinnabar (mercury sulfide), which is a possible source of mercury that could contaminate ground- or surface-water resources. Water samples collected at two sites on Steamboat Creek had a median mercury concentration of 3.2 µg/L [26] during 1977. The same report gives values of 1.7 and 2.0 µg/L of mercury in water samples collected from geothermal springs.

Evapotranspiration Concentrates Dissolved Chemical Constituents

Nevada is the driest State in the Nation. Hot summer weather in parts of the Study Unit, namely Las Vegas Valley and Carson Desert, cause high rates of evaporation on open water bodies (lakes and streams) and transpiration by plants, especially when these high temperatures are accompanied by wind. Surface and ground water are affected by these processes directly and indirectly. Evapotranspiration (combined evaporation and transpiration) in the Carson and Truckee River Basins causes dissolved-solids concentrations to increase in a downstream direction. Headwater reaches of the Carson and Truckee Rivers, in the Sierra Nevada and adjacent areas, had median dissolved-solids concentrations of 123 and 86 mg/L, respectively.

Ground-Water Quality

Shallow monitoring wells near the water table and deeper supply wells in principal aquifers in urban areas were sampled to determine the effect of urban activities on ground-water quality [12]. The effects of urban land use can be indicated by detections of nitrate (from sewage or fertilizers) and synthetic organic compounds (pesticides and VOCs). In Las Vegas Valley, 32 shallow monitoring wells and 22 supply wells in underlying, deeper principal aquifers were sampled. In the Reno-Sparks area, 28 shallow monitoring wells and 18 supply wells in underlying, deeper principal aquifers were sampled.

NAWQA samples from principal aquifers in urban areas generally did not contain nitrate or synthetic organic compounds at levels that pose a risk to human health. Other parts of principal aquifers not sampled during this study, however, are contaminated [1]. Small concentrations of synthetic organic compounds were present in many water-supply wells, which indicates that some water from the shallow water-table aquifers is moving into the principal aquifers.

Because the shallow water-table aquifers were contaminated in urban areas of the Study Unit, the potential exists for water and accompanying contaminants to make their way into principal aquifers [12]. Contamination of principal aquifers may be enhanced by certain natural and human-induced factors. Areas where ground water moves downward due to natural conditions or ground-water withdrawals, and where no confining layers separate the shallow water-table and deeper principal aquifers, are susceptible to contamination. Even in areas with a protective confining layer, several mechanisms exist that could compromise the integrity of the confining layers and allow migration of contaminants into the principal aquifers used for water supply. Among these mechanisms are overpumping of an aquifer, causing subsidence and disturbance of a confining layer; breaking of a well casing due to subsidence; improper well construction; and downward movement of water in abandoned or nonpumping wells. Artificial recharge is another mechanism that can move contaminants into ground-water supplies.

Nitrate Was High in the Shallow Aquifers beneath the Las Vegas Urban Area

Nitrate contamination of ground water in urban areas can be caused by excess fertilizer application and by sewage from septic systems or leaky sewer pipes. These sources of contamination commonly affect shallow water-table aquifers [43]. Once the shallow water-table aquifer is contaminated, it becomes a source of contaminants to the deeper principal aquifer.

Shallow water-table aquifers in urban areas within the NVBR Study Unit had a wide range of nitrate concentrations. In Las Vegas, the median nitrate concentration (4.4 mg/L as N) for 32 monitoring wells tapping the shallow aquifer was more than double the national NAWQA median. Twelve percent of the samples equaled or exceeded the nitrate MCL [44] of 10 mg/L as N [7]. In the Reno-Sparks area, nitrate concentrations (median, 0.9 mg/L as N) were about equal to the national median.

NAWQA samples from principal aquifers in urban areas within the NVBR Study Unit had nitrate concentrations that were less than the national NAWQA median. No NAWQA water sample from deeper principal aquifers in urban areas of the NVBR Study Unit exceeded the MCL for nitrate.

Pesticides Were Commonly Detected in Aquifers beneath the Reno-Sparks Urban Area

Forty-seven percent of the water samples from shallow monitoring wells in urban areas (Reno-Sparks and Las Vegas areas) had pesticide detections. Triazine herbicides were the most commonly detected compounds and included atrazine, deethyl atrazine (a degradation product of atrazine), prometon, and simazine. The Reno-Sparks area had a higher frequency of pesticide detections (68 percent) than the Las Vegas area (28 percent). Two or more pesticides were detected in 39 percent of the samples from the Reno-Sparks area [45] and in 9 percent of the samples from the Las Vegas area. All detections of pesticides were at concentrations that did not exceed available MCLs. Pesticide occurrences in samples from wells in shallow water-table aquifers were more prevalent in urban areas than in agricultural areas within the NVBR Study Unit. As a result of the pesticide detections by the NVBR NAWQA study, the Nevada Division of Agriculture is cooperating with the U.S. Geological Survey to develop a network for monitoring pesticides in shallow ground water [46]. The Nevada Division of Agriculture also is reviewing NVBR NAWQA data to evaluate pesticide registrations for Nevada.

In urban land-use areas, 44 percent of the samples from supply wells in the principal aquifer in the Reno-Sparks area and 14 percent from the Las Vegas area contained pesticides at low concentrations (less than 0.1 µg/L). Six different compounds, mostly triazine herbicides, were detected. Two or more pesticides were detected in 22 percent of the supply wells sampled in the Reno-Sparks area. Effects of multiple pesticides at low levels are not known.

VOCs Were Detected in About 50 Percent of the Sampled Wells

More than 50 percent of the samples taken from shallow monitoring wells had detections of one or more VOCs [12]. Three general types of VOCs commonly were detected in shallow ground water: solvents, THMs, and MTBE. Chloroform, a THM, was detected in samples from the Las Vegas (68 percent) and Reno- Sparks (21 percent) urban areas. Samples exceeded MCLs [7] for the chlorinated solvents tetrachloroethylene (three samples) and trichloroethylene (one sample), and two samples exceeded the USEPA drinking water advisory (20-40 µg/L) for MTBE [34].

VOCs were detected in 45 percent of the samples taken from supply wells in principal aquifers in urban areas. Among the 12 compounds detected, chloroform was the most common (39 percent of the samples). Chloroform was detected more commonly in Las Vegas (50 percent of the samples) than in the Reno-Sparks area (28 percent). Other VOCs detected in samples from principal aquifers in urban areas were tetrachloroethylene (13 percent), trichlorofluoromethane (8 percent), and a few other THMs in one or two samples each. Individual THM concentrations ranged from 0.2 to 23 µg/L, and combined concentrations did not exceed the MCL of 80 µg/L [7]. One sample from the principal aquifer beneath the Las Vegas urban area had a tetrachloroethylene concentration (21 µg/L) greater than the MCL of 5 µg/L [7].

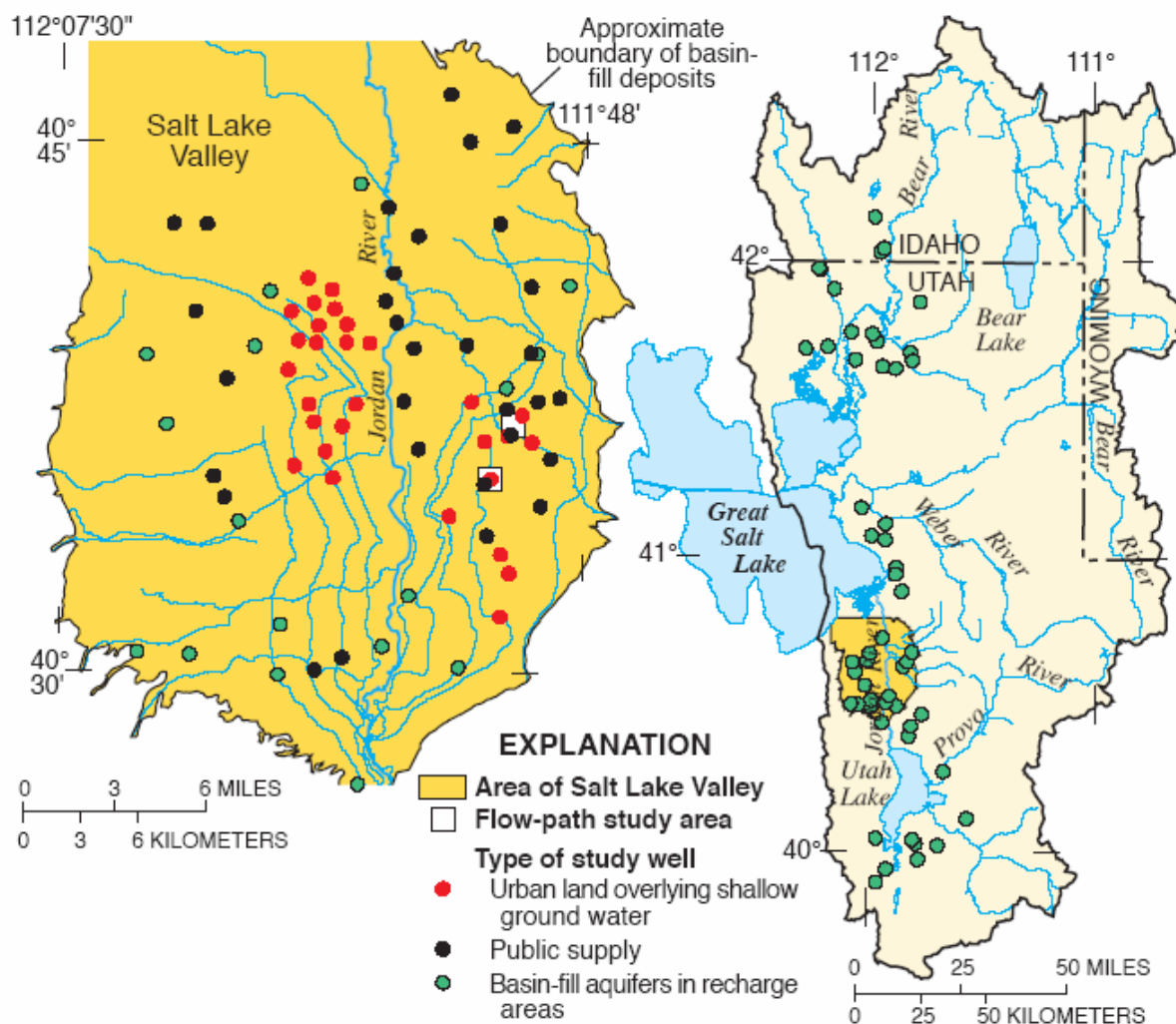
Shallow ground-water quality also may be affected by agricultural use of pesticides. During 1994-95, shallow ground-water samples were collected from 30 monitoring wells beneath agricultural areas in the Carson River Basin as part of the NAWQA study. Five samples from Carson Valley and three from Carson Desert had detections of pesticides at low concentrations (less than 1 µg/L), except for one sample that had 1.2 µg/L of atrazine, which was less than the MCL of 3 µg/L [7]. The most commonly detected pesticides were triazine herbicides, including atrazine, deethyl atrazine (a degradation product of atrazine), and simazine.

GRSL

Water Quality in the Great Salt Lake Basins Utah, Idaho, and Wyoming, 1998–2001

By Kidd M. Waddell, Steven J. Gerner, Susan A. Thiros, Elise M. Giddings, Robert L. Baskin, Jay R. Cederberg, and Christine M. Albano
Circular 1236

<http://water.usgs.gov/pubs/circ/2004/1236/>



Ground water is a major source of public supply in the Great Salt Lake Basins Study Unit. The basin-fill aquifers receive recharge from precipitation on the mountains through subsurface inflow and from infiltration of precipitation, irrigation, or seepage from streams and canals in the valleys. Shallow ground water in Salt Lake Valley contains elevated concentrations of nitrate; and pesticides and VOCs were detected in many samples from the shallow aquifer, indicating that the shallow aquifer is affected by human activities at the land surface. Water from the deeper underlying basin-fill aquifers generally meets existing Federal and State standards and regulations for drinking water.

GRSL Ground-Water Chemistry			
Shallow Ground Water Underlying Urban Areas in Salt Lake Valley	Physical parameters, major ions, selected trace elements, pesticides, VOCs, nutrients, radon, and stable isotopes. Selected samples to estimate ground-water ages using tritium, helium-3, and CFCs. To examine how land use in an urban area affects the quality of ground water in recently recharged shallow aquifers.	Installed 30 monitoring wells in the shallow ground-water system underlying recently developed (less than 30 years) residential and commercial areas in Salt Lake Valley, Utah.	Sampled once in summer 1999
Basin-Fill Aquifers Study	Physical parameters, major ions, selected trace elements, pesticides, VOCs, nutrients, radon, stable isotopes, and tritium. To characterize the water quality in the basin-fill aquifers that are most important for present and future use in the study area.	Included 55 wells (mostly used for domestic supply) in the recharge areas for the deeper basin-fill aquifers within the Great Salt Lake Basins.	Sampled once during July-November 1998
Ground-Water FlowPath Study	Physical parameters, major ions, selected trace elements, pesticides, VOCs, nutrients, radon, and stable isotopes. Selected samples to estimate ground-water ages using tritium, helium-3, and CFCs. To characterize shallow ground-water quality in relation to ground-water flow in a residential area.	Installed 11 monitoring wells along 2 hypothetical flow paths in residential areas in Salt Lake Valley, Utah. Sampled the monitoring wells, 6 nearby existing wells (including 3 land-use study wells) and 1 nearby stream.	Sampled 17 sites in spring 2000. Sampled or resampled 13 sites in fall 2000
Public-Supply Well Study	Physical parameters, major ions, selected trace elements, pesticides, VOCs, nutrients, radon, stable isotopes, and methylene blue active substances. Selected samples to estimate ground-water ages using tritium, helium-3, and CFCs. To characterize water quality in the principal aquifer used for public supply in an urban area.	Included 31 public supply wells, 15 of which represent the largest producers in Salt Lake Valley, Utah.	Sampled once during May-June 2001

Ground-Water Highlights

- The median nitrate concentration of 6.8 mg/L for shallow water from 30 monitoring wells in recently developed residential and commercial areas in Salt Lake Valley is the highest of 34 studies done across the Nation as part of the NAWQA Program. These wells are in areas with high potential for downward movement of water to the underlying basin-fill aquifer used for public supply. (See page 11)
- Tetrachloroethene (PCE) was detected more frequently in shallow ground water underlying residential and commercial areas in Salt Lake Valley than in most ground water sampled by NAWQA across the Nation. PCE was detected in water from 16 of 30 monitoring wells. Water from one monitoring well exceeded the USEPA drinking-water standard of 5 µg/L (micrograms per liter) for PCE, although water from the shallow aquifer is not used for drinking. (See pages 14 and 15)
- VOCs were frequently detected in water from the basin-fill aquifers in both agricultural and urban areas. Chloroform, likely formed as a byproduct of water disinfection, was the most frequently detected VOC, but at concentrations much less than the USEPA drinking-water standard. (See page 15)
- Atrazine was the most frequently detected pesticide (77 percent of samples) in shallow ground water underlying Salt Lake Valley, but mostly at concentrations much less than the USEPA drinking-water standard of 3 µg/L. Atrazine was detected in 23 percent of samples collected from wells in the basin-fill aquifers throughout the Study Unit. (See page 15)
- The age (time since recharge) of water from 31 public-supply wells in Salt Lake Valley ranged from 3 to more than 50 years. Manmade compounds generally were detected in water that was less than 50 years old, but not in water more than 50 years old. (See page 16)• Concentrations of arsenic in water from about 20 percent of wells (primarily domestic wells) in the basin-fill aquifers underlying recharge areas exceeded the USEPA drinking-water standard of 10 µg/L. Arsenic in basin-fill aquifers likely is derived naturally from geochemical reactions between aquifer minerals and water. (See page 21)

Physical Setting and Climate

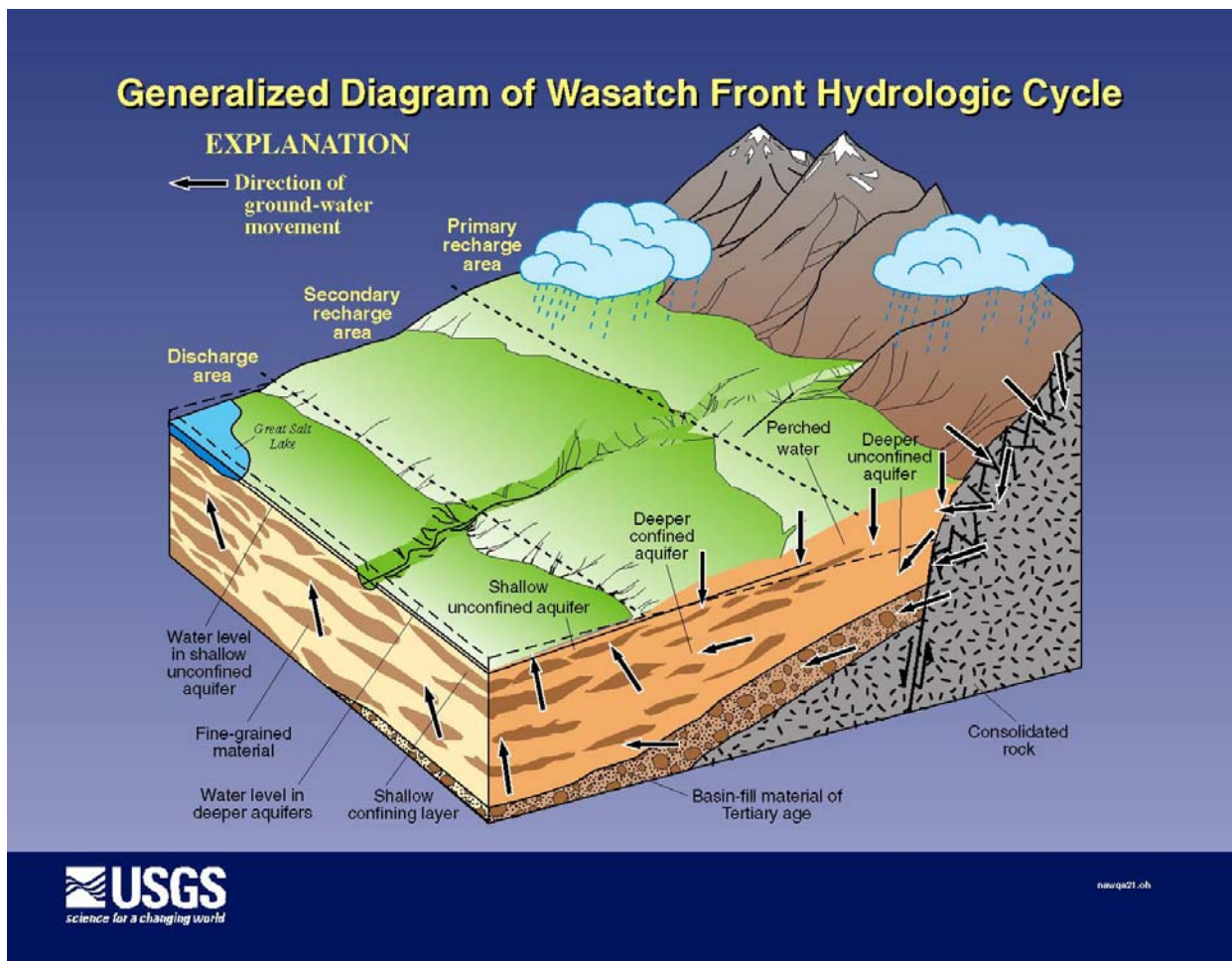
The Great Salt Lake Basins *Study Unit* includes about 14,500 square miles of northeastern Utah, southeastern Idaho, and southwestern Wyoming and three major river systems that enter Great Salt Lake: the Bear, Weber, and Utah Lake-Jordan River systems (Baskin and others, 2002). Altitude ranges from about 4,200 feet near Great Salt Lake to more than 12,000 feet in the Uinta Mountains near the eastern edge. About 1.7 million people or about 76 percent of the population in the State of Utah (2000 census) live along the western flanks of the Wasatch Mountains, where Utah's largest cities are located (fig. 1). The climate in the Study Unit is typical of mountainous areas in the Western United States. Temperature generally fluctuates widely between summer and winter and between day and night. The high mountains have long, cold winters and short, cool summers. The lower valleys are more moderate, with less variance between maximum and minimum temperatures. Average monthly maximum temperature reaches 92°F in July at the Salt Lake City International Airport, and the average monthly minimum reaches -1.5°F in January at Sage, Wyoming. Most of the annual precipitation falls as snow, which provides most of the annual runoff during the spring as snowmelt. Average annual precipitation ranges from less than 16 inches on the valley floors to greater than 70 inches in the high mountain areas.

Natural factors of physiography, geology, soils, climate, and hydrology largely determine the natural background quality of water, and cultural factors of population, land and water use, and water- and waste-management practices define the human influence on water quality. Hydrologic modifications of streams, along with *point* and *nonpoint* sources of contaminants, have had detrimental effects on the quality of ground-water and surface-water ways.

Rangeland covers about 48 percent of the Study Unit and forest and agricultural land covers 22 and 18 percent, respectively (Hitt, 1994). Urban land accounts for 4 percent of the total land cover, and the remaining 8 percent is distributed between wetlands (4 percent), water (3 percent), and bare ground (1 percent) (fig. 5). Mining occurred in several of the canyons of the Wasatch Mountains within a 30-mile radius of Salt Lake Valley. Land use and associated human activities alter the natural character of streams by changing water quality, channel shape, and flow characteristics. In headwater areas, logging, mining, and, more recently, recreation and residential construction have increased erosion and changed water quality. In the valleys, agriculture, grazing, and urbanization have removed many of the natural wetlands and *riparian* vegetation; channelized stream reaches in some areas; and contributed nutrients, sediments, and other contaminants to streams.

Ground Water

Ground water is contained within unconsolidated basin-fill deposits in the basins and in consolidated rocks in the mountains (fig. 6). The unconfined part of the basin-fill *aquifers*, which is the primary *recharge* area, contains the water table (or uppermost saturated zone in the subsurface) and is the most vulnerable to contamination from overlying land-use activities. The lack of *confining layers* near the mountain fronts results in relatively deep unconfined aquifers (water levels generally greater than 100 feet below land surface). Mountain-front areas are becoming increasingly developed for residential and commercial uses. Where confining layers exist in the subsurface, shallow unconfined aquifers (generally about 50 feet below land surface) overlie the *confined aquifers* (Thiros, 2003a). Water from the adjacent and overlying unconfined deposits recharges confined parts of the basin-fill aquifers. Although separated from the land surface by confining layers, the confined aquifers also are susceptible to contamination from the shallow and deeper unconfined aquifers by flow reversals caused by large withdrawals of ground water. The basin-fill aquifers are a major source of water for domestic and municipal supply and for irrigated agriculture.



Pumping likely affects the distribution of dissolved solids in the basin-fill aquifer in Salt Lake Valley

Pumping from wells accounts for about one-third of the total estimated discharge from the basin-fill aquifer in Salt Lake Valley, Utah, an urban area in the Study Unit (seepage to streams and evapotranspiration are the other major forms of discharge). Most ground-water withdrawals occur on the eastern side of the valley because of higher well yields (discharge rates) and lower dissolved-solids concentrations. In some areas on the western side, ground water is withdrawn and blended with water from other sources to improve its quality.

Comparison of water in the basin-fill aquifer sampled in 1988–92 and that sampled in 1998–2002, shows an increase in the extent of the aquifer containing water with a dissolved-solids concentration greater than 500 mg/L. Water with less than 500 mg/L dissolved solids extended to the northwest past the Jordan River in 1988–92, but was replaced by water with concentrations of dissolved solids greater than 500 mg/L by 1998–2002 (fig. 13). Concentrations of dissolved solids increased in water from some wells near the center of the valley, where the direction of ground-water movement (gradient) is naturally upward between the basin-fill aquifer and land surface. Withdrawals from the basin-fill aquifer for public supply may have caused gradients to reverse, allowing more mineralized water that was recharged in the valley to move to the aquifer.

Nitrate levels generally are low in the basin-fill aquifers

Concentrations of *nitrate* in water from the basin-fill aquifers used for water supply throughout the Study Unit generally were low—at or less than 2 mg/L—which is considered a naturally occurring or “background” concentration on the basis of available data for undeveloped areas across the Nation (U.S. Geological Survey, 1999). Only four samples exceeded the USEPA *drinking-water standard* for nitrate (10 mg/L). Water from these four wells is used for irrigation and stock watering, and not for drinking. The highest measured concentration of nitrate (27.9 mg/L) was in a sample from a well in an agricultural area (fig. 17).

The presence of clay confining layers between the land surface and the basin-fill aquifers impedes the movement of water and any included contaminants to the underlying aquifer. The median concentration of nitrate in water from the confined part of the basin-fill aquifers was 1.5 mg/L. Concentrations of nitrate were highest in wells in the primary recharge areas (fig. 6) of the basin-fill aquifers (relative to wells in the more downgradient, confined parts). Specifically, the median concentration of nitrate was 3.1 mg/L in the primary recharge areas of the aquifers, which was the seventh highest value for 94 major drinking-water aquifers studied by NAWQA across the Nation.

Although none of the sampled domestic-supply wells in the basin-fill aquifers contained nitrate at concentrations that exceeded the drinking-water standard, the elevated concentrations above background indicate that the basin-fill aquifers are susceptible to human activities at the land surface. A chemical analysis of the water is generally required before a domestic-supply well can be connected to a residence, but subsequent monitoring typically is not required. It may be important for domestic well owners to assess activities around their wells and to monitor nitrate in their wells if concentrations are elevated or changing.

Pesticides and VOCs are detected in basin-fill aquifers underlying all types of land use

Pesticides in water from the basin-fill aquifers throughout the Study Unit were detected more often and at higher concentrations in samples collected from wells in agricultural areas than in areas dominated by rangeland or urban land uses (fig. 23). Detections of pesticides in water from the aquifers consisted predominantly of the herbicides atrazine and its *degradation product* deethylatrazine (DEA), prometon, and simazine. Atrazine and simazine are selective herbicides commonly used on crops and orchards in agricultural areas and to control vegetation in industrial areas. Prometon is used primarily along road rights-of-way and urban corridors. Atrazine and DEA were detected in 23 and 31 percent, respectively, of samples collected from wells in the basin-fill aquifers. Many of the samples with detections were collected from wells in primary recharge areas that historically have been undeveloped or used for agricultural purposes but have become increasingly urbanized.

VOCs were detected in 63 percent of the ground-water samples collected from the basin-fill aquifers throughout the Study Unit. Wells surrounded by predominantly urban and agricultural land had a slightly higher frequency of detection, 66 and 63 percent respectively, than wells located in rangeland (54 percent) areas (fig. 23). The two most frequently detected VOCs in the basin-fill aquifers were the THMs chloroform and bromodichloromethane, which were detected in 37 and 19 percent of samples, respectively.

The detected pesticide and VOC concentrations were well within USEPA drinking-water standards (U.S. Environmental Protection Agency, 2002b) and are not a known health concern; however, their widespread occurrence in the aquifers indicates the presence of water recharged recently enough to be affected by human activities.

VOCs and pesticides commonly are detected in shallow ground water and in the basin-fill aquifer used for public supply in Salt Lake Valley

At least 1 VOC was detected in water from 27 of 30 monitoring wells completed in the shallow aquifer beneath recently developed residential and commercial areas of Salt Lake Valley (Thiros, 2003b; see inset on page 14). The most frequently detected VOCs were THMs (chloroform and bromodichloromethane) and those used as solvents (PCE and 1,1,1-trichloroethane). The widespread occurrence of THMs at low concentrations in shallow ground water is likely a result of chlorinated public-supply water used to irrigate lawns and gardens in residential areas of Salt Lake Valley that has recharged the aquifer. Water disinfected for public supply also can enter the ground-water system through leaking water lines, sewer lines, and swimming pools. PCE was detected in water from 16 monitoring wells completed in the shallow aquifer at concentrations up to 7.8 µg/L, exceeding the USEPA drinking-water standard of 5 µg/L. PCE is used primarily as a dry-cleaning agent and a solvent. The herbicides atrazine and its degradation product DEA were the most frequently detected pesticides in water from the shallow aquifer in Salt Lake Valley and were found in 77 and 70 percent of the samples, respectively. Atrazine concentrations in water from the monitoring wells ranged from less than 0.001 to 1.58 µg/L (fig. 24), less than the USEPA drinking-water standard of 3 µg/L. All but one of the water samples from the western side of the valley contained detectable concentrations of atrazine. The high detection frequency of atrazine in shallow ground water underlying primarily residential areas on the western side of the valley may be the result of past application in agricultural or industrial areas that have been converted to residential uses. Also, atrazine applied in nonurban areas may have been transported to urban areas in the Study Unit by ground water. The four wells with the highest concentrations of atrazine and its degradation products also had minimal concentrations of chloroform, which likely comes from watering lawns with chlorinated water. This correlation suggests that the recharge area for these wells includes more agricultural or nonirrigated industrial and vacant land than residential land.

The compounds detected in shallow ground water and used at the land surface have the potential to move to the deeper basin-fill aquifer in Salt Lake Valley, which is used for public supply. VOCs and (or) pesticides were detected, mostly at low concentrations, in water from 23 of 31 public-supply wells sampled in the valley. Although the concentration of these compounds measured in ground water used for public supply is not a known health concern according to current standards, their occurrence in the deeper ground water presents the possibility that water with higher concentrations may enter the aquifer in the future.

Most ground water used for public supply in Salt Lake Valley was recharged in the past 50 years and contains manmade compounds

Elevated nitrate concentrations and the occurrence of pesticides, VOCs, and other manmade compounds in the deeper basin-fill aquifer underlying Salt Lake Valley indicate relatively recent (within the past 50 years) recharge of water to the ground-water system (Thiros and Manning, 2004). The apparent age for water sampled from public-supply wells in the valley ranges from 3 to more than 50 years.¹ Ground water is generally older with distance from the mountain front, the oldest water being in the discharge area toward the center of the valley (fig. 25). Concentrations of manmade compounds generally decreased with the age of water and were not detected in water older than 50 years from many of the public-supply wells. For example, summed concentration of VOCs and pesticides generally was in the range of 0.1 to 10 µg/L in water younger than 20 years of age and was less than 0.01 µg/L in water about 50 years or more in age (fig. 26). Similarly, nitrate concentrations were lower in the older ground water. Manmade compounds were detected or chlorofluorocarbon (CFC) contamination occurred in water from five wells with shallow, open intervals in ground-water discharge areas within Salt Lake Valley. This indicates that the aquifer in the discharge area can receive modern water recharged in the valley despite the natural upward gradient.

Minerals in rocks are sources of arsenic and radon in ground water

Concentrations of arsenic in shallow and basin-fill aquifers in the Great Salt Lake Basins Study Unit were highly variable, ranging from less than 0.4 to 95 µg/L, with a median of 2.1 µg/L (fig. 36). Arsenic, a naturally occurring trace element in ground water within the Study Unit, is derived from geochemical reactions with certain types of minerals. Tertiary volcanic rocks and the Salt Lake Formation exposed near the base of the Oquirrh Mountains may be a source of arsenic on the western side of the Salt Lake Valley

that is not present on the eastern side. Water from wells in the western and northwestern parts of the valley generally had higher arsenic concentrations than water from wells in other areas. The median concentration of arsenic in water from relatively shallow monitoring wells, many of which are on the western side of the valley, was 7.3 µg/L, with 43 percent of the samples exceeding the USEPA drinking-water standard of 10 µg/L. Concentrations of arsenic were generally lower in water from public-supply wells completed in the basin-fill aquifer underlying the valley (median of 1.1 µg/L with 16 percent exceeding the standard) because many of these wells are on the eastern side of the valley. About 20 percent of the wells sampled in the recharge areas of the basin-fill aquifers throughout the Study Unit exceeded the drinking-water standard. Water from most wells sampled is used for domestic supply (provides drinking water to a small number of families). The Safe Drinking Water Act does not regulate domestic wells and therefore homeowners may not be aware of possible risks associated with elevated arsenic concentrations in their wells.

Quality and Sources of Ground Water Used for Public Supply in Salt Lake Valley, Salt Lake County, Utah, 2001

By Susan A. Thiros and Andrew H. Manning
Water-Resources Investigations Report 03-4325

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Description of Study Area

Salt Lake Valley is an urban area bounded by the Wasatch Range, Oquirrh Mountains, Traverse Mountains, and Great Salt Lake (fig. 1). It is near the transition between the Basin and Range and Rocky Mountain Physiographic Provinces (Fenneman, 1931). The valley is about 28 mi long and 18 mi wide (about 500 mi²) and generally corresponds to the populated part of Salt Lake County, which contains the Salt Lake City metropolitan area. The population of 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 1,029,000 in 2010 and 1,233,000 in 2020 (Wasatch Front Regional Council, 2000), which will increase water demand for public supply. Because the natural boundaries of the valley restrict expansion of residential areas beyond the extent of agricultural land, population growth will occur mainly through increased population density.

The climate in Salt Lake Valley is semiarid, with a 1961-90 average annual precipitation of about 10 to 20 in. (Daly and Taylor, 1998). Lawns and gardens in the valley require irrigation to supplement precipitation during the growing season. Precipitation in the mountains is about 50 in/yr. Mountain streams draining the Wasatch Range discharge into the Jordan River, which flows north along the axis of the valley, is diverted into canals, and eventually discharges into Great Salt Lake.

Land and Water Use

Urban areas within the valley more than doubled from 1960 (89 mi²) to 1994 (198 mi²) (Utah Department of Natural Resources, Division of Water Resources, 1999). Land use in urban areas of the valley in 1994 was about 63 percent residential, 6 percent commercial, 14 percent industrial, and 17 percent other (includes open spaces, idle spaces, transportation, and utilities) (Utah Department of Natural Resources, Division of Water Resources, 1995). Much of the industrial land use in Salt Lake Valley is near the Jordan River (fig. 2). Historically, the urban area was centered in the northeastern part of the valley in a recharge area. Many of the recently developed residential/commercial areas are along the mountain front bounding the east side of the valley and are also replacing agricultural areas on the west side (fig. 2).

Surface water from streams draining the Wasatch Range provided about 70 percent of the water used for public supply in Salt Lake Valley in 2000. This water is chlorinated and distributed across the valley. Demand for water peaks during July through August, when lawns and gardens require more water because of the summer heat. Water systems that receive surface water also use wells during the summer to meet the increased demand. Water systems without surface-water sources rely on water from wells throughout the year. Groundwater withdrawal in 2000 from wells in the valley was about 28 percent of the public supply. Springs and tunnels in the Wasatch Range provided a minor amount (about 2 percent) of the valley's public supply.

An average of about 79,000 acre-ft/yr was withdrawn from 161 public-supply wells in the valley during 1996-99 (Utah Division of Water Rights, written commun., 2001). The average well withdrawal for this period is shown in figure 3. Most of the ground-water withdrawal occurs on the east side of the valley because of higher yields and lower dissolved-solids concentrations. In some areas, water is blended with water from other sources to improve its quality.

Ground-Water Hydrology

Salt Lake Valley contains basin-fill deposits derived from the surrounding mountains that are deposited mainly in alluvial fans, stream channels, deltas, and lacustrine features associated with Lake Bonneville and other paleolakes. Lake Bonneville covered much of northern Utah about 15,000 years ago. Tertiary-age deposits are semiconsolidated to consolidated and consist mainly of alluvial-fan deposits interbedded with volcanic ash and tuffs. These deposits crop out along the western and southern margins of the valley and are overlain by generally unconsolidated sediments of Quaternary age that are considerably more permeable.

The saturated Quaternary deposits range from less than 200 ft thick along the margins of the valley to more than 1,000 ft thick in the northern part of the valley (Hely and others, 1971). Nearly all the wells in the valley are screened in the Quaternary deposits. Lake deposited clay layers occur throughout the valley, except near the mountain fronts where coarser-grained deposits exist.

A generalized model of the saturated basin-fill deposits in Salt Lake Valley consists of a relatively deep unconfined aquifer near the mountain fronts that becomes confined toward the center of the valley by layers of fine-grained deposits (fig. 4). Collectively, the deeper aquifers are known as the principal aquifer. Where the principal aquifer is confined, it is overlain by a shallow unconfined aquifer. The primary recharge area for the principal aquifer includes the mountains surrounding the valley and the part of the valley near the mountain fronts where there are no substantial layers of fine-grained deposits to impede the downward movement of water.

The secondary recharge area is where downward movement of water from the shallow aquifer to the deeper confined aquifer is possible because a downward gradient exists and confining layers are thin, and (or) discontinuous. In the discharge area an upward gradient exists from the deeper confined aquifer to the overlying shallow aquifer. The recharge and discharge areas in Salt Lake Valley (fig. 5) were mapped by Anderson and others (1994, p. 6) on the basis of lithologic and water-level data.

Depending on the continuity of the confining layers, the shallow ground water overlying these layers may be either localized in extent (perched) or laterally continuous forming an aquifer. The shallow aquifer is susceptible to contamination from activities related to land use because of its proximity to land surface. Water quality in the confined part of the principal aquifer can be degraded by secondary recharge of contaminated water from the shallow aquifer. The unconfined part of the principal aquifer also is vulnerable because of a lack of confining layers that can

impede the downward movement of contaminated ground water. The principal aquifer in Salt Lake Valley has been divided into areas where the susceptibility of ground water to contamination is similar on the basis of differences in geology, the rate of ground-water movement, and the direction of vertical hydraulic gradients (Waddell and others, 1987a). Recharge to and discharge from the principal aquifer in the valley has been estimated in previous studies by Hely and others (1971), Waddell and others (1987b), and Lambert (1995) (table 1). Mountain-front recharge is thought to comprise about 50 percent of recharge to the principal aquifer and includes subsurface inflow from the adjacent mountains (mountain block recharge) and seepage from streams near the mountain front. Infiltration of unconsumed irrigation water from fields, lawns, and gardens; infiltration of precipitation on the valley floor; and seepage from canals are the other major sources of recharge. Model simulations of the ground-water flow system were used to better understand the distribution of these recharge components in the valley (Waddell and others, 1987b; Lambert, 1995). Seepage through the shallow aquifer to the Jordan River and other streams and to evapotranspiration are large components of discharge from the principal aquifer. Total withdrawals from wells for all uses is about one-third of the total estimated discharge from the system and affects the amount of ground water available to other forms of discharge.

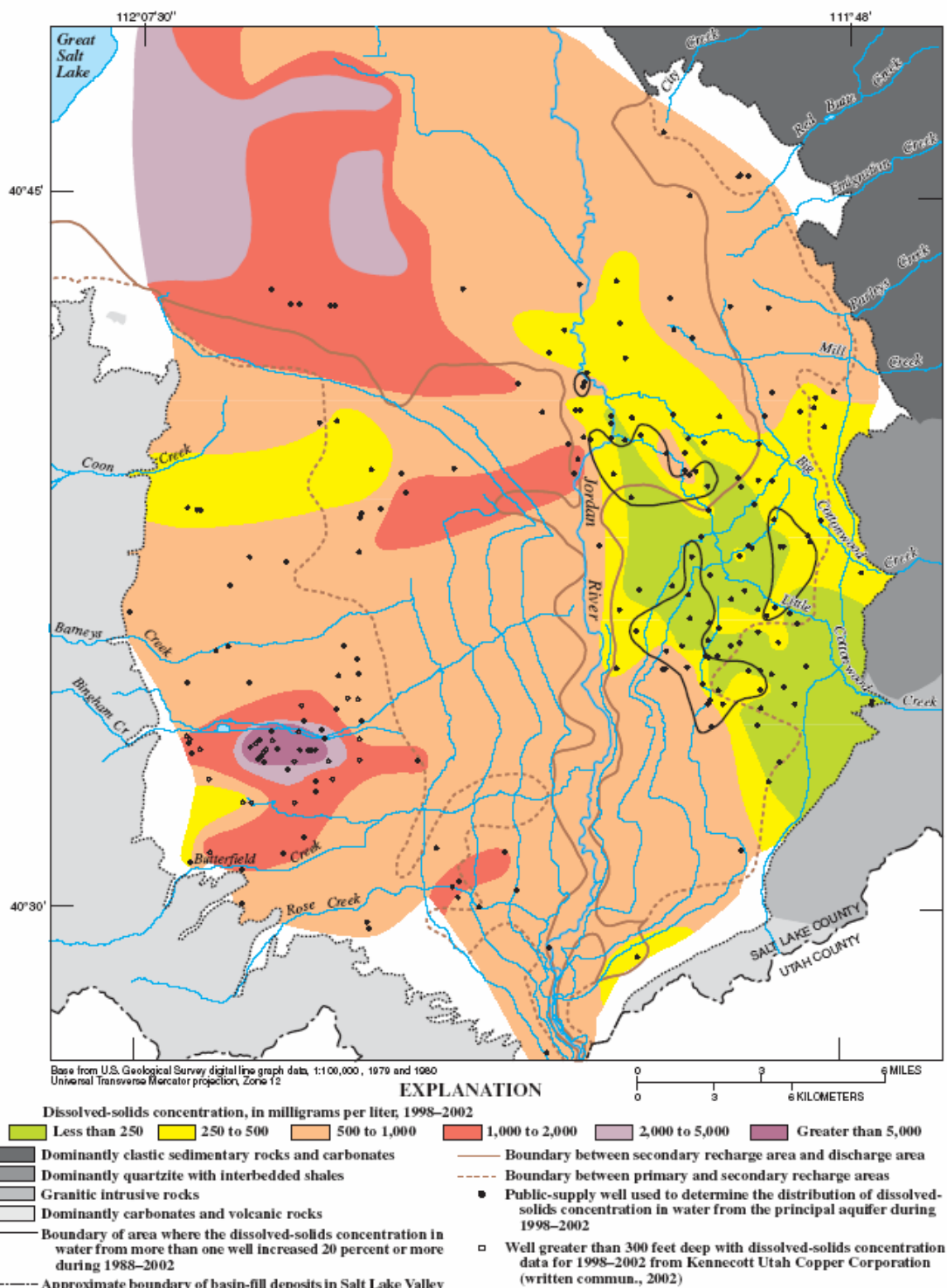


Figure 8. Dissolved-solids concentration in water sampled from the principal aquifer in Salt Lake Valley, Utah, 1998–2002.

RELATION BETWEEN GROUND-WATER QUALITY AND SOURCES OF RECHARGE

Pesticides and (or) VOCs were detected, mostly at very low concentrations, in water from 23 of the 31 public-supply wells sampled. Produced and used exclusively by humans, pesticides and VOCs are known as anthropogenic compounds. Although measured concentrations of these compounds are not a health concern, their widespread occurrence indicates the presence of water young enough to be affected by humans in much of the principal aquifer. Detection of these compounds in water from a well indicates the possibility that water with higher concentrations may enter the well in the near future. Wells with water samples 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 method reporting levels (LRLs or MRLs) are shown in figure 26. Also shown are wells that contain water with nitrate concentrations greater than 2 mg/L, an estimated background level. The source of nitrate concentrations above the background level is probably the application of fertilizers and leaking or improperly functioning septic systems and sewer pipes in the valley. Wells with water that contains anthropogenic compounds above reporting levels and (or) nitrate concentrations above 2 mg/L are referred to below as “affected wells.” Eighteen of the 31 public-supply wells sampled (58 percent) are affected wells. The fact that a well is affected confirms that it has a reasonably high level of susceptibility.

The presence of anthropogenic compounds at concentrations above reporting levels and elevated nitrate concentrations in the principal aquifer (fig. 26) generally is not well correlated with the distribution of minimum MFR (fig. 18) and minimum MBR (fig. 20) fractions of water derived from $\delta^{18}\text{O}$ and T_r data, respectively. Many of the affected wells have high MFR and MBR fractions of water, indicating that wells containing even small amounts of valley recharge (probably 10 to 20 percent) are reasonably susceptible. Only two of the wells sampled for anthropogenic compounds fall into the highest susceptibility group according to both $\delta^{18}\text{O}$ and T_r data: well 31 in the Murray zone and well 20 in the Midvale zone. Well 31 is affected, but well 20 is not. It should be noted, however, that $\delta^{18}\text{O}$ data indicate that the Midvale zone may extend farther to the north and southeast to include wells 1 and 21, and both of these wells are indeed affected. Therefore, the minimum MFR and MBR fractions may still be useful tools for identifying those highest susceptibility areas where wells are clearly drawing in a substantial amount of valley recharge.

The presence of anthropogenic compounds and elevated nitrate concentrations in the principal aquifer (fig. 26) is very well correlated with the distribution of interpreted-age categories (fig. 23 and 26). Nearly all of the affected wells (17 of 18) have either dominantly modern water or a mixture of modern and pre-bomb waters, meaning that they probably contain more than 20 percent modern water. Well 23 is the only affected well that contains dominantly pre-bomb water. Well 23 is also the only affected well that does not contain anthropogenic compounds above reporting levels. Most of the unaffected wells (10 of 13) contain dominantly pre-bomb water and thus contain little modern water (less than 20 percent). All of the wells (10 of 10) with dominantly modern water are affected. Seventy percent (7 of 10) of the wells with dominantly modern or a mixture of modern and pre-bomb waters are affected. Nine percent (1 of 11) of the wells with dominantly pre-bomb water are affected. These results indicate that much of the modern ground water in Salt Lake Valley contains anthropogenic-compound concentrations above reporting levels or nitrate concentrations greater than the estimated background level of 2 mg/L. As expected, pre-bomb water generally is free of these human effects. Therefore, $^3\text{H}/^3\text{He}$ data appear to be of great value in evaluating well susceptibility in the principal aquifer.

Wells that contain water contaminated with at least one CFC and with CFC concentrations indicative of recharge prior to 1955 (low CFC concentrations) are also shown in figure 26. Eleven of the 14 wells that contain CFC-contaminated water also are affected wells, indicating that the

presence of CFC contamination may be a relatively useful indicator of high susceptibility. Six of the seven wells that contain water with low CFC concentrations are not affected, indicating that low CFC concentrations may be a reliable indicator of low susceptibility. Well 23 is the single affected well (elevated nitrate concentrations) that has water with low CFC concentrations (indicative of recharge prior to 1955). This is consistent with the $3\text{H}/3\text{He}$ results that indicate dominantly pre-bomb water from well 23.

The relation between the occurrence and concentration of anthropogenic compounds, nitrate, and other variables was examined with the Spearman rank correlation test (Helsel and Hirsch, 1992, p. 209-218). The correlation coefficient (ρ) varies from 1 to -1 and describes the strength of the correlation. Values closer to 1 indicate a positive correlation, values closer to -1 indicate an inverse correlation, and values closer to 0 indicate no correlation. The p-value (probability of no correlation) was used to evaluate the significance of the correlation. P-values less than 0.10 are considered statistically significant, whereas larger p-values indicate a greater probability of no correlation. The number of anthropogenic compounds detected (includes estimated concentrations less than LRL or MRLs) in water sampled from each well is inversely correlated with the apparent $3\text{H}/3\text{He}$ age (fig. 27). Anthropogenic compounds were not detected in water with an apparent age older than 50 years, except for water from well 25. Interpretation of the trace amount of atrazine detected in water from this well ($0.004\text{ }\mu\text{g/L}$) and application to the aquifer as a whole should be done with caution because of the large depth to the well openings (636-1,202 ft below land surface). Nitrate concentration in water from the 31 sampled public-supply wells is correlated with many factors, such as depth to the top of the well's open interval, the $\delta^{18}\text{O}$ ratio of the water, the apparent age of the water determined from the $3\text{H}/3\text{He}$ method, and the number of anthropogenic compounds detected in water per well. 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 $\delta^{18}\text{O}$ ratios became heavier and, therefore, the MFR fraction became smaller; as the apparent age of the water became younger; and as the number of anthropogenic compounds detected in water per well increased (fig. 27). On the basis of these correlations, nitrate concentration in water from many of the wells is related to the occurrence of modern valley recharge.

The concentration of anthropogenic compounds in a water sample was considered to be the sum of all VOC and pesticide compounds detected. For most wells where VOC or pesticide compounds were detected in the water, total concentrations decreased as the apparent $3\text{H}/3\text{He}$ age increased (fig. 28). Generally, wells with dominantly pre-bomb water had very low or undetected total concentrations of anthropogenic compounds. All of the samples that contained mostly modern water contained at least one anthropogenic compound. Anomalously high concentrations of chloroform, bromodichloromethane, and chlorodibromomethane were measured in water samples from wells 8 and 13, likely the result of chlorinated surface water occasionally injected into the principal aquifer at these wells as artificial recharge. Note, however, that these wells are only used as injection wells for a short duration each spring, and the injected water constitutes a very small fraction of the total volume removed from these wells since the initiation of the artificial recharge program in 2000. Well 26 is the only well containing modern or a mixture of re-bomb and modern waters in which no anthropogenic compounds were detected. Total concentration of VOCs and pesticides in water generally decreased as the depth to the top of the open interval in the well became deeper (fig. 28). Artesian conditions exist at all six of the wells where the top of the open interval is less than 150 ft below land surface. Anthropogenic compounds or CFC contamination occur in water from five of these wells, indicating that wells with shallow openings are susceptible to receiving recently recharged water despite the natural upward gradient in a discharge area. Human activities in the adjacent recharge area may be the source of these compounds in water from wells 10, 14, and 30. Wells 27 and 31 are farther from

the primary recharge area and CFC contamination in water samples from these wells may be the result of a reversal in the water-level gradient near the wells.

Areas with an increase in dissolved-solids concentration greater than 20 percent from 1988-92 to 1998-2002 (figs. 7 and 8) generally correspond to areas with relatively large components of modern water (fig. 23), except for near the Jordan River west of the Murray zone and in the Sandy zone. Although in the discharge area, water sampled from wells 1 and 27 contained tritium and CFC contamination indicating mixing with a modern component of water. Withdrawals from the principal aquifer in the area may have allowed recently recharged water to move downward into the aquifer in the vicinity of these wells. Ground water with dissolved-solids concentrations greater than 500 mg/L has extended farther into the Sandy zone during the last decade. As discussed in previous sections, water in this area is isotopically lighter than water from the Jordan River, has a warmer recharge temperature than ground water to the east, and was recharged before the 1950s. Lateral movement of ground water with higher dissolved-solids concentrations from the west and southwest may have been induced by withdrawals in the area.

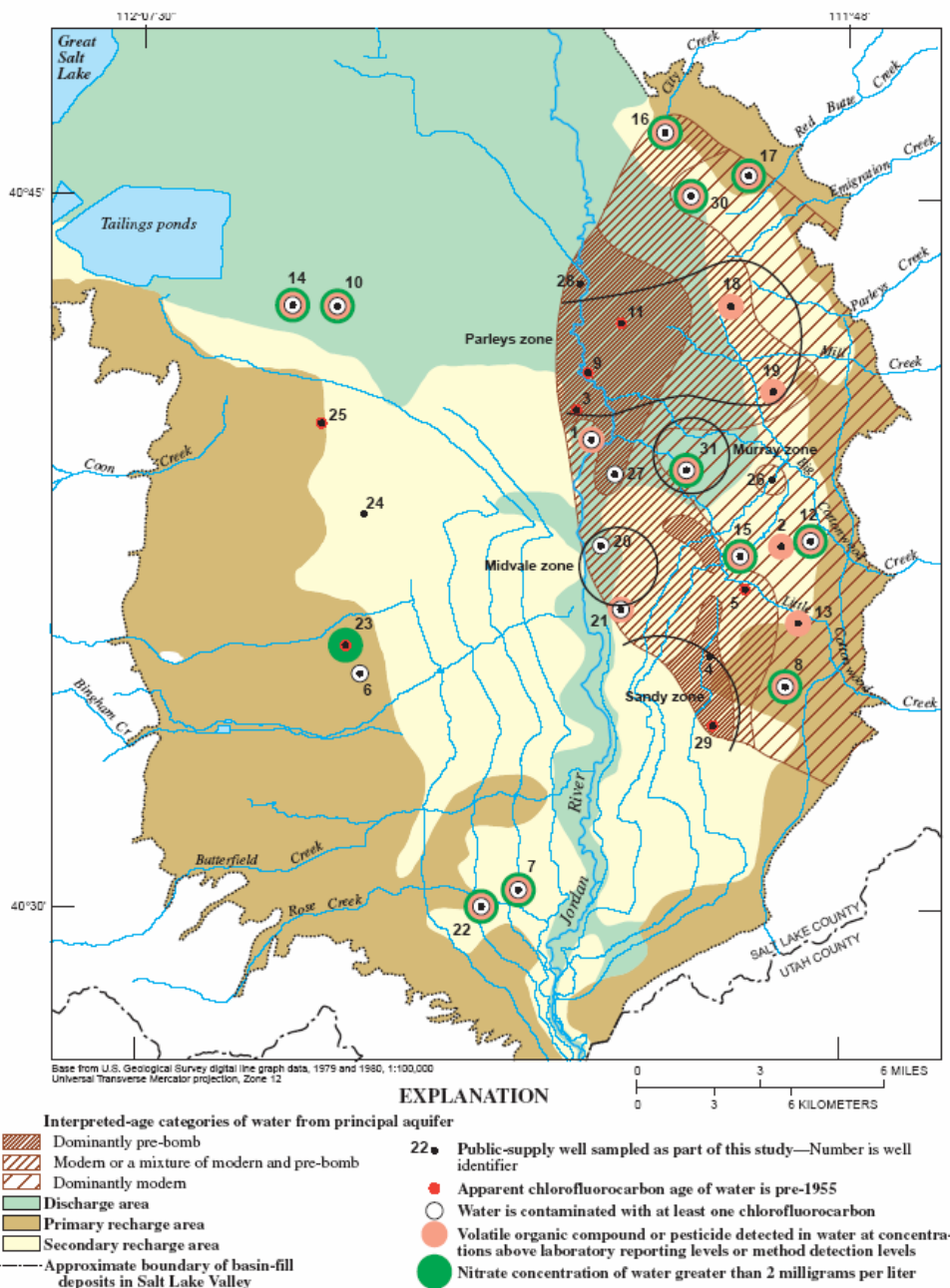


Figure 26. Interpreted-age category, chlorofluorocarbon, anthropogenic-compound, and nitrate information for water sampled from 31 public-supply wells in Salt Lake Valley, Utah, 2001.

SUMMARY

Ground water supplies about one-third of the water used by the public in Salt Lake Valley, Utah. The purpose of this study is to evaluate the occurrence and distribution of natural and anthropogenic compounds in ground water used for public supply in the valley and to determine the general sources of recharge to the principal aquifer. Water samples were collected from 31 public-supply wells. Water-quality data for the wells consist of analyses of field parameters, major ions, trace elements, radon, nutrients, dissolved organic carbon, methylene blue active substances, pesticides, and volatile organic compounds. Dissolved-solids concentration ranged

from 157 to 1,280 mg/L in water from the 31 public-supply wells. Comparison of water sampled from the principal aquifer during 1988-92 and 1998-2002 shows a reduction in the area where the dissolved-solids concentration is less than 500 mg/L. Ground water with a dissolved-solids concentration less than 500 mg/L in an area extending to the northwest, past the Jordan River, receded to the southeast likely because of pumping in the area and in upgradient areas to the southeast. Dissolved arsenic concentrations in water from five wells exceeded the drinking-water maximum contaminant level of 10 µg/L. Water from wells in much of the western part of the study area generally had higher arsenic concentrations than water from other areas. Nitrate concentration in water sampled from 12 of the 31 public-supply wells was higher than an estimated background level of 2 mg/L, indicating a likely human influence.

At least one pesticide or pesticide degradation product was detected at concentrations much lower than year 2002 U.S. Environmental Protection Agency drinking-water standards in water from 13 of the 31 wells sampled. The relatively high detection frequency of atrazine in water samples from the west side of the valley may be the result of application in agricultural or industrial areas that also are recharge areas for the aquifer. The most frequently detected volatile organic compounds were chloroform (54.8 percent), bromodichloromethane (35.5 percent), and 1,1,1-trichloroethane (19.4 percent). The widespread occurrence of chloroform and bromodichloromethane in deeper ground water is likely a result of the recharge of chlorinated public-supply water used to irrigate lawns and gardens in residential areas of Salt Lake Valley.

Environmental tracers were used to determine the sources of recharge to the principal aquifer used for public supply in the valley. The stable isotopes of water (deuterium and oxygen-18) and recharge temperatures computed from dissolved noble gases in the ground water were used to differentiate between mountain and valley recharge. Tritium, helium-3, and chlorofluorocarbons were used to identify and date ground water recharged within the past 50 years. Because the local population resides in the valley, along with most of the local industry, water that enters the ground-water system in the valley has a greater likelihood of containing anthropogenic compounds than does mountainblock recharge.

Maximum recharge temperatures in the eastern part of the valley generally are below the range of valley water-table temperatures indicating that mountainblock recharge must constitute a substantial fraction of recharge to the principal aquifer in this area. Together, the recharge temperature and stable-isotope data define two zones that contain large fractions of valley recharge on the east side of the valley. To determine if a sample contains a substantial fraction of pre-bomb water (more than 50 years old) the initial tritium concentration of each sample measured tritium plus measured tritiogenic helium-3) is compared to that of local precipitation at the apparent time of recharge. Three interpreted-age categories were determined for water from the sampled wells: (1) dominantly pre-bomb; (2) dominantly modern; and (3) modern or a mixture of pre-bomb and modern. Apparent tritium/helium-3 ages range from 3 years to more than 50 years. Ground water on the east side of the valley generally becomes older with distance from the mountain front, the oldest water being in the discharge area. Generally, wells with apparent ages for water less than 20 years contain dominantly modern water, wells with apparent ages for water of 20-40 years contain modern water or a mixture of modern and pre-bomb waters, and wells with apparent ages for water greater than 40 years contain dominantly pre-bomb water.

Chlorofluorocarbon-age data were collected for the 31 wells sampled during this study. Water sampled from nearly half of the wells was contaminated with at least 1 of the CFCs analyzed. Contamination levels of CFCs are potentially useful in gauging well susceptibility because wells with CFC-contaminated water most likely contain an appreciable component of modern water.

The presence of anthropogenic compounds at concentrations above reporting levels and elevated nitrate concentrations (affected wells) in the principal aquifer is very well correlated with the distribution of interpreted-age categories. All of the wells (10 of 10) with dominantly modern water are affected. Seventy percent (7 of 10) of the wells with dominantly modern or a mixture of modern and pre-bomb waters are affected. Only 9 percent (1 of 11) of the wells with dominantly pre-bomb water are affected. Both the number and total concentration of anthropogenic compounds detected in water sampled from each well is inversely correlated with the apparent $^3\text{H}/^3\text{He}$ age. Anthropogenic compounds were not detected in water with an apparent age of more than 50 years, except for water from one well. All of the samples made up mostly of modern water contained at least one anthropogenic compound.

Areas with an increase in dissolved-solids concentration greater than 20 percent from 1988-92 to 1998-2002 generally correspond to areas with relatively large components of modern water. Although in the discharge area, water sampled from two wells did contain tritium and CFC contamination, an indication of mixing with a modern component of water. Withdrawals from the principal aquifer in the area may have allowed recently recharged water to move downward into the aquifer in the vicinity of these wells.

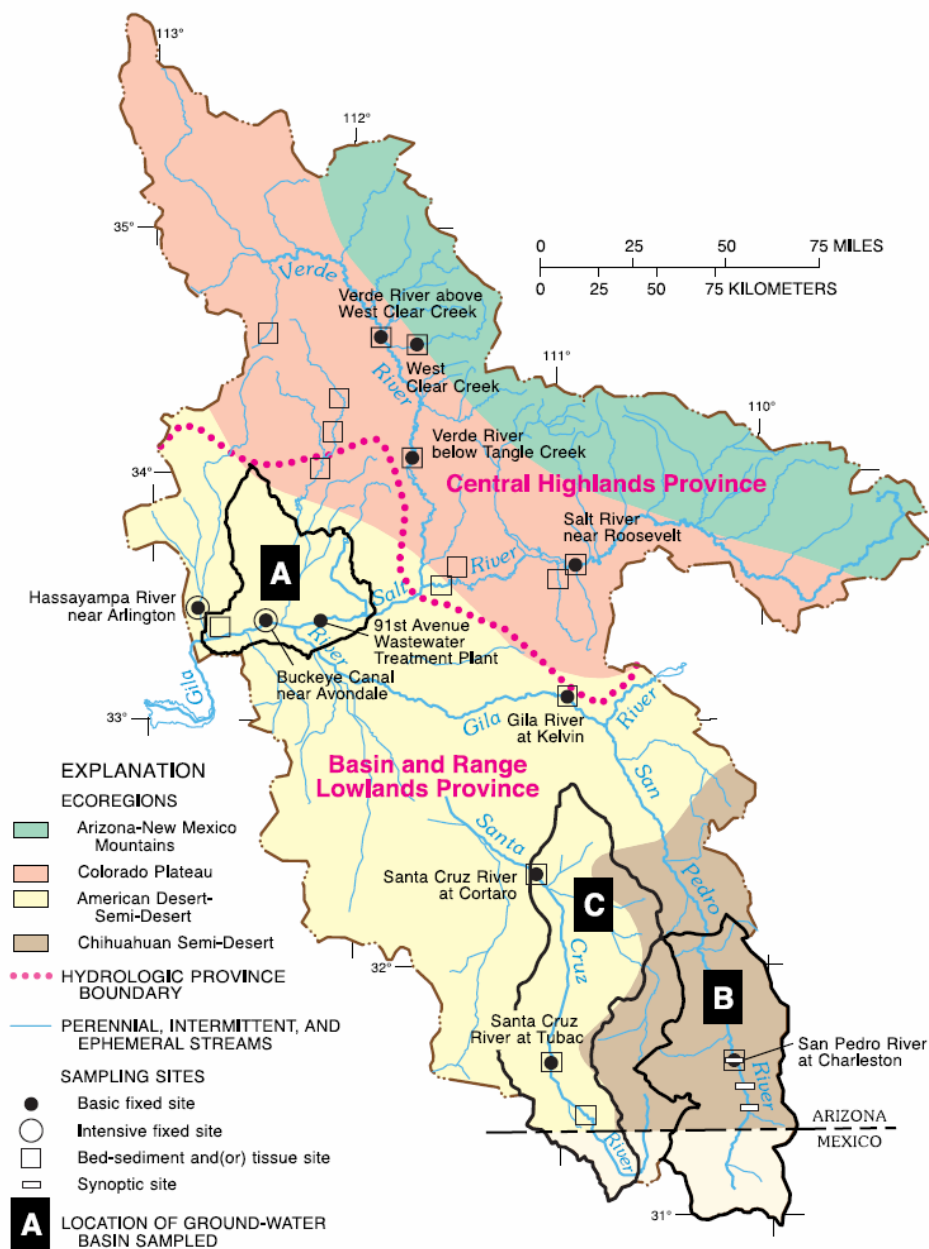
CAZB

Water Quality in the Central Arizona Basins, Arizona, 1995–98

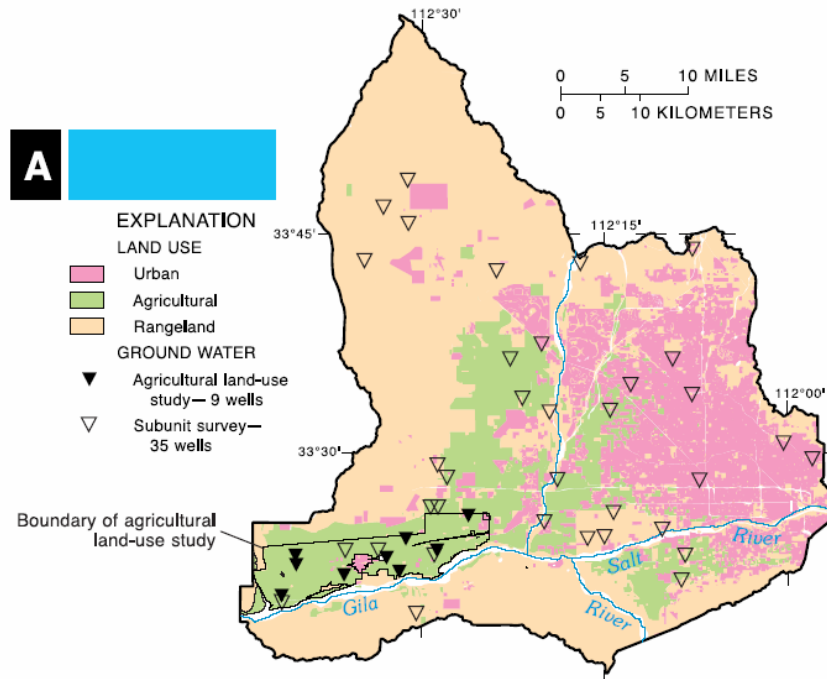
By Gail E. Cordy, D.J. Gellenbeck, Joseph B. Gebler, David W. Anning, Alissa L. Coes, R.J. Edmonds, Julie A. H. Rees, and H.W. Sanger

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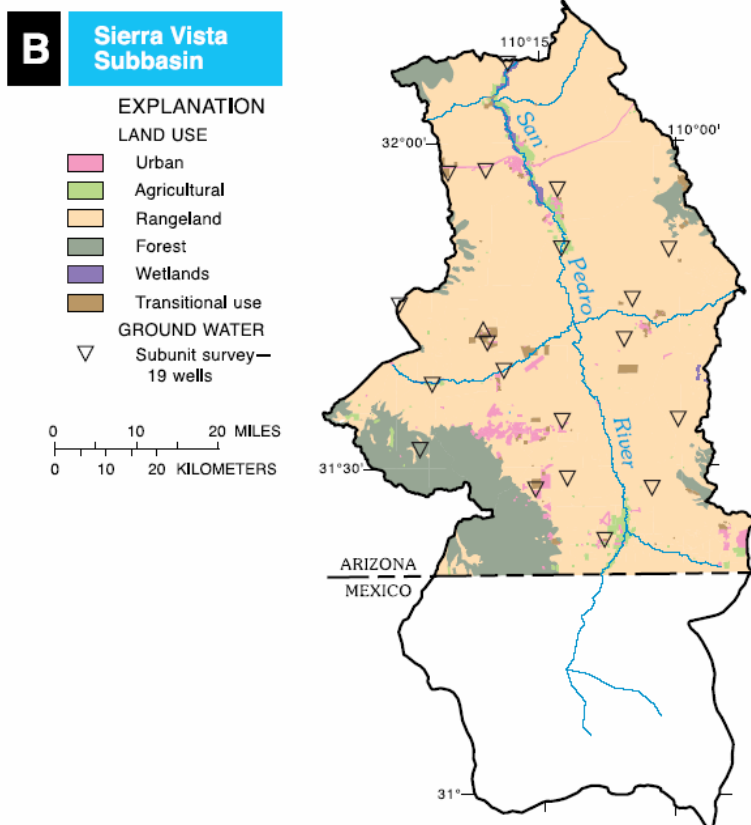
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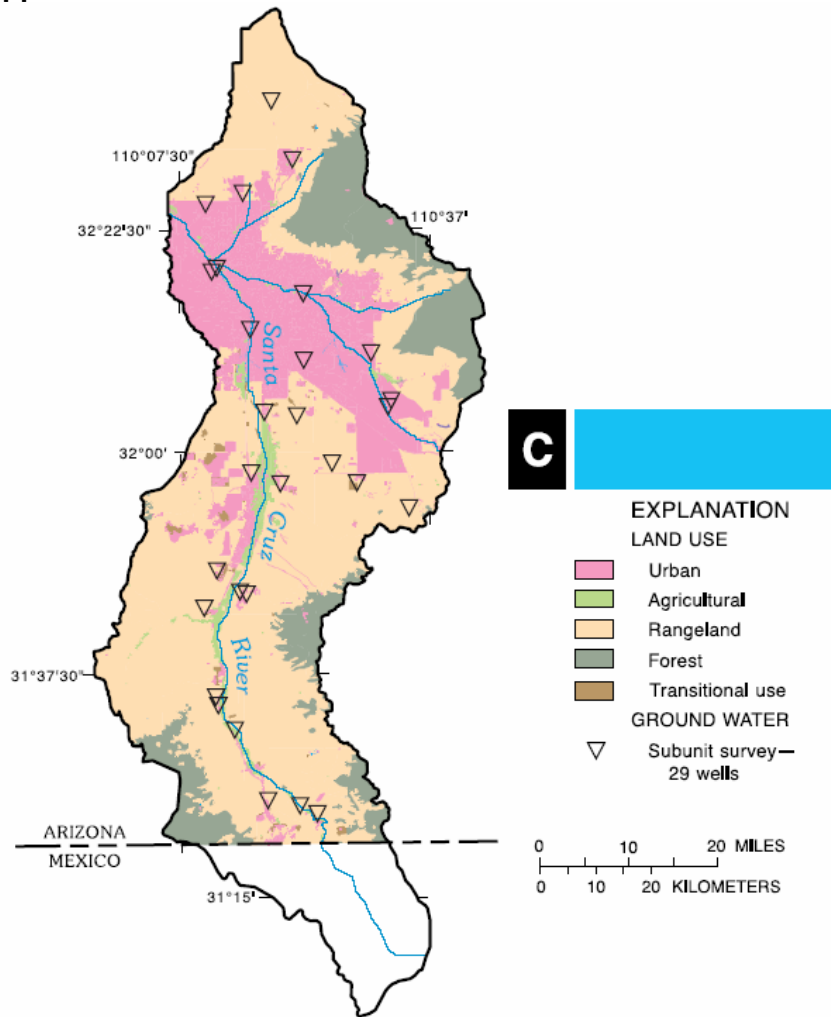
West Salt River Valley



Sierra Vista Subbasin



Upper Santa Cruz Basin



CAZB Ground-Water Chemistry				
Study Unit West Salt River Valley - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, livestock, and industrial wells.	35	Once 1996–1997
Study Unit Upper Santa Cruz Basin - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, livestock, and industrial wells.	29	Once 1998
Study Unit Sierra Vista subbasin - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, and livestock wells.	19	Once 1996
Land use West Salt River Valley - agricultural	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to determine effects of agricultural land use on shallow ground-water quality.	Shallow monitoring wells.	9	Twice Aug. 1997 Feb. 1998

Ground-Water Highlights

Most of the ground water used in the CAZB Study Unit is pumped from basin-fill aquifers in the Basin and Range Lowlands. Water from major aquifers (basinwide) in the West Salt River Valley (WSRV), the Upper Santa Cruz Basin (USCB), and the Sierra Vista subbasin (SVS) generally meets existing USEPA standards and guidelines for drinking water with some exceptions. Nitrate and dissolved-solids concentrations in some samples from the WSRV and USCB exceeded USEPA drinking-water standards and guidelines.

Shallow ground water from an agricultural area in the WSRV exceeded drinking-water standards and guidelines for nitrate and dissolved solids in more than 75 percent of samples. More than 90 percent of groundwater samples from the three basins exceeded the USEPA's proposed drinking-water standard for radon. A small percentage of samples exceeded drinking-water standards for arsenic, fluoride, and molybdenum.

Samples from urban and agricultural areas contained low concentrations of numerous chemicals (pesticides and volatile organic compounds) that can be linked to household, industrial, and agricultural uses.

- Most of the deep wells yield old ground water that generally has not been affected by land uses in the last 50 years.
- Use of fertilizers and treated wastewater on agricultural and urban lands and the evaporation of irrigation water have resulted in the accumulation of nitrate and dissolved solids in shallow ground water.
- Adoption of draft or proposed USEPA drinking-water regulations for arsenic, radon, and uranium—constituents that occur naturally in the study area—will require most water suppliers and municipalities to treat their water to remove these constituents or find alternative supplies.
- Pesticides detected in ground-water basins with substantial agricultural and (or) urban development did not exceed USEPA drinking-water standards and guidelines.

Though trends in ground-water quality over time were not determined for the CAZB Study Unit, the data indicate possible future changes. As urban land use spreads with the growing population in the area, ground-water quality is likely to deteriorate, as indicated by detections of pesticides and volatile organic compounds in urban areas. Nitrate and dissolved solids accumulating in shallow ground water in the WSRV have the potential to degrade the quality of deeper drinking-water supplies.

The Central Arizona Basins (CAZB) Study Unit encompasses a 34,700-square-mile area in central and southern Arizona and northern Mexico (fig. 1). The Study Unit includes large parts of two hydrologic provinces—the Central Highlands in the north and the Basin and Range Lowlands in the south (U.S. Geological Survey, 1969). Climate, hydrology, geology, land use, and water use are distinctly different in these two provinces.

The Basin and Range Lowlands (fig. 1) are characterized by a lack of perennial streams, the largest water demands, and reliance on ground water. Deep, broad alluvial basins separated by mountain ranges of small areal extent characterize this hydrologic province. The basins are filled with thick deposits of gravel, sand, silt, and clay and include interbedded evaporite deposits and volcanic rocks in places (Anderson and others, 1992). These basin-fill sediments can be 2,000 feet to as much as 12,000 feet thick and constitute the major aquifers that are often referred to as “basin-fill aquifers.” The basin-fill aquifers contain large reserves of ground water that were recharged when Arizona’s climate was much wetter than at present, possibly thousands of years ago.

Ephemeral streams are characteristic of the Basin and Range Lowlands (fig. 2). Very little natural streamflow is generated because the average annual rainfall is less than 10 to 15 inches except at the highest elevations. With the exception of some small, higher elevation streams and sections of the San Pedro River, most perennial streams in the Basin and Range Lowlands are effluent dependent; that is, their flow is sustained all year by treated wastewater (fig. 2). Effluent-dependent streams have beneficial uses. They support riparian and aquatic communities where those communities would not otherwise exist. By recharging effluent, cities can accrue “credits” toward pumping of ground water from other locations in a basin (Arizona Department of Water Resources, 1994). Rangeland is the predominant land use in the Basin and Range Lowlands. The two largest urban areas—Phoenix and Tucson—account for about 5 percent of the land use and include 75 percent of Arizona’s 4.9 million people (Arizona Department of Economic Security, rev. July 7, 2000). Agricultural development, which is mostly west and south of Phoenix, is about 5 percent of the land use (Cordy and others, 1998). Cropland is the primary agricultural land use, and cotton is the main crop.

Water use in the Basin and Range Lowlands represents 96 percent of all water use in the CAZB Study Unit (Cordy and others, 1998). Agriculture is the largest water user (73 percent in 1990; fig. 3). Because of the general lack of surface-water resources in the Basin and Range Lowlands, ground-water is relied upon heavily to meet agricultural and municipal demands (fig. 3). In areas with substantial agricultural and (or) urban development, ground water has been and continues to be used more quickly than it can be replenished naturally.

Ground-water levels have declined several hundred feet in areas with the heaviest pumping, and land subsidence has resulted in a loss in aquifer storage capacity (Arizona Department of Water Resources, 1994). To mitigate some of the problems caused by overpumping of ground water, Colorado River water is delivered to central Arizona by the Central Arizona Project (CAP) canal (fig. 2). CAP water is used for aquifer recharge and municipal and agricultural purposes.

Ground water was sampled from wells in three alluvial basins in the Basin and Range Lowlands—the West Salt River Valley, the Upper Santa Cruz Basin, and the Sierra Vista subbasin. Existing wells were sampled in the three basins to assess overall water quality as well on water quality. In the West Salt River Valley, shallow monitoring wells were installed and sampled to determine the effects of irrigated agriculture on shallow groundwater quality. Existing groundwater-quality data were used to assess overall water quality in alluvial basins of the Basin and Range Lowlands that were not sampled.

Streamflow in the Basin and Range Lowlands is difficult to characterize because it is controlled by dams and (or) wastewater-treatment plants. The Hassayampa River near Arlington is an example of a Basin and Range Lowlands stream that is a combination of effluent and irrigation return flows most of the time, supplemented by flows from storm runoff (fig. 4).

Natural Ground-Water Quality

It is important to understand how natural processes affect ground-water quality in order to identify the effects of urban and agricultural development under similar hydrogeologic conditions. In the CAZB, the majority of ground-water basins do not have significant urban or agricultural development. The ground-water quality in these basins is primarily a product of natural processes such as the interaction of ground water with rocks and sediment in the basins (Robertson, 1991).

Natural sources of dissolved solids and nitrate can control ground-water quality in basins with minimal urban development. Specific-conductance values (an indirect measure of the dissolved-solids concentration) and nitrate concentrations for ground water in basins with minimal urban development increase northwestward from southeastern Arizona toward the central part of the State (figs. 16 and 17). The increasing specific-conductance values can be attributed to a corresponding increase in evaporite deposits in basin sediments from southeast to northwest (Gellenbeck and Coes, 1999). Evaporite deposits in the basins contain minerals such as halite (salt) and gypsum that can be easily dissolved in ground water (Robertson, 1991). The increasing nitrate concentrations can be largely attributed to natural sources; however, human activities such as agriculture can be a source in some basins. In some locations in the CAZB, high nitrate concentrations in ground water reported prior to any agricultural or urban development indicate that natural sources of nitrate are present in some basins (Hem, 1985; Robertson, 1991; Gellenbeck, 1994; Gellenbeck and Coes, 1999). Dissolution of evaporite deposits, decay of buried organic matter, precipitation, weathering of rocks and soils, and fixation by microorganisms are just a few of the possible sources of naturally occurring nitrate in ground water.

Effects of Human Activities on Ground-Water Quality

The contamination of major aquifers is largely controlled by hydrology and land use (U.S. Geological Survey, 1999). In the CAZB Study Unit, deep ground water that was recharged prior to 1953 typically has not been affected by human activities (see p. 18). In areas with recent recharge (after 1953), ground water is more likely to be contaminated by nutrients and man-made chemicals associated with urban and agricultural land uses.

Ground-water quality deteriorates in irrigated areas. Irrigation water that seeps downward is a principal source of ground-water recharge in irrigated areas of the CAZB. Dissolved-solids concentrations in seepage can be as much as five times those in the original irrigation water (Bouwer, 1990) because of concentration by evaporation and plant use (see p. 9). The greater the dissolved-solids concentration in the applied irrigation water, the greater the concentration in the seepage moving downward to the ground water.

To determine the effects of irrigated agriculture on shallow groundwater quality, nine monitoring wells were drilled and sampled in the southwestern part of the West Salt River Valley (see "Study Unit Design," p. 26). Because the average depth to ground water in the nine wells is 32 feet (table 1) compared to 230 feet for wells sampled basinwide, irrigation seepage does not have to travel far to reach the shallow ground water in the agricultural area. Sources of irrigation water in this area include treated sewage effluent, water from the Salt River and CAP canal, irrigation return flows, and ground water. Dissolved-solids concentrations of these sources range from about 900 mg/L for treated sewage effluent (Tadayon and others, 1998) to 650 mg/L for CAP water and 470 mg/L for Salt River water (Salt River Project, 1997). The median dissolved-solids concentration in water from the nine shallow wells exceeded 3,000 mg/L (table 1). In addition, the effects of nitrate from fertilizer applications and reuse of irrigation return flows were evident from the median nitrate concentration that was nearly twice the MCL of 10 mg/L (table 1).

Study area	Median concentration, in milligrams per liter		Average depth to ground water, in feet
	Nitrate	Dissolved solids	
<i>West Salt River Valley</i>			
<i>Agricultural area</i>	19.0	3,050	32
<i>Basinwide</i>	2.7	560	230
<i>Upper Santa Cruz Basin</i>	1.5	305	230
<i>Sierra Vista subbasin</i>	0.78	262	171
<i>U.S. Environmental Protection Agency drinking-water standard</i>	¹ 10	² 500	

¹ Maximum Contaminant Level.

² Secondary Maximum Contaminant Level.

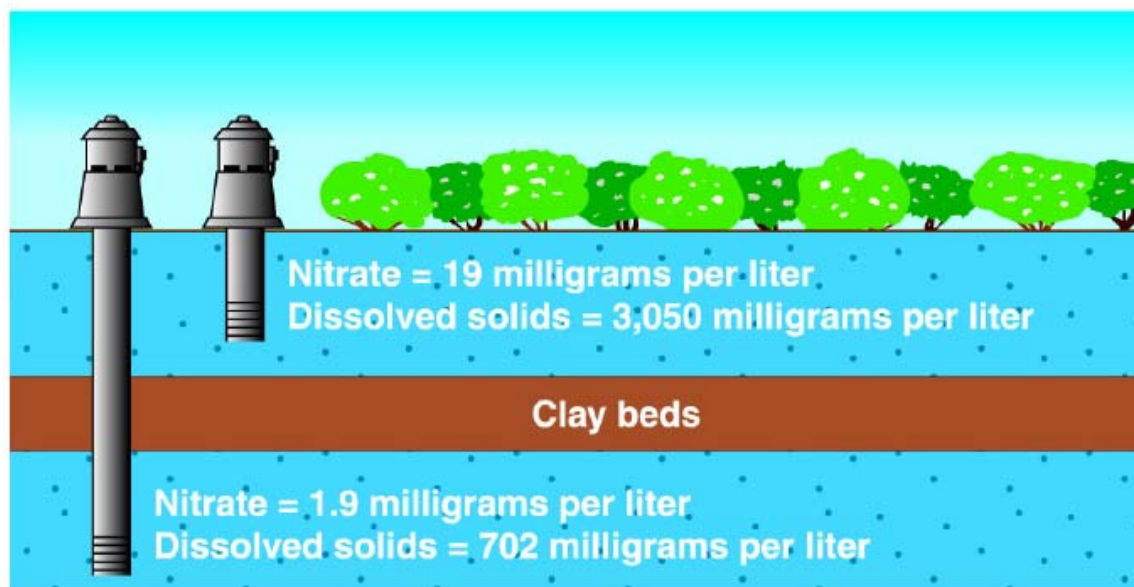
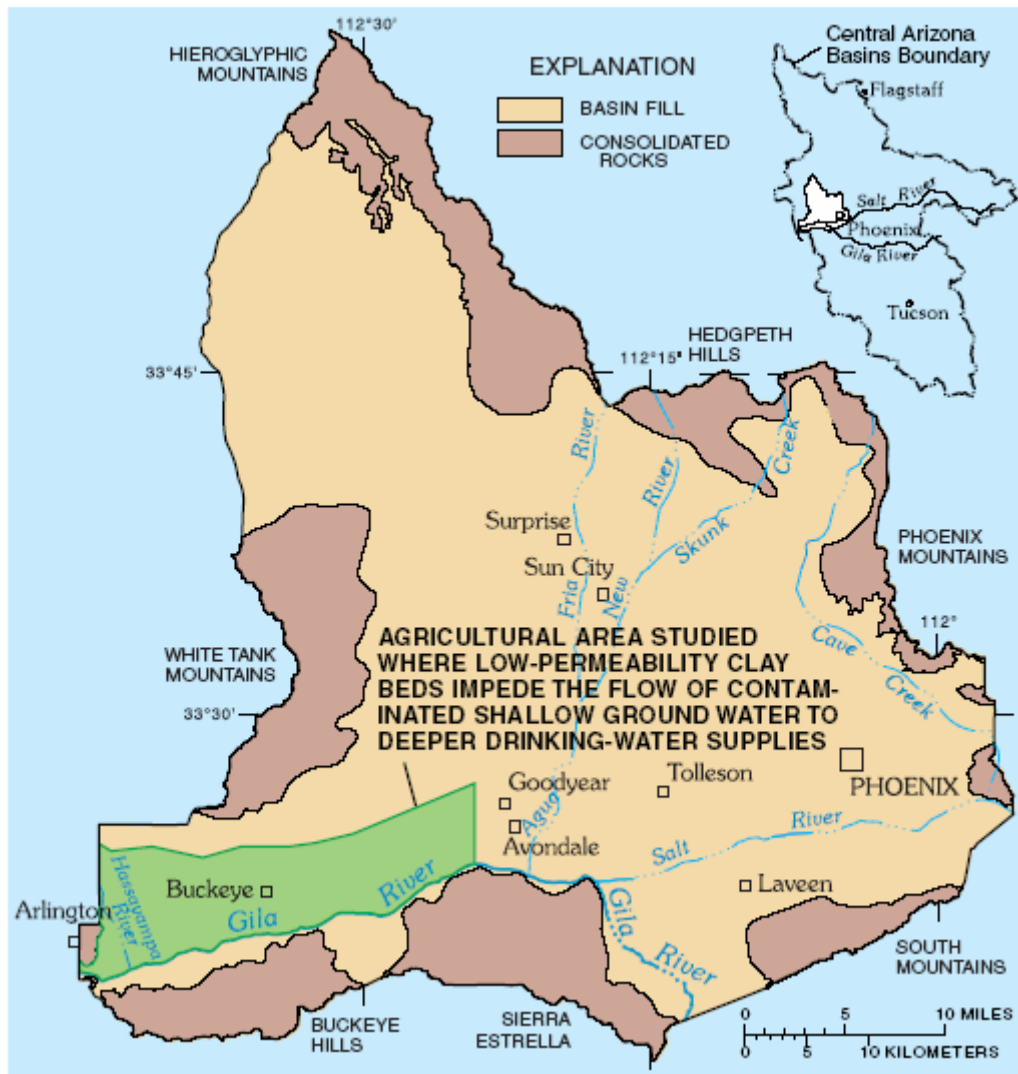
The highest concentrations of nitrate and dissolved solids were in shallow ground water beneath an irrigated agricultural area. Shallow ground water from the agricultural land-use study area in the West Salt River Valley had median concentrations of nitrate (19 mg/L) and dissolved solids (3,050 mg/L) that exceeded the USEPA MCL and SMCL, respectively (table 1). Nitrate and dissolved solids from irrigation and agricultural practices are accumulating in shallow ground water (see p. 9 and 11). The shallow ground water in this area is not used for drinking water, and clay beds reduce the likelihood of contamination of the aquifers below that are used for drinking water (see p. 22).

Deeper ground water from urban, rangeland, and agricultural areas in other parts of the West Salt River Valley had a median nitrate concentration that was less than the MCL of 10 mg/L; however, the median concentration of dissolved solids exceeded the SMCL of 500 mg/L (table 1). Median concentrations of nitrate from the Upper Santa Cruz Basin and the Sierra Vista subbasin also were less than the MCL, and median concentrations of dissolved solids were less than the SMCL (table 1).

Clay beds that currently protect deep ground water from contamination may not do so in the future.

In the agricultural land-use study area of the West Salt River Valley (fig. 20), the tops of low-permeability clay beds are about 150 to 400 ft below the land surface. These clay beds impede the downward movement of irrigation seepage and reduce the likelihood of contaminants reaching deeper drinking-water supplies. Domestic wells in the area yield water from beneath the protective clay beds. Ground water above the clay beds has higher nitrate and dissolved-solids concentrations than ground water from beneath the clay beds (fig. 21). In this area, ground-water samples from above the clay beds had a median dissolved-solids concentration of 3,050 mg/L and a median nitrate concentration of 19.0 mg/L (table 1). Groundwater samples from below the clay beds had a median dissolved-solids concentration of 702 mg/L and a median nitrate concentration of 1.9 mg/L. Care must be taken in drilling and completing drinking-water wells below the clay beds to ensure that shallow ground water above the clay beds does not contaminate the well and aquifer below.

Analyses of the tritium from ground water in this area indicated that water above the clay beds generally had been recharged after 1953, and water below the clay beds generally had been recharged before 1953 (see information about age dating ground water on p. 18). Although the clay beds currently reduce the likelihood that irrigation seepage will contaminate the ground water below, future large-scale withdrawals of ground water from below the clay beds could possibly result in the movement of shallow, poor quality water through the clay beds and into the domestic ground-water supply.



Occurrence and distribution of pesticides in ground water in the CAZB reflect both agricultural and urban land uses. Ten pesticides were detected in shallow ground water from the agricultural land-use study area in the West Salt River Valley, west of Phoenix (fig. 22). In other parts of the West Salt River Valley, consisting of agricultural, urban, and rangeland areas, eight pesticides were detected in ground water. Five pesticides were detected in ground water from the Upper Santa Cruz Basin, where there is a mixture of land-use types, but 60 percent of the basin is undeveloped rangeland (Coes and others, 2000). In the Sierra Vista subbasin, where urban and agricultural land uses are minimal (3.3 percent of basin; Coes and others, 1999) and have been minimal in the past, no pesticides were detected in ground-water samples. During 1996–98, the largest quantities of pesticides used among the three basins were for agriculture in the West Salt River Valley (Ken Agnew, University of Arizona, Pesticide Information and Training Office, written commun., 1999).

In particular, the shallow depth to ground water in the agricultural land-use study area means that irrigation seepage and recharge, containing pesticides and their breakdown products, do not have to travel far to contaminate the ground water. Clay layers impede the movement of pesticides into the deeper aquifers in the area. The soils in the agricultural area have been identified as a source of DDE for the ground water (Brown, 1993). The only detection of DDE in the West Salt River Valley outside of the agricultural area was in a sample from the northern part of the Phoenix metropolitan area. DDE was not detected in samples from the Upper Santa Cruz Basin or the Sierra Vista subbasin. The large depths to ground water and small amounts of DDT used in most of the West Salt River Valley, the Upper Santa Cruz Basin, and the Sierra Vista subbasin limit the potential for introduction of DDE to the ground water.

Shallow ground water from the nine wells in the agricultural land-use study area had the largest number of VOC detections (35). Ground water from the other 35 wells in the West Salt River Valley had 32 detections. The Upper Santa Cruz Basin (18) and the Sierra Vista subbasin (13) had fewer detections. The larger area of urban land use in the West Salt River Valley appears to be the reason for the greater number of detections there than in the other basins sampled.

Three wells that had five or more VOCs detected in ground water were located in the metropolitan area of Phoenix in the West Salt River Valley. The VOCs detected in these wells were refrigerants, solvents and chemicals used to make solvents, or gasoline additives. These detections are typical of detections found in small capacity wells in the metropolitan Phoenix area (Marsh, 1994). Combinations of solvents and gasoline additives are often detected in ground water because their use is widespread, not necessarily because they are from the same source (Squillace and others, 1999).

Detections of VOCs in ground water in the relatively undeveloped Sierra Vista subbasin indicate that ground water in localized areas of the subbasin may be affected by human activities. These detections are not widespread; therefore, the effects of human activity on present-day ground-water quality are not considered significant for the entire subbasin. These detections are an “early warning” of what could occur in the future in a basin that is presently considered minimally affected by urban activities.

Ground-Water Quality in the West Salt River Valley, Arizona 1996–98—Relations to Hydrogeology, Water Use, and Land Use

By R.J. Edmonds and D.J. Gellenbeck

Water-Resources Investigations Report 01—4126

<http://az.water.usgs.gov/pubs/01-4126intro.html>

DESCRIPTION OF THE STUDY AREA

The West Salt River Valley is a sediment-filled basin of about 1,330 mi² in central Arizona (fig. 1). This area is equivalent to the West Salt River Valley Subbasin of the Phoenix Active Management Area (Arizona Department of Water Resources, 1994). The Salt River Valley comprises the West Salt River Valley and the East Salt River Valley (fig. 1). Most of the city of Phoenix is within the West Salt River Valley. Land use is shifting from agricultural to urban in the entire Salt River Valley as the population increases rapidly in Phoenix and its suburbs.

Physiography and Climate

The West Salt River Valley is an oval structural basin in the Basin and Range Lowlands hydrologic province (Arizona State Land Department, 1963). The valley is drained by the Gila River and its tributaries, which include the Salt and Agua Fria Rivers (fig. 1). The valley is bordered by desert mountain ranges that include the White Tank Mountains on the west; the Hieroglyphic Mountains and the Hedgpeth Hills on the north; the Phoenix Mountains and Camelback Mountain on the east; and the South Mountains, the Sierra Estrella, and the Buckeye Hills on the south. Altitude ranges from about 800 ft above sea level along the Gila River west of Buckeye to about 4,500 ft in the Sierra Estrella.

The climate of the valley is arid and is characterized by hot summers, mild winters, and large diurnal temperature variations. Mean monthly maximum temperature at Buckeye was 68.1°F in January and 109.2°F in July from 1961 through 1990. Mean monthly minimum temperature at Buckeye was 36.4°F in January and 76.0°F in July from 1961 through 1990 (Owenby and Ezell, 1992). Mean annual precipitation was 7.3 in. at Buckeye and 8.5 in. at Phoenix Sky Harbor International Airport for the same period (Owenby and Ezell, 1992). Precipitation varies greatly from year to year, and wet years may have as much as four times the precipitation of dry years (Cordy and others, 1998). The climate is characterized by two rainy periods. Convective storms occur during July and August and are sometimes intense. Frontal storms of lesser intensity occur from December through mid-March (Sellers and Hill, 1974). Evaporation rates are high, and free-water surface evaporation exceeds 5 ft/yr (Farnsworth and others, 1982).

Geology

The West Salt River Valley is one of a series of structural basins along a northwestward-southeastward trend characterized by exposed lower-plate crystalline rocks (Spencer and Reynolds, 1989) and deep basins containing 8,000 to 12,000 ft of basin-fill sediments (Anderson and others, 1992). The basins were formed by high-angle faulting of the Basin and Range disturbance [15 to 5 million years ago (m.y. ago); Menges and Pearthree, 1989] superimposed on the effects of crustal extension and the low-angle detachment faults of the mid-Tertiary disturbance (37 to 15 m.y. ago; Dickinson, 1989; [fig. 2](#), this report). Detachment faulting and later block faulting disrupted pre-existing drainage patterns. Continued subsidence formed closed basins with interior drainage that slowly filled with locally derived sediments and evaporate deposits. When the basins filled with sediment, streams began to flow through the lowest divides into adjacent basins. A slow continuation of this process has resulted in the integrated drainage system of the Gila River and its tributaries (Damon and others, 1984).

The valley is divided into northeastern and southwestern parts by a major linear subsurface structure that is generally along U.S. Route 60 (Brown and Pool, 1989). The northeastern part is dominated by a series of structural blocks tilted to the northeast and trending northwest. Each block is composed of Tertiary volcanic rocks overlying Precambrian crystalline rocks. The thickness of basin fill in the northeastern part generally is less than 2,000 ft. The thickness of basin fill in the southwestern part may exceed 10,000 ft. The crystalline bedrock forms an impermeable boundary at the basin margin and beneath the basin fill. In the study area, the bedrock is composed principally of a variety of granitic and metamorphic rocks of Precambrian to middle Tertiary age (Brown and Pool, 1989; Reynolds, 1985).

Sedimentary and volcanic rocks older than middle Tertiary age are exposed in the Hieroglyphic Mountains, the Hedgpeth Hills, Camelback Mountain, and the Phoenix Mountains (Reynolds, 1988) in the northeastern part of the study area. The principal pre-Basin and Range sedimentary unit is a well-cemented, red conglomerate with clast sizes ranging from clay to boulders that are 15 ft in diameter. This unit may underlie the basin fill in many places in the southeastern part of the basin (Brown and Pool, 1989) and crops out on Camelback Mountain at the eastern margin of the study area (G.E. Cordy, hydrologist, USGS, oral commun., 1999). The unit is not exposed along the mountain fronts in the southwestern part of the basin. In this report, the red unit is considered to be a part of the bedrock of the mountains where it is exposed above the water table and is included with the basin-fill sediments at depth where it is saturated.

The basin fill is composed of alluvial sediments deposited in the structural basin since the mid-Tertiary disturbance particularly during and after the Basin and Range faulting (Brown and Pool, 1989). The basin fill consists of beds of clay, silt, sand, and gravel typical of a continental environment. Beds usually are lenticular and cannot be traced laterally for long distances either in outcrops or in the subsurface. Sediments penetrated in two wells drilled only a few hundred feet apart are quite different in many cases.

Sediments tend to be coarse grained near the mountain fronts and fine grained toward the center of the basin. The basin-fill sediments also tend to be coarse grained at the base of the unit where it overlies the crystalline bedrock. Evaporites including anhydrite, gypsum, and especially halite were deposited near the center of the southwestern part of the basin in the lower part of the basin fill (Brown and Pool, 1989). The Luke salt body (Eaton and others, 1972) lies in the center of the southwestern part.

The uppermost 400 to 500 ft of basin fill was deposited during and after the transition to through flowing drainage and typically is coarser than sediments at greater depths. The upper part of the basin fill contains more sand, gravel, and fine-grained material derived from outside the basin; but it also contains extensive beds of silt and clay across the southern part of the study area. Brown and Pool (1989) reported that the upper part of the basin fill (identified as the "middle unit") in an area west of Goodyear contained more than 80-percent silt and clay. The upper part of the basin fill is coarser grained in the northern part of the study area than in the southern part. Beds of silt and clay are not as widespread in the upper part of the basin fill north of Interstate Highway 10 (I-10; [fig. 1](#)) as they are south of I-10.

Stream alluvium overlies the basin fill along and beneath the present river channels, principally the Agua Fria River, the Salt River, and the Gila River. The stream alluvium was deposited after the filling of the basins with sediment and consists of flood-plain and channel-fill deposits. The stream alluvium ranges from clay and silt in the flood-plain deposits to sand and gravel in the channel-fill deposits. Gravel of cobblesized clasts is prominent particularly along the channel of the Salt River. The thickness of stream alluvium ranges from 0 where the Salt River flows over bedrock as it enters the study area to as much as 400 ft near the confluence of the Salt and Gila Rivers (Brown and Pool, 1989).

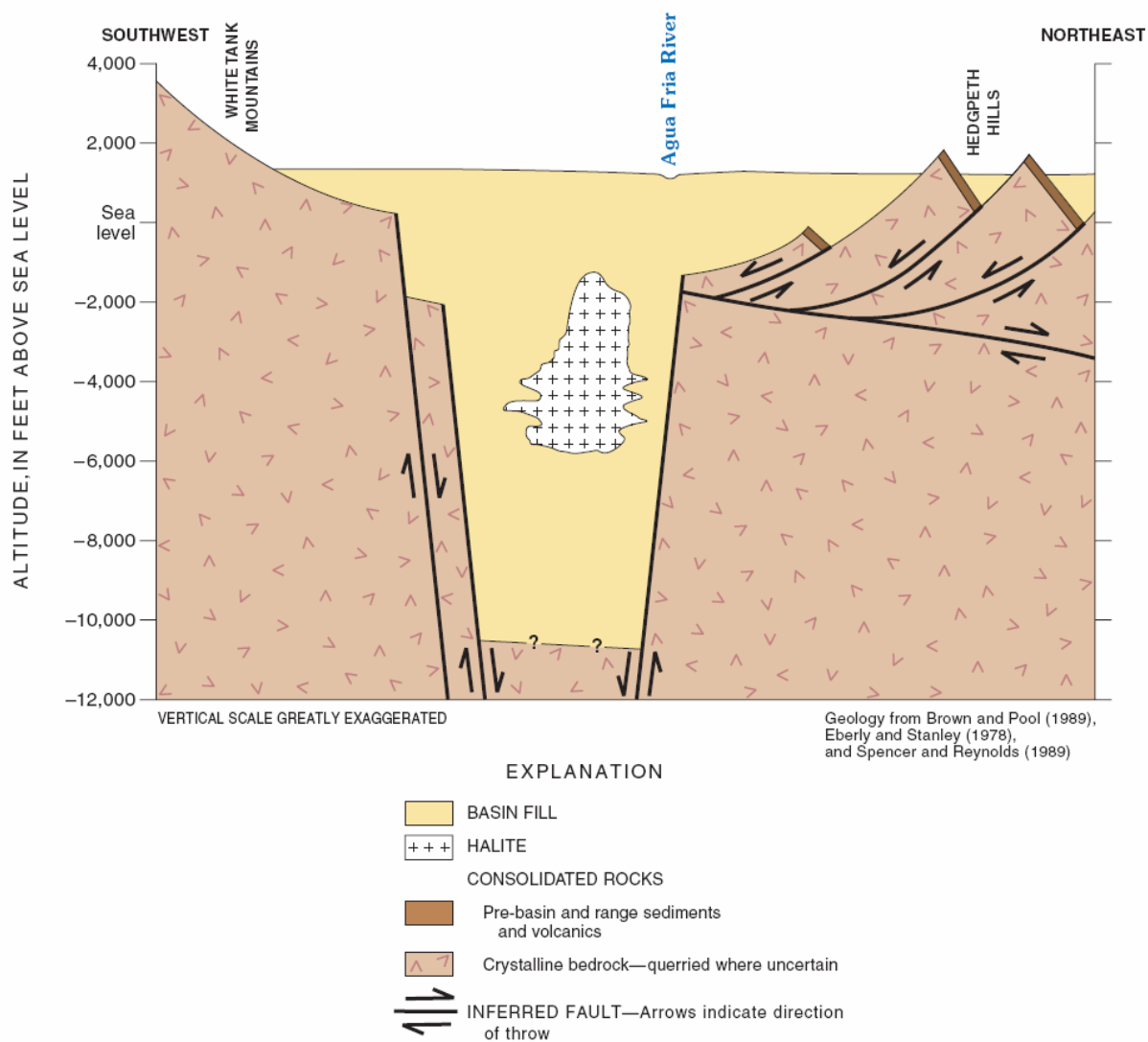


Figure 2. Generalized geologic section, West Salt River Valley, Arizona.

Hydrology

The West Salt River Valley receives an average of less than 10 in./yr of precipitation. Sufficient water to support irrigated agriculture and a growing urban population has been obtained by building reservoirs on the Salt, Verde, and Agua Fria Rivers and by pumping ground water from the alluvial sediments in the basin. The water used does not depend on local precipitation, but comes from snowmelt in the mountains of eastern and central Arizona and from ground water recharged to the aquifer in prehistoric times. Since 1985, additional water has been imported from the Colorado River and either used directly or stored in the basin-fill aquifer. The hydrologic system is managed carefully, and the use of surface water and ground water is regulated strictly.

Surface Water

The streams that drain the mountains bordering the study area are ephemeral. Channels of the mountain streams typically are dry except during periods of precipitation. Flows in the local rivers—the Agua Fria, Salt, and Gila—are almost completely regulated. In years of normal precipitation, all flow is captured and stored behind dams that are upstream from the study area. Downstream from the dams, only occasional flow in local tributaries reaches the riverbeds except at times of high flows during extremely wet years.

Flow in the Salt River is stored behind Roosevelt Dam and three other dams downstream. Flow in the Verde River, the largest tributary of the Salt River, is stored behind Horseshoe and Bartlett Dams. The flow of the Gila River is stored in the San Carlos Reservoir behind Coolidge Dam. Flow in the Agua Fria River is stored in Lake Pleasant behind New Waddell Dam. Water from the reservoirs is distributed in the study area through a system of canals (fig. 3). Originally, the canals were designed to provide only surface water for irrigated agriculture, but as the area developed, ground water augmented the supply.

Since the early 1960s, treated effluent from metropolitan Phoenix has been delivered through some canals in the study area. As the area has undergone a transition from agricultural to urban land use, the canals have become integrated into urban water-supply systems. Water from the Salt River is diverted upstream from Phoenix and transported into the study area through the Arizona, Grand, and Western Canals that are operated by the Salt River Project (SRP). The RID Canal receives treated effluent from the 23rd Avenue wastewater-treatment plant (WWTP) operated by the city of Phoenix and provides irrigation water for farmland west of the Agua Fria River in the southwestern part of the study area. The Buckeye Canal diverts treated effluent from the 91st Avenue WWTP operated by the city of Phoenix and delivers it to farmland between the RID Canal and the Gila River in the southwestern part of the study area. The lands served by the Buckeye Canal have been irrigated for more than 100 years. Lands served by the RID Canal have been irrigated since the 1920s when the canal was constructed. (Montgomery and Associates, 1988). The Beardsley Canal, operated by the Maricopa Water District, transports water from New Waddell Dam to the west-central part of the study area.

The water supply from rivers within the basin is augmented by the Central Arizona Project (CAP) which brings water from the Colorado River to central and southern Arizona. Some of this water is available for use in the study area. The cities of Phoenix and Glendale treat CAP water and distribute it through their municipal systems. CAP water also can be delivered to the study area through SRP canals and the Beardsley Canal. CAP water is stored in Lake Pleasant with water from the Agua Fria River.

Ground Water

The sediments of the basin fill and stream alluvium form the most productive and important aquifer in the study area and are referred to as the basin-fill aquifer in this report. Although the basin-fill aquifer in the study area may be more than 11,000 ft thick, most ground water is pumped from the top 1,000 ft (Anderson and others, 1992). Ground water occurs under unconfined and semiconfined conditions in this aquifer. The semiconfined conditions occur locally where lenticular clay and silt beds form a confining layer. Properly constructed wells with perforations open to the basinfill aquifer can yield as much as several thousand gallons per minute. Where saturated, the upper part of the basin-fill aquifer yields more water than the lower part because the upper part generally is more coarse grained and less cemented than the lower part. Brown and Pool (1989) estimated that the hydraulic conductivity of the upper part ranges from 180 to 1,700 ft/d, and the hydraulic conductivity of the lower part ranges from 3 to 60 ft/d.

Before agricultural development began in the late 1800s, the surface of the water table sloped to the south and southwest, which indicates that ground water moved in that direction (Corkhill and others, 1993). Discharge was primarily to the Gila River at the southwestern edge of the study area and through transpiration by the phreatophytes growing on the flood plains. Ground water also left the valley by underflow to the southwest through the basin-fill aquifer between the Buckeye Hills and the White Tank Mountains (fig. 1). The basin-fill aquifer receives recharge by infiltration of runoff from the mountains along the edge of the basin and by infiltration from the major streams flowing through the basin. Little, if any, recharge results from precipitation falling directly on the valley floor (Anderson and others, 1992).

Development of agricultural and urban lands in the valley provided new sources of recharge and discharge to the basin-fill aquifer. A major new source of groundwater discharge began with the widespread use of highcapacity turbine pumps for irrigation in the 1920s. The application of irrigation water in excess of plant needs provides a major new source of ground-water recharge in some areas that had not been receiving direct recharge previously. Leakage from canals and laterals also provides ground-water recharge on the part of the valley floor that is away from the major streams. Water also is recharged deliberately through the thousands of dry wells installed in urban areas for collecting runoff.

Limited natural, areally distributed recharge and the replacement of recharge from natural streamflow with recharge from a managed system of canals, irrigation, and urban distribution has resulted in a ground-water system significantly different from systems in most parts of the United States. The direction of ground-water flow has changed in response to large-scale ground-water pumping. Before extensive pumping began, ground-water flow was primarily to the southwest. Major cones of depression have developed where ground-water levels have declined

more than 300 ft in Deer Valley northwest of Glendale and between Luke Air Force Base and the White Tank Mountains. By the 1960s, ground-water flow in the center of the West Salt River Valley had been redirected primarily toward these cones of depression (Anderson, 1968).

In response to the continuing decline of groundwater levels, the Groundwater Management Code was passed by the Arizona Legislature in 1980 to eliminate severe ground-water overdraft and to provide a means for allocating Arizona's limited ground-water resources. As a result of the Groundwater Management Code, the Arizona Department of Water Resources (ADWR) and the Phoenix Active Management Area (AMA) were established (Arizona Department of Water Resources, 1998). A principal goal of the Phoenix AMA is to reduce ground-water pumping by 2025 to a quantity that is equal to or less than the quantity being recharged.

Water Use and Land Use

Two major trends in water use and land use have affected ground-water quality in the West Salt River Valley. The first occurred from the 1870s until the 1920s when irrigated agriculture replaced native vegetation. The second is the rapid population increase in metropolitan Phoenix and the accompanying conversion of agricultural and desert land to urban land (Cordy and others, 2001). The quality of ground water may reflect prior land use because of the slow movement of ground water and rapid changes in land use. In addition, the quality of ground water is affected by the type of land use only where enough water is applied to the land to recharge the aquifer.

The construction of dams on the Salt and Verde Rivers hastened the growth of irrigated agriculture by providing a dependable year-round supply of surface water. The principal crops are citrus, cotton, alfalfa, and grains. Irrigation seepage and canal leakage recharged ground water over broad areas away from river channels. The increased use of surface water, however, led to rising ground-water levels and waterlogging by the 1920s (Smith and others, 1982). The Buckeye Water Conservation and Drainage District constructed ditches in the 1920s to help control waterlogging (Errol L. Montgomery and Associates, 1986). Increased pumping of ground water (fig. 4) led to water-level declines in other parts of the study area, particularly in the years after World War II.

Irrigation seepage generally has been recognized as a major factor in the increase of the concentrations of dissolved solids and nitrate in ground water in the West Salt River Valley (Schmidt, 1983). More than half of the water applied during irrigation is used by plants or returned to the atmosphere by evaporation. Almost all the dissolved solids in the applied irrigation water remain in the water unused by the crops thereby increasing the dissolved-solids concentration in the remaining water (Bouwer, 1997). This water seeps into the soil during irrigation and, in many cases, eventually moves down to the water table. The nitrate contained in fertilizers is dissolved in the irrigation water and the portion of the nitrate unused by the plants may be carried down in the irrigation seepage. Long before the introduction of chemical fertilizers (Lee, 1905; Smith and other, 1982), ground-water analyses from the West Salt River Valley indicated that nitrate concentrations exceeded the present-day Maximum Contaminant Level (MCL) of 10 mg/L as nitrogen (U.S. Environmental Protection Agency, 1996). Smith and others, (1982) suggested that, in this case, naturally occurring nitrate had been flushed out of the soil by the first application of irrigation water. In some cases, pesticides applied to the crops also are carried down to the water table with the irrigation seepage. Rapid population growth of metropolitan Phoenix has resulted in continuous conversion of agricultural land to urban land, particularly in the eastern part of the study area (fig. 5, this report; Knowles-Yanez and others, 1999). By 1995, about 252 mi² of the study area remained in agriculture and about 285 mi² was in urban-land use, which is 19.0 percent and 21.5 percent of the study area, respectively. The population in Maricopa County has increased continually since 1945 (fig. 6).

In the study area, the population was estimated to be about 1,435,000 in 1995 (Hitt, 1994; Arizona Department of Economic Security, 2000). As the urbanization of agricultural land progressed, water used for agriculture has been gradually redirected to urban use. A different group of potential contaminants, volatile organic compounds (VOCs), has been introduced to the water supply by urban development. These contaminants can reach ground water when sufficient water is applied to the land surface. The volume of effluent from WWTPs also has increased as cities have grown, and reclaimed effluent has become a source of irrigation water in the study area.

RELATION OF GROUND-WATER QUALITY TO HYDROGEOLOGY, WATER USE, AND LAND USE

Ground-water quality and its variation in the study area can be better understood by examining its relation to hydrogeology, water use, and land use. Although samples from the 35 wells included in the SUS provide a reasonably unbiased, areal representation of ground water in the study area, the data set was selected so that comparisons with data from other NAWQA study areas in the United States could be made. Use of a larger subset of the 73 wells (64 SUS and 9 ALUS) sampled in the study area can result in a better interpretation on the basis of the

land-use, water-use, and well characteristics. All sampled wells having adequate construction and log information were selected for additional interpretation. These 59 wells were grouped into five classes (A, B, C, D, and E) on the basis of land and water use, well construction, and particular geologic conditions existing within parts of the study area (fig. 19).

Classification of Wells

Class A consists of those wells in undeveloped areas that have no surrounding intensive urban development and have never been irrigated for agriculture. Land use is and has been low-density rural residential or rangeland. Irrigation systems or urban water-distribution systems had not been built near these wells at the time of sampling. Wells in class A are near the margins of the valley, particularly in the northwestern part between the White Tank and the Hieroglyphic Mountains (fig. 20).

Class B consists of wells in irrigated agricultural areas north of the RID Canal or east of the Agua Fria River. Additionally, the logs of wells in class B indicate that the wells yield water from an unconfined part of the aquifer. The logs show either that there are no significant, fine-grained beds penetrated by the well, or that the wells were perforated above any finegrained confining beds. Municipal and irrigation wells are included in class B. Some class B wells could be affected by canal seepage, and other class B wells could be affected by flow in the Agua Fria or Salt Rivers. Wells in class C yield ground water from the upper part of the basin-fill aquifer beneath lands that are irrigated with water from the Buckeye or RID Canals.

The extensive silt and clay beds in the middle unit described by Brown and Pool (1989) form a barrier to the downward movement of water and serve as a confining bed in this area. Wells were included in this class if they were perforated above the fine-grained unit according to construction records and drillers' logs. Many of the wells of this group are equipped with turbine pumps and are used for irrigation. Wells 43, 57, and 72 were included in class C, even though perforation information was missing, because well depths were shallower than the clay beds found in nearby wells.

Rising ground-water levels require dewatering in the area of class C wells and have been a problem since the 1920s. Although water levels declined somewhat from the 1930s until the early 1960s, they have risen significantly since the 1960s (Errol L. Montgomery and Associates, 1988). The depth to ground water generally is shallow in this area and ranges from less than 10 ft near the Gila River to about 150 ft near the RID Canal. The area served by the Buckeye and RID Canals is supplied by water imported from outside the service areas. The supply is supplemented by wells within each service area including some of the class C wells. Treated effluent and surface water are delivered through the Buckeye Canal. Ground water from the western part of the SRP service area, treated effluent, and some surface water is delivered through the RID Canal. The presence of ground water recharged by irrigation seepage and the shallow water table provided the opportunity to study the effects of irrigated agriculture on ground water by sampling class C wells.

Wells in class D are wells within the area of agricultural or urban irrigation constructed so that shallow ground water cannot enter the well. Perforations in these wells are completely below a bed of clay with unperforated casing extending from land surface to the clay bed. Sediments of the basin-fill aquifer are lenticular and of variable grain size. The wells in class D are concentrated in the southern part of the study area (fig. 20) where fine-grained sediments, including silt and clay, are most extensive (Brown and Pool, 1989). Wells were included in class D using the following criteria: (1) presence of a clay bed indicated on drillers' logs, (2) all perforations were below the clay bed as indicated on the ADWR construction records, and (3) tritium was not detected in well samples. Many of the wells in this group are domestic wells equipped with submersible pumps.

Class E consists of the nine wells that were constructed for the ALUS and is considered to be a separate group for comparison because the wells were located specifically in an irrigated area where depths to water were shallow, and the wells were constructed to allow samples to be collected at the water table. Samples from class E wells provided a group of analyses from ground water that is known to be affected directly by irrigated agriculture. Except for pesticides, comparisons of ground-water quality of class E wells with other well classes was based on the samples collected in August 1997. Counts of pesticide detections in class E wells were based on samples collected in August 1997 and February 1998.

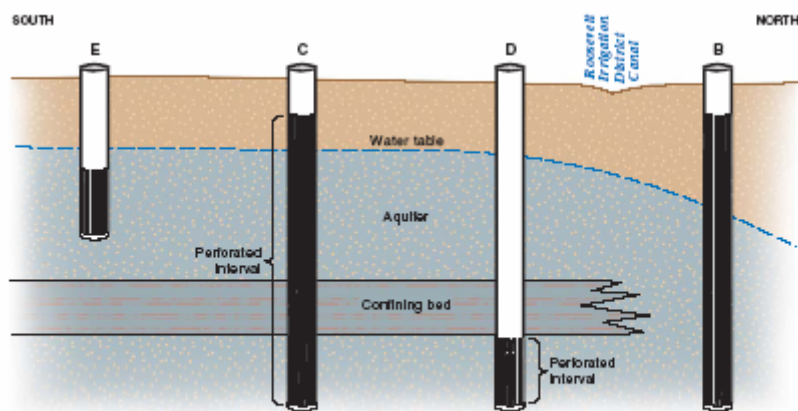


Figure 19. Construction typical of classified wells, West Salt River Valley, Arizona. (Class A not shown.)

SUMMARY AND CONCLUSIONS

Ground-water quality in the West Salt River Valley was assessed by the USGS during 1996–98 as a part of the NAWQA program. Samples were collected to characterize the ground-water quality in the valley in the context of water use, land use, and hydrogeologic factors. Samples were collected from 64 existing wells selected by a stratified-random procedure to assess the chemical characteristics of ground water throughout the valley. Nine monitoring wells were constructed and sampled to observe the effects of agricultural land use on ground-water quality.

Water samples were collected using standard NAWQA sampling techniques and protocols. Field measurements of specific conductance, pH, alkalinity, turbidity, dissolved oxygen, and water temperature were made at the time of sampling. Water samples were analyzed in USGS laboratories. Samples were analyzed for major ions, trace constituents, pesticides, VOCs, stable isotopes of hydrogen and oxygen, tritium, and radon. Replicate and blank samples were collected to ensure that the analyses of the ground-water samples were representative of actual water quality.

The West Salt River Valley was subdivided into 35 cells of equal area. One well from each cell was selected randomly and sampled to provide uniform geographic coverage for statistical analysis of the ground-water quality of the SUS. The MCL of 10 mg/L for nitrate was exceeded in 11 of these 35 wells. Wells that yield water that exceeds the MCL for nitrate are in areas that have past and (or) present agricultural or urban irrigation. Fluoride exceeded the MCL of 4 mg/L in 2 of the 35 wells, and exceeded the SMCL of 2 mg/L in 7 wells. Although the water from 2 sampled wells exceeded the MCL of 50 µg/L for arsenic, 17 wells (49 percent) would exceed the proposed drinking-water standard of 5 µg/L. Pesticides were detected in 23 percent of the SUS wells in both agricultural and nonagricultural settings; however, the concentrations did not exceed MCLs of the USEPA or standards of the State of Arizona.

The most frequently detected pesticide was deethylatrazine, a degradation product of the herbicide atrazine. VOCs were detected in samples from 21 of the 30 wells sampled for analysis (70 percent); however, most detections were below the MRL and none exceeded established standards. Detections of VOCs at such low levels may or may not imply contamination in the ground water; it is difficult to identify specific sources of the VOCs in these samples at the low concentrations. Tritium was not detected in samples from 57 percent of the wells, which indicates that the water from these wells had fallen as precipitation sometime before 1953 and implies that ground-water quality may not be related to current or recent land use.

Samples collected from the nine wells constructed for the ALUS are representative of ground water that has been recharged by recent irrigation seepage. Samples were collected in August 1997 and again in February 1998 to detect seasonal differences in water quality. The dissolved-solids concentration and the concentrations of principal inorganic constituents were higher in the ALUS wells than in the SUS wells. No significant change in the concentration of dissolved solids was noted between the samples collected in August 1997 and the samples collected in February 1998. The median nitrate concentration in the ALUS wells exceeded the MCL. The median nitrate

concentration in the samples collected in February 1998 (19.6 mg/L) was significantly higher than the median nitrate concentration of the samples collected in August 1997 (16.9 mg/L). The increase in nitrate concentration in shallow ground water several months after the irrigation season could be a function of the rate at which irrigation seepage reaches the water table.

Median concentrations of trace constituents in the ALUS wells are not greatly different than in the SUS wells, and none of the concentrations exceeded the MCLs. Water from ALUS wells is enriched in ^{18}O compared with water from the SUS wells as a result of recharge of evaporated irrigation seepage. Pesticides and VOCs were found in shallow ground water in the ALUS area. Pesticides were detected in eight of the nine wells sampled in August 1997 and in all nine wells in February 1998. None of the pesticide concentrations exceeded established standards. Atrazine and deethylatrazine were the most commonly detected pesticides. Low levels of dieldrin and DDE were detected in two and six monitoring wells, respectively. Detections of these two compounds, which have been banned from use, indicates that residual pesticides in the soil can affect ground water.

At least one VOC was detected in each of the samples from the ALUS wells although none of the concentrations exceeded established USEPA or State of Arizona standards. Eight different VOCs were detected a total of 23 times in samples collected in August 1997, and 8 different VOCs were detected 20 times in samples collected in February 1998. Trichloromethane (chloroform) was the VOC detected most frequently in samples collected from the ALUS wells.

The wells sampled by NAWQA were grouped into five classes on the basis of well construction, water use, and screened depth relative to a confining bed to evaluate ground-water quality. Statistically significant differences between classes were found in the quality of the sampled ground water. Median nitrate concentrations in ground water from classes of wells that have been affected by irrigation seepage (classes B, C, and E) exceeded the MCL of 10 mg/L (as nitrogen). Median nitrate concentrations at wells in undeveloped areas (class A) and at wells in which the aquifer is protected by a confining bed (class D) were 1.7 and 2.0 mg/L, respectively. The median dissolved solids concentration for class C and E wells yielding shallow ground water in an area west of the Agua Fria River and south of the RID Canal was much higher than concentrations for the other well classes.

Evaporation of recharge water during irrigation probably has caused an enrichment of D and ^{18}O in ground water of well classes B, C, and E. Pesticides were not detected in wells in undeveloped areas (class A) or those wells completed in aquifers protected by confining beds (class D).

Ground-Water Quality in the Upper Santa Cruz Basin, Arizona, 1998

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Water-Resources Investigations Report 00-4117

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Physical Setting

The Upper Santa Cruz Basin encompasses about 7,430 km² in northern Sonora, Mexico, and in Pinal, Pima, and Santa Cruz Counties, Arizona (**fig. 1**). About 1,150 km² of the basin lies south of the international boundary in Mexico and was not included in this study. The basin consists of a northward-sloping alluvial valley that ranges from 8 to 32 km wide. Altitudes of the valley floor vary from about 750 to 1,200 m above sea level.

The Upper Santa Cruz Basin is bounded on the west by the Pajarito, Atascosa, Tumacacori, Cerro Colorado, Sierrita, Tucson, and Tortolita Mountains; on the east by the Patagonia, San Cayetano, Santa Rita, Rincon, and Santa Catalina Mountains; and on the north by Black Mountain (**fig. 1**). The southern boundary of the study area is the international boundary between Arizona and Mexico.

The Santa Cruz River flows northward to northwestward through the Upper Santa Cruz Basin (**fig. 1**). The river is mostly ephemeral, flowing only in direct response to rainfall or snowmelt (Condes de la Torre, 1970); however, the river is perennial in two reaches downstream from effluent releases from the Nogales International Wastewater-Treatment Plant (WWTP) and the Roger Road and Ina Road WWTPs (**fig. 2**). Major tributaries of the Santa Cruz River within the study area also are ephemeral, except Nogales Wash, which is perennial. Nogales Wash is fed by

springs, rainfall, and uncontrolled sewage discharges from Mexico (International Boundary and Water Commission, 1998).

The Upper Santa Cruz Basin has an arid to semiarid climate. Temperatures above 32°C prevail from May through September (U.S. Department of Commerce, 1998b). In 1998, the average temperature at Tucson was 20.5°C, and the total precipitation was 35.0 cm; the average temperature at Nogales was 16.1°C, and the total precipitation was 45.7 cm (U.S. Department of Commerce, 1998a). Generally, more than 50 percent of the annual precipitation occurs during the summer monsoon season (usually July through September), and over 20 percent occurs in the winter months (December through March; U.S. Department of Commerce, 1998b).

Land Use and Population

The major land use (about 60 percent) in the Upper Santa Cruz Basin is rangeland (figs. 2 and 3; Anderson and others, 1976). Other land uses are urban, 22.5 percent; forest, 14.3 percent; agricultural, 2.2 percent; and transitional (mines, quarries, bare rock, gravel pits, and sandy areas), 0.7 percent. A small percentage of land use is wetlands (Anderson and others, 1976). Agricultural land in the Upper Santa Cruz Basin historically covered a greater area than it does today. Irrigated acreage in Pima County, which includes a large part of the Upper Santa Cruz Basin, reached a plateau in 1955 and remained fairly constant until 1975; the cropped acreage was about 202 to 243 km² (Water Resources Research Center, 1999). The cropped acreage declined after 1975 as agricultural land was retired or developed for urban use, particularly in the Tucson and Marana areas and along the Santa Cruz River (Water Resources Research Center, 1999). In 1997, irrigated acreage in Pima County had declined to about 93 km² (Water Resources Research Center, 1999).

Several historical and present land-use activities in the Upper Santa Cruz Basin (fig. 2) have locally affected the ground-water quality (in terms of inorganic constituents). Prior to 1994, chromium concentrations in ground water near the Raytheon Systems Company (previously the Hughes Aircraft Company) and the Tucson International Airport exceeded the Maximum Contaminant Level (MCL; 100 µg/L) of the U.S. Environmental Protection Agency (USEPA; Graham and Monical, 1997). Cleanup procedures for chromium were completed in this area in 1994 when measured chromium concentrations were below the USEPA MCL. Ground water near the open-pit copper, silver, and molybdenum mines south of Tucson (fig. 4) has concentrations of dissolved solids and sulfate that exceeded USEPA Secondary Maximum Contaminant Levels (SMCLs; 500 and 250 mg/L, respectively; Pima Association of Governments, 1983). Ground water downgradient from treated-effluent releases to the Santa Cruz River from the Roger Road and Ina Road WWTPs (fig. 5) has concentrations of nitrate that exceeded the USEPA MCL (10 mg/L). The historical agricultural areas surrounding the Roger Road and Ina Road WWTPs were irrigated for decades with treated effluent from the WWTPs (Martin, 1980). Ground water downgradient from uncontrolled sewage discharges to Nogales Wash from Mexico (fig. 6) has concentrations of nitrate that exceeded the USEPA MCL (10 mg/L; International Boundary and Water Commission, 1998). This site was listed on Arizona's Water-Quality Assurance Revolving Fund Priority List in 1987 for nitrate and volatile organic compounds (Arizona Department of Environmental Quality, 1996).

About 86 percent of the Upper Santa Cruz Basin contains less than 50 people per square kilometer (Hitt, 1994). Tucson (fig. 7), the largest city in the basin, had a population of 468,500 in 1998 (Arizona Department of Economic Security, 1998). By the year 2020, the population of Tucson is expected to be about 589,900 (Arizona Department of Economic Security, 1997). The population of other cities in the basin in 1998 were: 25,500 in Oro Valley, 21,200 in Nogales, 10,000 in Marana, 5,700 in South Tucson, and 2,900 in Sahuarita (Arizona Department of Economic Security, 1998). By the year 2020, populations of other cities in the basin are expected to rise: 59,400 in Oro Valley, 27,800 in Nogales, 76,600 in Marana, 7,200 in South Tucson, and 10,600 in Sahuarita (Arizona Department of Economic Security, 1997).

Geohydrologic Setting

The Upper Santa Cruz Basin is a northward- to northwestward-trending alluvial basin bounded by block-faulted mountains (fig. 8). The basin's alluvial deposits, which are derived from the surrounding mountains, are referred to collectively as the basin fill. The basin fill of the Upper Santa Cruz Basin differs in thickness, lateral extent, and composition north and south of an inferred fault that is believed to connect the Sopori Wash and Elephant Head-Pantano Wash Faults (Halpenny and Halpenny, 1988). The basin fill north of the inferred fault may be as much as 6,000 m thick, and the basin fill south of the inferred fault is less than 1,000 m thick (Oppenheimer and Sumner, 1980).

The difference in the geology between the two areas probably is the result of downfaulting of the basin fill north of the inferred fault at some time after the basin-fill deposition. Subsequent additional downward movement of the northern basin fill resulted from continued compaction of basin-fill sediments (Halpenny and Halpenny, 1988). For purposes of this report, the geology of the basin fill north of the inferred fault will be discussed separately from the geology of the basin fill south of the inferred fault. The basin fill north and south of the inferred fault is hydrologically connected to form a single basin-fill aquifer (Davidson, 1973); therefore, the hydrology of the basin will be discussed as one unit.

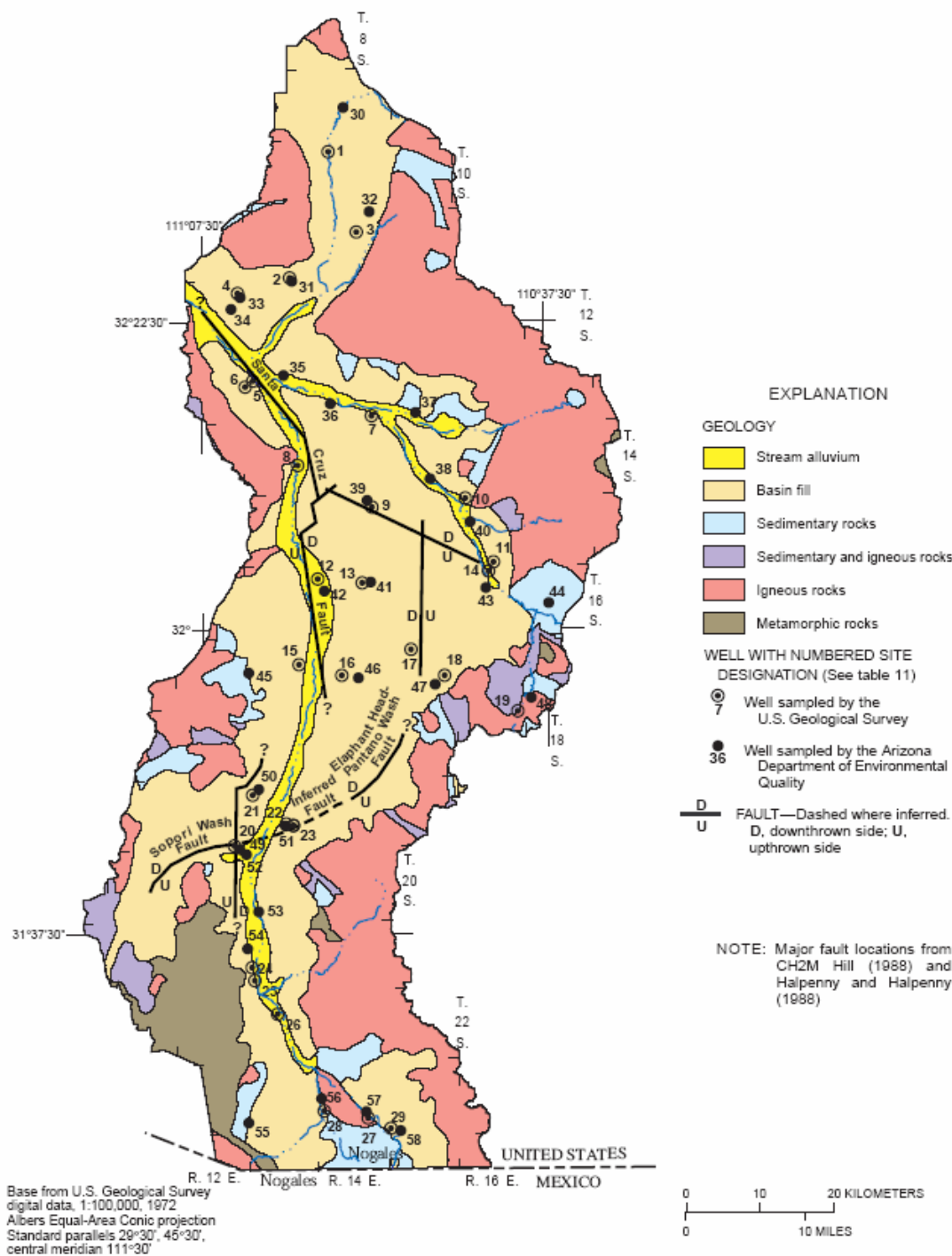


Figure 8. Generalized surficial geology, major faults, and locations of sampled wells, Upper Santa Cruz Basin, Arizona. Geology digital data modified from Reynolds (1988).

Geology

The mountains surrounding the Santa Cruz Basin consist of igneous, metamorphic, and sedimentary rocks of Precambrian to Tertiary age (fig. 8). The mountains are largely composed of granite, andesite, rhyolite, basalt, monzonite, granodiorite, gneiss, limestone, quartzite, conglomerate, sandstone, and shale (Davidson, 1973; Anderson, 1988; Halpenny and Halpenny, 1988). The igneous, metamorphic, and sedimentary rocks generally are impermeable; however, secondary fractures can store water locally.

North of the Inferred Fault.—The Pantano Formation is a consolidated to semiconsolidated conglomerate of Tertiary age and overlies the basal bedrock in the basin north of the inferred fault (Davidson, 1973). The formation ranges from hundreds to thousands of meters in depth, consists of silty sandstone to gravel, and is strongly tilted and offset by faulting (Davidson, 1973). 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 outcrops also contain interbedded volcanic flows and tuffs (Anderson, 1987).

The Tinaja beds overlie the Pantano Formation. The Tinaja beds are of Tertiary age, range from hundreds to thousands of meters in depth, are coarse grained along the margins of the basin, and grade into finer-grained sediments and evaporite deposits in the center of the basin (Anderson, 1988). Davidson (1973) interpreted the Tinaja beds as a sedimentary detrital filling of a subsiding basin. The Tinaja beds comprise three unconformable units—the lower, middle, and upper beds. The lower Tinaja bed consists of gravel and conglomerate to clayey silt and mudstone and ranges in thickness from tens to hundreds of meters thick.

The middle Tinaja bed lies mainly in the center of the basin, consists of gravel and conglomerate to gypsiferous and anhydritic clayey silt and mudstone, and ranges in thickness from tens to hundreds of meters thick. The upper Tinaja bed consists of gravel, sand, and clayey silt and is tens of meters thick (Anderson, 1988). The Tinaja beds crop out along the southern slopes of the Santa Catalina Mountains, the western slopes of the Rincon Mountains, and the eastern slopes of the Tucson and Sierrita Mountains (Anderson, 1987).

Loosely packed to weakly cemented sediments of Quaternary age, known collectively as the Fort Lowell Formation, unconformably overlie the Tinaja beds. The Fort Lowell Formation grades from silty gravel near the edges of the basin to silty sand and clayey silt in the center of the basin and ranges in thickness from 100 m in the center of the basin to a few meters near the edges of the basin (Davidson, 1973). The formation crops out extensively in the foothills of the Santa Catalina and Rincon Mountains (Davidson, 1973). For the purposes of this study, the Fort Lowell Formation is shown as basin fill over most of the valley floor north of the inferred fault on the surficial geology map (fig. 8); however, throughout most of this area, it is commonly overlain by a veneer of alluvium (Anderson, 1987).

Stream alluvium unconformably overlies the Fort Lowell Formation along the Santa Cruz River and its tributaries north of the inferred fault. The stream alluvium is of Quaternary age, consists mainly of gravel and gravelly sand, and ranges in depth from a few meters to tens of meters thick (Davidson, 1973).

South of the Inferred Fault.—The Nogales Formation is a consolidated conglomerate of Tertiary age and overlies the basal bedrock in the basin south of the inferred fault (Halpenny, 1963). The Nogales Formation consists of sandstone, claystone, and conglomerate derived from limestone, granite, and volcanic material and is at least 450 m thick (Halpenny, 1963). The lower Tinaja bed north of the inferred fault is thought to be correlative with the Nogales Formation (Anderson, 1987). The formation crops out between the Santa Cruz River and Nogales Wash and on the southern slopes of the San Cayetano Mountains (Simons, 1974).

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). For the purposes of this study, the older alluvium is shown as basin fill over most of the valley floor south of the inferred fault on the surficial geology map (fig. 8).

South of the inferred fault, stream alluvium of Quaternary age has been deposited along the Santa Cruz River. The alluvium is composed of gravel, sand, and occasional lenses of silt and ranges in thickness from about 25 to 35 m (Halpenny and Halpenny, 1988).

Hydrology

North of the inferred fault, the stream alluvium is not part of the basin-fill aquifer because the water table is below the base of these deposits; however, the stream alluvium north of the inferred fault may become saturated for as long as several weeks after sustained streamflow (CH2M Hill, 1988). South of the inferred fault, the water table is above the base of the stream alluvium; in this area, the stream alluvium is the most productive part of the basin-fill aquifer (Halpenny, 1963).

The Fort Lowell Formation is the most productive part of the aquifer north of the inferred fault—hydraulic-conductivity values range from about 6 m/d to as much as 29 m/d (Davidson, 1973). In some areas, however, pumping has lowered the water table below the base of the formation (CH2M Hill, 1988).

The Tinaja beds form the thickest part of the aquifer north of the inferred fault, and hydraulic-conductivity values range from about 0.4 to 16 m/d (Davidson, 1973). Hydraulic-conductivity values of the Pantano Formation range from about 0.2 to 4 m/d (Davidson, 1973). North of the inferred fault, computed transmissivity values for the basin-fill aquifer range from about 12 to almost 6,200 m²/d and generally are less than 620 m²/d (Anderson, 1972). South of the inferred fault, computed transmissivity values for the older alluvium range from 120 to 270 m²/d (Halpenny and Halpenny, 1988). The Nogales Formation does not yield a substantial quantity of ground water to wells (Halpenny, 1963).

The basin-fill aquifer of the Upper Santa Cruz Basin is unconfined or partly confined (Davidson, 1973; Halpenny and Halpenny, 1988). North of the inferred fault, water levels range from about 25 to 70 m below land surface along the Santa Cruz River and are as much as 160 m below land surface near Vail (Tucson Water, 1998). South of the inferred fault, water levels generally are less than 10 m below land surface along the Santa Cruz River and may exceed 100 m below land surface near the base of mountain ranges (Murphy and Hedley, 1984). A zone of perched water near the southeast slope of the Tucson Mountains has been reported (Tucson Water, 1998).

Recharge to the basin-fill aquifer of the Upper Santa Cruz Basin primarily occurs from infiltration of winter precipitation and surface flow in the Santa Cruz River and its tributaries (Halpenny, 1963; CH2M Hill, 1988). Other sources of recharge to the basin-fill aquifer include infiltration of precipitation along the mountain fronts, infiltration of effluent in the Santa Cruz River from the Roger Road and Ina Road WWTPs and the Nogales International WWTP, infiltration of effluent in the Santa Cruz River and in percolation ponds from the Green Valley WWTP, infiltration of reclaimed effluent from the Sweetwater Effluent Recharge Basins, infiltration of Central Arizona Project (CAP) water from the Pima Mine Road Recharge Project, infiltration of sewage return flows in the Nogales Wash, infiltration of agricultural return flows from irrigated fields adjacent to the Santa Cruz River, and infiltration of mine return flows as seepage from tailing ponds (fig. 2; Pima Association of Governments, 1983, 1985; CH2M Hill, 1988; Hanson and Benedict, 1994). Discharge of effluent to the Santa Cruz River from the Roger Road and Ina Road WWTPs began in 1950, and discharge of effluent from the Nogales International WWTP began in 1972 (Schmidt and Associates, 1988; Hanson and Benedict, 1994). Discharge of effluent to the Santa Cruz River from the Green Valley WWTP began in 1964 and ended in 1981 when recharge of effluent began from a percolation pond (Pima Association of Governments, 1985). Recharge of effluent from the Sweetwater Effluent Recharge Basins began in 1987, and recharge of CAP water from the Pima Mine Road Recharge Project began in 1998 (Mitch Basefsky, Public Information Supervisor, City of Tucson, oral commun., 1999). Agricultural return flows have been decreasing since 1965, and mine return flows have been decreasing since the 1980s (Hanson and Benedict, 1994).

Ground-water movement generally is from the mountain-front areas toward the valley floors, and then northward in the southern part of the Upper Santa Cruz Basin and northwestward in the northern part of the basin (Murphy and Hedley, 1984). Ground-water discharge in the Upper Santa Cruz Basin occurs as underflow beneath the northwest boundary of the basin, as evapotranspiration, and as pumping from wells (CH2M Hill, 1988).

The hydrologic system in the Upper Santa Cruz Basin was considered to be in approximate equilibrium prior to 1940 (Anderson, 1972), which means that recharge from natural sources equaled discharge from the basin. Coates and Halpenny (1954) reported that in the early to mid-1800s, the ground-water table was shallow enough for the Santa Cruz River to be a perennial stream through much of its course and for phreatophytes to be more plentiful along the river banks than they were in 1954. From the 1930s until about 1975, ground-water pumping steadily increased in the central and northern parts of the basin, and ground-water levels began to decline substantially in these areas (CH2M Hill, 1988). Extensive pumping from the basin-fill aquifer has resulted in ground-water level declines of more than 40 m in the central part of the basin near Green Valley and more than 60 m in the northern part of the basin in the Tucson area (Tucson Water, 1998). Ground-water level declines in the basin have had several

effects—shifting of natural groundwater flow paths toward pumping centers; increased vertical-hydraulic gradients; reduction in perennial streamflow in the Santa Cruz River and its tributaries as water levels are lowered below the river bottom; reduction in evapotranspiration as water levels are lowered below plant roots; development of perched zones in the aquifer as water levels are lowered below nonpermeable layers; and compaction of the aquifer, resulting in land subsidence near Tucson of as much as 0.15 m from 1951 to 1980 and as much as 0.06 m from 1980 to 1996 (Anderson, 1988; Hanson and Benedict, 1994; Evans and Pool, 2000).

South of the inferred fault, the aquifer has not been affected by ground-water level declines because less ground water is pumped from the basin-fill aquifer in this area than in the northern part of the basin, and because the narrow, shallow river valley in the southern part of the basin is conducive to rapid recharge to the basin-fill aquifer (Halpenny and Halpenny, 1988).

Natural Controls on Ground-Water Quality

Natural controls on ground-water quality in the Upper Santa Cruz Basin must be understood before the effects of human activities on ground-water quality in the basin can be determined. Natural controls on ground-water quality were examined by identifying variations in ground-water quality related to well depth and geology (basin-fill units, distance from faults, distance from stream alluvium, and location north or south of the inferred fault; [fig. 8](#); [table 11](#) in the section entitled “Basic Data” at the back of the report). Well depth was determined using well-drillers’ logs. Locations of fault zones, basin-fill units, and stream alluvium were determined using geologic maps and well-drillers’ logs.

Depth

The sampled wells ranged in depth from 12 to 259 m. Only one well was perforated exclusively below 213 m; therefore, the basin’s “deep ground water” (at depths greater than 213 m) described by Laney (1972) generally was not sampled. This study focused on the chemical quality of Laney’s “shallow ground water” (at depths less than 213 m).

Temperature and pH were found to significantly correlate with well depth throughout the entire basin ([fig. 11](#)). The temperature generally increased 4°C per 100 m of well depth, and the pH generally increased 0.15 standard units per 100 m of well depth. The increase in ground-water temperature with depth in the aquifer was described by Laney (1972).

Geology

This study focused on ground-water quality in the basin-fill aquifer. The majority of the sampled wells were perforated in the Fort Lowell Formation, the upper Tinaja bed, or the older alluvium. Six of the sampled wells were perforated in consolidated rock near the edges of the basin, and there were no perforation data for 10 of the sampled wells. Groundwater quality was not significantly different among the three basin-fill units. A significant difference was not identified with the Kruskal-Wallis test statistic.

Previous investigations did not compare the groundwater quality of the different basin-fill units. Previous investigations identified high concentrations of bicarbonate, sulfate, calcium, nitrate, and fluoride in ground water along the Santa Cruz River (Laney, 1972). For this study, ground-water quality was examined in relation to distance of the sampled wells from the stream alluvium. The quality of ground water less than 2 km from the stream alluvium was not significantly different than the quality of ground water more than 2 km from the stream alluvium. A significant difference was not identified with the Wilcoxon rank-sum test statistic.

Ground-water quality north of the inferred fault was compared to ground-water quality south of the inferred fault to identify water-quality variations related to the structural and compositional differences of the basin fill north and south of the fault. The ground-water quality north of the inferred fault was not significantly different than the ground-water quality south of the inferred fault. A significant difference was not identified with the Wilcoxon rank-sum test statistic. Previous investigations did not compare ground-water quality north and south of the inferred fault.

Dissolved solids, alkalinity, calcium, potassium, chloride, and sulfate concentrations were higher in samples collected from wells less than 2 km from major faults in the basin fill than in samples collected from wells more than 2 km from major faults. A significant difference was identified using the Wilcoxon rank-sum test statistic ([fig. 12](#)). High concentrations of dissolved solids, alkalinity, calcium, chloride, and sulfate in ground water near the Santa Cruz Fault was identified by Laney (1972). Laney attributed these concentrations to upward migration of ground water through faults from gypsiferous mudstones of the Tinaja beds.

The dissolved-solids concentrations in groundwater samples significantly correlate with calcium and sulfate concentrations ([fig. 13](#)). This correlation suggests a gypsiferous source of the ground water. Dissolved-solids concentrations above the USEPA SMCL (500 mg/L) at sites 5, 6, 12, 14, 16, 18, 19, 20, 28, 42, 45, 49, 52, and 56, and the sulfate concentrations above the USEPA SMCL (250 mg/L) at sites 6 and 14 can be partially attributed to the upward migration of ground water. Ground-water samples from many of these wells also had high (greater than 100 mg/L) calcium concentrations.

In addition to upward migration of ground water along faults, the high concentrations of dissolved solids and sulfate at site 14 also may be attributed to a zone in the basin fill first described by Laney (1972). Laney described a zone containing high concentrations of dissolved solids, calcium, and sulfate that extended from near Vail northwestward toward central Tucson. The high concentrations in this area were attributed to movement of ground water through gypsiferous rocks of the Pantano Formation and (or) older rocks in the headwaters of Pantano Wash.

The high dissolved-solids concentrations at some sites can be attributed to geology; however, at some sites, it also may be attributed to human activities near the wells. The Upper Santa Cruz Basin is east of an area of massive evaporite deposits in the basin fill and high specific-conductance values in the ground water that is called the “Gila Low” (Pierce, 1974). Specific conductance values (which are related to dissolved-solids concentrations) measured in 1998 in the Upper Santa Cruz Basin are higher than would be expected for a basin outside the “Gila Low” (Gellenbeck and Coes, 1999). In addition, dissolved-solids concentrations in the Upper Santa Cruz Basin generally are higher than those of the eastward neighboring basin of Sierra Vista, which has less urban and agricultural development (Coes and others, 1999).

One ground-water sample had a fluoride concentration greater than the USEPA MCL (4 mg/L; site 34; [figs. 2 and 8](#)), and one ground-water sample had a fluoride concentration greater than the USEPA SMCL (2 mg/L; site 8; [figs. 2 and 8](#)). Laney (1972) attributed fluoride concentrations in ground water in the Upper Santa Cruz Basin to chemical reactions between the aquifer material and the ground water. Site 8 is at the base of the Tucson Mountains, which are primarily volcanic in origin (Davidson, 1973). Fluoride-bearing minerals are common in these volcanic rocks, and clays in the basin-fill deposits downgradient from the Tucson Mountains probably have fluoride weakly attached to cation-exchange sites.

The ground-water sample collected at site 8 had a pH above the USEPA SMCL range (6.5 to 8.5 standard units). The high pH may contribute to conditions that allow fluoride ions to be replaced by hydroxyl ions in clays, consequently, fluoride ions are released into solution. The ground-water sample from site 34 did not have a high pH, and samples from two wells near site 34 (sites 4 and 33) had fluoride concentrations of less than 1 mg/L; thus, the source of the high fluoride concentration at site 34 is not clear. Analyses from the NWQL and the ADHS laboratory were biased toward high concentrations of fluoride; therefore, the high concentrations in these two samples may be a factor of laboratory bias.

The ground-water sample from site 8 had an arsenic concentration almost double the current USEPA MCL (50 µg/L; U.S. Environmental Protection Agency, 1996). The high concentration at site 8 may be attributed to arsenic compounds in the basin fill; however, it also may be attributed to human activities near the well. Robertson (1991) attributed high arsenic concentrations in ground water in southern Arizona to oxidized arsenic compounds in the basin-fill sediments, which are originally derived from sulfide and arsenide deposits in the surrounding mountains. The volcanic rocks in the Tucson Mountains may be a source of arsenic to the basin-fill sediments near site 8.

Effects of Human Activities on Ground-Water Quality

Effects of human activities on ground-water quality in the Upper Santa Cruz Basin were examined by identifying variations in present ground-water quality among areas of different land use and by changes in ground-water quality over time.

Land Use

For ground-water quality to be affected by land use, the ground water must have received recharge from the land surface after that land use was established. To determine if wells received recharge from specific areas of existing or previous land uses, well depths, water levels, well locations, and tritium concentrations were analyzed ([tables 10 and 11](#) in the section entitled “Basic Data” at the back of the report). Tritium, a radioactive isotope, can be used to estimate time of recharge and distinguish younger ground water from older ground water. Large quantities of tritium were released to the atmosphere during thermonuclear-weapons testing from 1952 until the late 1960s. Atmospheric

tritium is incorporated into water molecules that form precipitation and recharge ground water. The amount of tritium in ground water at a given time is controlled by the amount of tritium in the atmosphere when recharge occurs and by the radioactive-decay rate of tritium. Ground water that does not contain detectable tritium can be assumed to have been recharged before 1953, and ground water that does contain detectable tritium can be assumed to have some component of ground water that was recharged after 1953 (fig. 14). For this study, a well in which ground water had less than 2.5 pCi/L (the MRL) of tritium was considered to yield water that had been recharged before 1953 (old recharge), and a well at which ground water had more than 2.5 pCi/L of tritium was considered to yield some component of ground water that was recharged after 1953 (recent recharge).

The land-use type for the area surrounding the wells that contained recent recharge was then identified from [figure 2](#); the wells that contained recent recharge were surrounded by areas in which the land use was urban, agricultural, or rangeland. Data from wells that contained old recharge were compared to data from wells that contained recent recharge from urban areas and to wells that contained recent recharge from agricultural areas. Agriculture, however, was practiced in the basin before 1953, and some of the wells defined as containing “old recharge” may contain some component of recharge from areas of historical agricultural activities. As previously described, wells that contained recent recharge from rangeland areas were not included in the analysis because of the small sample size. The sample from well 25 had less than 2.5 pCi/L of tritium; however, this well is considered to be in an area that had received recent recharge because this well is only 24.4 m deep, has a water level of 9.6 m, and is adjacent to the Santa Cruz River, which is a major recharge source.

Urban Activities.—Concentrations of nitrite plus nitrate (as nitrogen) were higher in ground water that contained recent recharge from urban areas than in ground water that contained old recharge ([fig. 15](#)). A significant difference was identified with the Kruskal-Wallis and the Tukey test statistics. Major nitrogen sources in the urban areas include fertilizers applied to lawns and effluent discharges from WWTPs. Additionally, many present-day urban areas were historically agricultural areas, some of which were irrigated with wastewater. Of the five ground-water samples that exceeded the USEPA MCL and State of Arizona aquifer water-quality standards for nitrite plus nitrate, two samples were from wells in urban areas (sites 45 and 28; [fig. 2](#)).

The ground-water sample from site 45, west of Green Valley at the base of the Sierrita Mountains, exceeded the USEPA SMCL for dissolved solids (500 mg/L) and the USEPA MCL and State of Arizona aquifer water-quality protection standard for nitrite plus nitrate (10 mg/L). Because this well did not have a detectable level of tritium, however, the ground water near the well probably had not received recent recharge from the land surface. High dissolved-solids and nitrate concentrations in this area have been previously identified by the Pima Association of Governments (1983). This well is perforated in consolidated rock, and flow through fractures may be contributing some recharge. In addition, this well is near open-pit mines; therefore, the source of the high dissolved-solids concentration may be leachate from nearby mining activity (Pima Association of Governments, 1983). The amount of information available is insufficient to determine a source for the high nitrite plus nitrate concentration.

The ground-water sample from site 28, which is north of Nogales next to Nogales Wash, exceeded the USEPA MCL and State of Arizona aquifer water quality standards for nitrite plus nitrate (10 mg/L) and the USEPA SMCLs for manganese (50 µg/L) and dissolved solids (500 mg/L). The ground-water sample from site 56, which also is next to Nogales Wash, exceeded the USEPA SMCL for dissolved solids (500 mg/L) and had a nitrite plus nitrate concentration higher than the 75th percentile value for the nitrite plus nitrate data (5 mg/L; [table 4](#)). High nitrogen concentrations in the Nogales Wash area have been identified by the ADEQ (Arizona Department of Environmental Quality, 1996). High manganese and dissolved-solids concentrations, however, have not been previously identified in this area. The sources of the high concentrations of nitrite plus nitrate, dissolved-solids, and manganese probably are uncontrolled releases of untreated wastewater to the Nogales Wash south of the international boundary.

Agricultural Activities.—Nitrite plus nitrate, calcium, and potassium concentrations were higher in ground water that contained recent recharge from present agricultural areas than in ground water that contained old recharge ([fig. 15](#)). Calcium and potassium concentrations also were higher in ground water that contained recent recharge from present agricultural areas than in ground water that contained recent recharge from urban areas. Significant differences were identified with the Kruskal-Wallis and the Tukey test statistics. The major sources of nitrogen and potassium in agricultural areas are fertilizers applied to irrigated fields. The major source of calcium in agricultural areas is the dissolution of calcite concentrated by evaporation during irrigation of agricultural areas. This irrigation water then recharges the ground water. Of the five ground-water samples that exceeded the USEPA MCL and State of Arizona aquifer water-quality standards for nitrite plus nitrate, three samples were from wells in present agricultural areas (sites 12, 16, and 42; [fig. 2](#)). Laney (1972) suggested that high nitrate concentrations along the

Santa Cruz River also may result from the decomposition of organic matter in former marshes. Alkalinity and dissolved-solids concentrations were higher in ground water that contained recent recharge from present agricultural areas than in ground water that contained recent recharge from urban areas (fig. 15).

Significant differences in concentrations were identified with the Kruskal-Wallis and the Tukey test statistics. The major sources of alkalinity and dissolved solids in agricultural areas are the dissolution of salts and calcite concentrated by evaporation during irrigation of agricultural areas. This irrigation water then recharges the ground water.

The arsenic concentration at site 8 that exceeded the current USEPA MCL (50 µg/L; fig. 2) could be related to human activities rather than natural factors. The tritium value for site 8 indicates that the well contained some component of ground water that was recharged from 1954 to 78 (table 10 in the section entitled “Basic Data” at the back of the report; fig. 14). The ground-water sample also contained several anthropogenic compounds including pesticides (Tadayon and others, 1999). Presently (1998), site 8 is in an area of urban land use, but historically, agriculture was predominant in this area (Water Resources Research Center, 1999), and contamination of the local aquifer from historical agricultural drainage is possible.

Various arsenic compounds were used as pesticides in this area (Pima Association of Governments, 1989); however, analysis of arsenical pesticides was outside the scope of this study. Although the use of most arsenical pesticides has been discontinued in the United States (Reigart and Roberts, 1999), the arsenic compounds may still be present in the ground water near areas in which arsenical pesticides were used. Arsenic concentrations in wells less than 1 km from site 8 are about an order of magnitude lower than the arsenic concentration at site 8 (U.S. Geological Survey National Water-Information System; Bryn Enright, hydrologist, City of Tucson, oral commun., 1999; Nancy Peterson, Environmental Project Coordinator, City of Tucson, oral commun., 1999). The high arsenic concentration in the sample from site 8 probably is indicative of a localized condition.

SUMMARY AND CONCLUSIONS

Ground-water quality of the Upper Santa Cruz Basin was assessed in 1998 by the USGS and the ADEQ. The study included analyzing ground-water quality data collected by the two agencies from 58 wells and analyzing pre-existing ground-water quality data for 6 wells to: (1) characterize present (1998) conditions; (2) analyze natural controls on ground-water quality; and (3) analyze the effects of human activities on ground-water quality. Individual and combined quality-control data collected by the USGS and the ADEQ confirmed the validity of combining the ground-water quality data collected by the two agencies.

Twenty-nine percent of the ground-water samples collected had concentrations of at least one constituent that exceeded a Federal or State water-quality standard. The USEPA MCLs and State of Arizona aquifer water quality standards were exceeded in 1 sample for arsenic, 1 sample for fluoride, and 5 samples for nitrite plus nitrate; the USEPA SMCLs were exceeded in 1 sample for fluoride, 1 sample for iron, 1 sample for manganese, 2 samples for pH, 2 samples for sulfate, and 14 samples for dissolved solids. In addition, at least 10 samples had arsenic concentrations that exceed a proposed lower arsenic MCL; additional samples may also have concentrations that exceed the proposed MCL—the ADEQ MRL for arsenic was higher than the proposed MCL.

Ground-water quality in the Upper Santa Cruz Basin is affected by natural factors and human activities. The natural factors that have the most effect on ground-water quality in the basin are depth in the aquifer and distance from major faults. Ground-water quality was not significantly different among the various basin-fill units; between parts of the basin fill that differ in thickness, lateral extent, and composition north and south of an inferred fault; or among areas that differ in distance from stream alluvium.

Groundwater temperatures and pH significantly increased in relation to the depth of the well from which the sample was collected. Concentrations of dissolved solids, alkalinity, calcium, potassium, chloride, and sulfate were significantly higher in samples collected from wells less than 2 km from major faults than in samples collected from wells more than 2 km from major faults. Laney (1972) attributed this difference to upward migration through faults of ground water from gypsiferous mudstones. Fluoride concentrations above the USEPA MCL are attributed to fluoride-bearing volcanic rocks in the Tucson Mountains, which probably are the source of local basin-fill deposits; pH dependent ion exchange on clay minerals contributes to the high concentrations of fluoride in this area.

Human activities were found to have an effect on ground-water quality in the Upper Santa Cruz Basin. Ground water that contained recent recharge from urban areas had significantly higher concentrations of nitrite plus nitrate than ground water that did not contain recent recharge from the land surface. Major sources of nitrogen in

urban areas are fertilizers and discharges from WWTPs. A sample from one well in an area that contained recent recharge from an urban area had concentrations of nitrite plus nitrate above the USEPA MCL. Water quality at the well probably is affected by uncontrolled wastewater releases to the Nogales Wash south of the international boundary.

Ground water that contained recent recharge from present agricultural areas had significantly higher concentrations of nitrite plus nitrate, calcium, and potassium than ground water that did not contain recent recharge from the land surface. The major source of nitrogen and potassium in agricultural areas is fertilizers, and the major source of calcium in agricultural areas is the dissolution of calcite concentrated by evaporation during irrigation of agricultural areas. This irrigation water then recharges the ground water. Samples from three wells in areas that contained recent recharge from agricultural areas had concentrations of nitrite plus nitrate above the USEPA MCL. Ground water that contained recent recharge from agricultural areas also had higher concentrations of calcium, potassium, alkalinity, and dissolved solids than ground water that contained recent recharge from urban areas. The major source of these constituents in agricultural areas is the dissolution of salts and calcite concentrated by evaporation during irrigation of agricultural areas. This irrigation water then recharges the ground water. One well had an arsenic concentration above the current USEPA MCL, and water quality at this well probably is affected by pesticides that were in historical agricultural return flows.

Human activities also had an effect on groundwater quality from the 1980s to 1998 in the Upper Santa Cruz Basin. Concentrations of nitrite plus nitrate and dissolved solids significantly increased at a well in an area of agricultural land use; historical and present agricultural areas probably have contributed to the ground-water quality at this site. Concentrations of nitrite plus nitrate significantly increased at a well where the land use has changed from rangeland to urban; an increase in the use of fertilizers on lawns may be affecting the ground-water quality at this site.

Concentrations of nitrite plus nitrate and dissolved solids significantly decreased at a well in an urban area. The decrease in concentrations may be indicative of a decrease in recharge from agricultural areas as historical agricultural land was retired for urban development. Constituents did not significantly increase or decrease at another well in an agricultural area, at another well where the land use has changed from rangeland to urban, and at a well where the land use has changed from agricultural to urban. All of the constituent concentrations in samples used for the historical analyses were below USEPA MCLs and SMCLs.

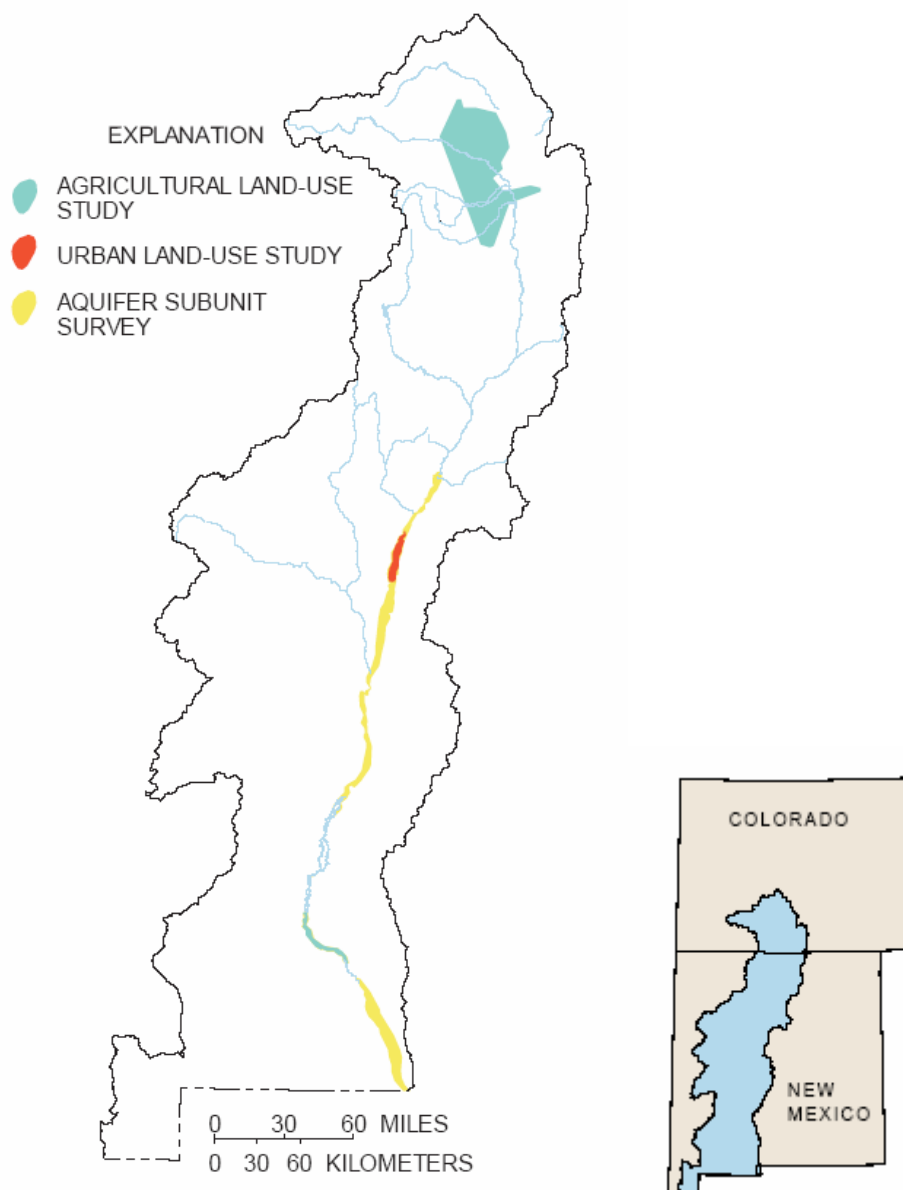
RIOG

Water Quality in the Rio Grande Valley, Colorado, New Mexico, and Texas, 1992-95

By Gary W. Levings, Denis F. Healy, Steven F. Richey, and Lisa F. Carter

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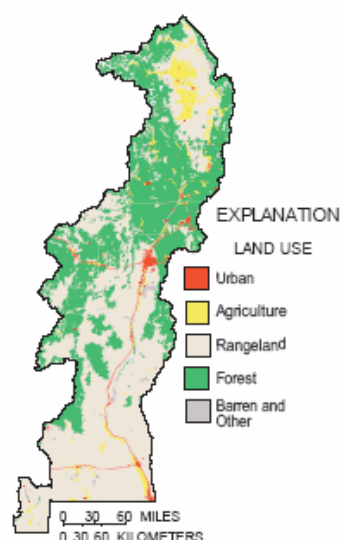


RIOG Ground-water Chemistry

Land-use studies	Examine natural and human factors that affect the quality of shallow ground water that underlies areas of urban land, overhead, center-pivot sprinkler-irrigated agriculture land, or flood-irrigated agricultural land.	Install shallow wells and sample water from wells for analysis of dissolved solids, major constituents, nutrients, dissolved organic carbon, trace elements, pesticides, volatile organic compounds, and radionuclides.		
		(1) San Luis Valley agricultural land-use study of an area irrigated with ground water by overhead, center-pivot sprinklers.	35	1993
		(2) Albuquerque urban land-use study.	24	1993
		(3) Rincon Valley agricultural land-use study of an area flood irrigated with surface water.	30	1994
Aquifer subunit survey	Describe the overall water quality in the part of the basin-fill aquifer used for domestic supply, public supply, or irrigation.	Randomly select and sample existing wells located in the Rio Grande flood plain from Cochiti Lake, New Mexico, to El Paso, Texas, for analysis of major constituents, nutrients, dissolved organic carbon, trace elements, pesticides, volatile organic compounds, and radionuclides.	30	1 1995



The Rio Grande Valley Study Unit is located in three physiographic provinces. Extreme contrasts in precipitation, runoff, and temperature characteristics exist between the Southern Rocky Mountains and the Basin and Range Provinces. These characteristics strongly affect land and water use in the Study Unit.



Rangeland is dominant in the Basin and Range Province, and forest is dominant in the Southern Rocky Mountains and Colorado Plateaus Provinces. Agricultural land use is limited primarily to areas where surface water or shallow ground water is available for irrigation.

Issue: Ground water is the main source of drinking water in the Rio Grande Valley Study Unit; its quality is a major concern. A variety of chemicals used in human activities, including pesticides, volatile organic compounds (VOCs), and nitrate, were detected in ground-water samples from shallow wells (within the top 10-15 feet of the water table). Samples from deeper ground water underlying the Rio Grande flood plain, which is more typically used as a drinking-water source, contained one pesticide, no VOCs, and nitrates (pages 10, 13, and 14).

- Nitrate concentrations exceeded the U.S. Environmental Protection Agency (EPA) drinking-water standard in 31 percent of shallow wells in the San Luis Valley and 17 percent of shallow wells in the Rincon Valley. Both of these areas are predominantly agricultural land use.
- Pesticides were detected in both agricultural and urban land-use areas, with 29 percent of shallow wells containing at least one pesticide. Prometon and metolachlor were the most frequently detected, yet neither compound exceeded EPA drinking-water standards, although standards do not exist for all pesticides detected. Only one pesticide was detected in the deeper ground water.
- Six VOCs were detected in shallow ground water from 11 percent of the wells sampled, most commonly in an urban land-use area. No VOCs were found in deeper ground water. No concentrations exceeded EPA drinking-water standards; however, standards exist for only three of the six compounds detected.
- Radon, a naturally occurring radionuclide in the Rio Grande Valley, was detected in all ground-water samples in concentrations that ranged from 190 to 2,300 picocuries per liter. The highest median concentrations occurred in shallow wells in the San Luis Valley and in the deeper ground water. About 57 percent of the samples exceeded 300 picocuries per liter, the proposed EPA drinking-water standard that has recently been withdrawn for further evaluation.

The primary focus was along the Rio Grande flood plain. Limited work also was conducted on tributary and nontributary streams in the northern part of the Study Unit. The only exception was an agricultural land-use study component assessing shallow ground water in basin-fill deposits in the San Luis Valley located primarily outside the flood plain.

The Study Unit, approximately 45,900 square miles, contains diverse hydrologic and climatological regimes. Altitudes range from peaks exceeding 14,000 feet in the northern part of the Study Unit to approximately 3,700 feet at the lower basin boundary. The climate in the high mountain headwater areas of the Rio Grande and its northern tributaries is alpine tundra where average annual precipitation may exceed 50 inches; most of this precipitation is in the form of snow. Annual runoff in mountainous areas may exceed 30 inches. In contrast, 750 miles downstream near the lower basin boundary of the Study Unit, the Rio Grande flows through a portion of the Chihuahuan Desert where average annual precipitation is less than 8 inches; most of this precipitation is in the form of summer thunderstorms. Annual runoff is less than 0.01 inch. In the mountainous headwater areas, annual potential evaporation is less than 70 percent of annual precipitation, whereas in the southern part of the Study Unit, annual potential evaporation may exceed 1,000 percent of annual precipitation.

The existence of a reliable source of surface water in the southern part of the Study Unit is the result of human alteration of the river system by the construction of reservoirs to control runoff. The storage of runoff in the springtime allows for a dependable source of water to support irrigation needs throughout the long growing season. This alteration of the river system has affected the water quality of the Rio Grande.

Within the Study Unit, two groundwater land-use studies and one aquifer subunit survey focused on the flood plain of the Rio Grande from Cochiti Lake in New Mexico, to El Paso, Texas. One land-use study was conducted in the basin-fill deposits in the San Luis Valley in Colorado. Only when the hydrologic system along the Rio Grande is understood can the complexity of the many influences that affect water quality throughout the Study Unit be appreciated. A detailed discussion of the ground-water system in the Study Unit is presented in Ellis and others (1993). The following is a brief description of the ground-water system to aid in understanding the following water-quality discussions.

Complex interactions occur between ground water and surface water in the Rio Grande flood plain. A system of canals distributes surface water for agricultural irrigation and a system of drains intercepts shallow ground water and returns it to the Rio Grande. Surface water leaks from the Rio Grande and canals to recharge the shallow ground-water system. In places, deeper ground water flows upward to recharge the shallow ground-water system and/or to contribute flow to the Rio Grande. In addition, excess applied irrigation water infiltrates and recharges the shallow ground-water system. Evapotranspiration losses from vegetation, land, and water surfaces, can have a major effect on the quality of ground water.

Two land-use studies, in the Albuquerque area and in the Rincon Valley, focused on the upper 10-15 feet of ground water in the flood-plain alluvium (referred to as shallow ground water in this report). This water generally is not used for domestic supply. However, because of the interaction with surface water and, in isolated areas, because declines in the water level in the deeper aquifer have caused the shallow aquifer to be a recharge source in some areas, knowing the quality of the shallow ground water is important in making management decisions.

The aquifer subunit survey, from Cochiti Lake to El Paso, focused on deeper water underlying the floodplain (referred to as deeper ground water in this report). This water is used for domestic supply in some areas. The land-use study in the San Luis Valley was not entirely in the Rio Grande flood plain. Most of the area is in the San Luis Closed Basin, which is both a ground-water and surface-water closed basin. Neither ground water nor surface water flows out of the Closed Basin. The shallow ground water is in unconsolidated basin-fill deposits. The water in these basin-fill deposits is used extensively for irrigation and, on a limited basis, for domestic supply. However, these wells generally tap several tens of feet of the aquifer. The study wells were completed in the upper 10-15 feet of the saturated basin-fill deposits.

Historically, streamflow in the Rio Grande was caused by spring snowmelt (April through June) and summer monsoon thunderstorms (July and August). This natural streamflow pattern has been altered and regulated by the construction of reservoirs on the main stem and tributaries that impound and store water for later use, primarily irrigation. All surface water in the Rio Grande is appropriated by various compacts, treaties, and individual water rights. Appropriated surface-water rights on the Rio Grande in Colorado and New Mexico usually exceed the annual mean flow of the river (Ellis and others, 1993).

As the discussion of the water-quality data is presented, it is important to realize that 89 percent of water use in the Study Unit in 1990 was for irrigation, 8 percent for public supply, and 3 percent for all other uses. The effect that irrigation and urban use has on the quality of surface water is related to the processes to which the water is subjected as it moves downstream. It is repeatedly diverted, applied to fields, returned to the Rio Grande directly or through drains, impounded in reservoirs, lost to evapotranspiration, or consumptively used. Urban areas contribute wastewater and anthropogenic organic chemicals such as volatile organic compounds and pesticides; agricultural areas contribute chemicals from fertilizers and pesticides; mining areas contribute trace elements; atmospheric deposition contributes chemicals such as nitrate and phosphorus; and the use and reuse of water increases dissolved-solids concentrations because of evapotranspiration. All of these processes can contribute to deterioration in surface-water quality of the Rio Grande as it moves through the basin.

Pesticides in Ground Water

During 1993-95, 65 wells that tap shallow ground water were sampled in two agricultural land-use studies (table 2). Eighteen wells had detectable pesticides, with 6 different pesticides detected a total of 23 times. Of the 24 wells that represent urban land use overlying shallow ground water, 8 wells had detectable pesticides, with 5 different pesticides detected a total of nine times. The aquifer subunit survey, which sampled water from existing wells that are completed in the deeper aquifer, yielded only 1 pesticide detection for 30 wells. The pesticides detected most often were prometon (12) and metolachlor (10) (table 2). No concentrations of pesticides exceeded any EPA MCL or HA.

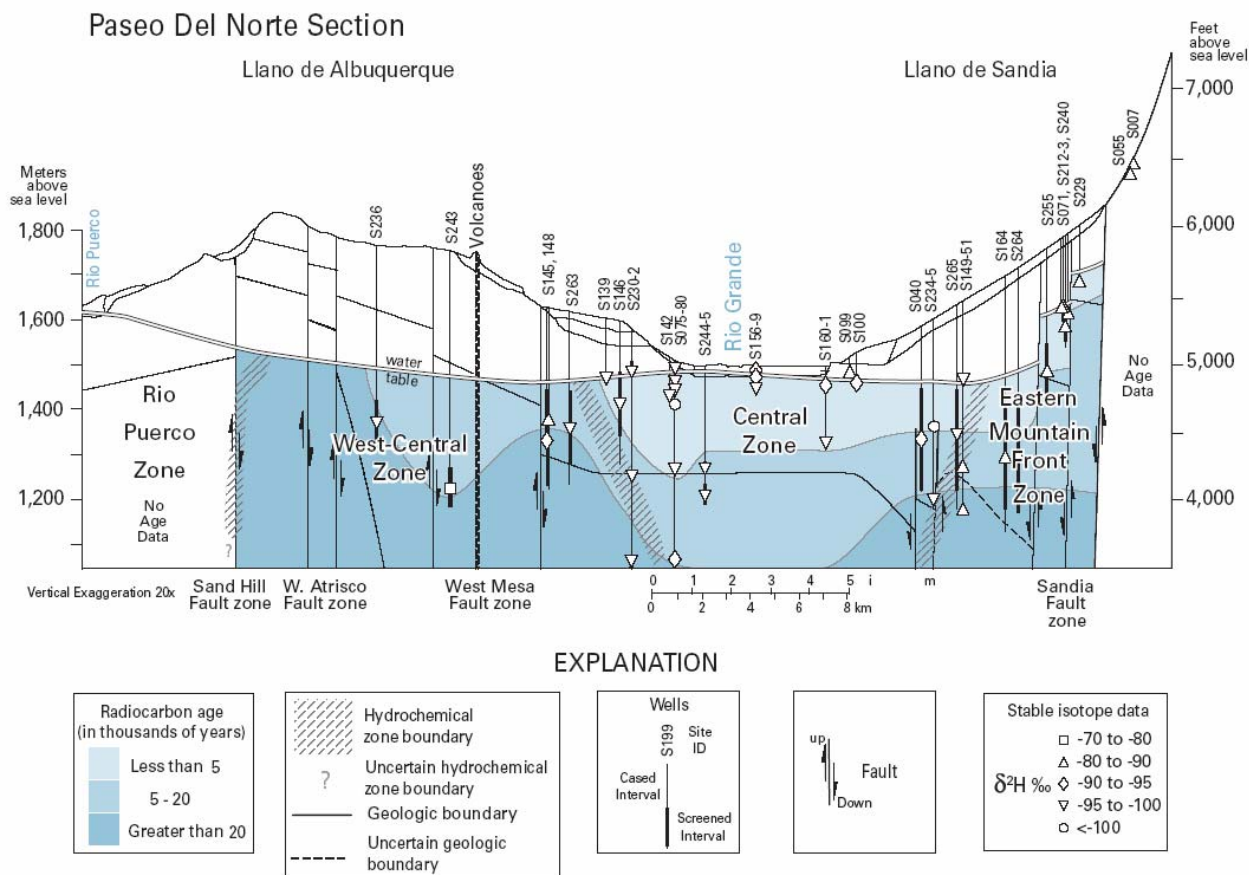
The small number of different pesticides detected and the small concentrations of those pesticides indicate little leaching of pesticides from land surface to shallow ground water in these land-use areas. The detection of only one pesticide in the samples from the aquifer subunit survey suggests that shallow ground water containing pesticides has not moved into deeper parts of the aquifer.

Nutrients in Ground Water

In the San Luis Valley agricultural land-use study, water from 11 of the 35 wells sampled contained nitrate concentrations greater than the EPA MCL; the largest concentration was 58 mg/L (table 6). In the Rincon Valley agricultural land-use study, water from 5 of the 30 wells sampled exceeded the EPA MCL; the largest concentration was 33 mg/L. These elevated nitrate concentrations are indicative of leaching of fertilizers into shallow ground water. However, the spatial variation in nitrate concentrations indicates that leaching of fertilizer is not uniform throughout these areas. The spatial variation may be the result of variable fertilizer application rates, timing of fertilizer application, timing of application of irrigation water, and recharge rates controlled by soil type, texture, permeability, precipitation, and biological/geochemical processes in the unsaturated zone.

In the Albuquerque urban land-use study, the largest nitrate concentration was 2.8 mg/L, which is considerably below the EPA MCL. Although infiltration of septic tank effluent results in considerable loading of nutrients to the shallow aquifer in this area, the small nitrate concentrations suggest that nitrogen compounds in the effluent are not being converted to nitrate. This is probably due to the lack of dissolved oxygen and the relatively large dissolved organic carbon concentrations in shallow ground water in the area.

Large ammonia concentrations, which would be expected, are not present in shallow ground water because ammonia is adsorbed on clays in the aquifer or soil zone. Nitrate concentrations in water from deep wells sampled during the aquifer subunit survey ranged from less than 0.05 to 1.9 mg/L. Throughout most of the area of the aquifer subunit survey, the small nitrate concentrations in water from deeper parts of the basin-fill aquifers probably indicate that shallow ground-water recharge into the deeper parts of the aquifer is limited.



Abstract from Water-Resources Investigations Report 97-4067.

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

By Scott K. Anderholm

ABSTRACT

This report describes the quality of shallow ground water and the relations between land use and the quality of that shallow ground water in an urban area in and adjacent to Albuquerque, New Mexico. Water samples were collected from 24 shallow wells. Samples were analyzed for selected common constituents, nutrients, trace elements, radionuclides, volatile organic compounds, and pesticides.

The study area, which is in the Albuquerque Basin in central New Mexico, was limited to the Rio Grande flood plain; depth to water in this area generally is less than 25 feet. The amount and composition of recharge to the shallow ground-water system are important factors that affect shallow ground-water composition in this area. Important sources of recharge that affect shallow ground-water quality in the area include infiltration of surface water, which is used in agricultural land-use areas to irrigate crops, and infiltration of septic-system effluent in residential areas.

Agricultural land use represents about 28 percent of the area, and residential land use represents about 35 percent of the total study area. In most of the study area, agricultural land use is interspersed with residential land use and neither is the dominant land use in the area. Land use in the study area historically has been changing from agricultural to urban.

The composition of shallow ground water in the study area varies considerably. The dissolved solids concentration in shallow ground water in the study area ranges from 272 to 1,650 milligrams per liter, although the relative percentages of selected cations and anions do not vary substantially.

Calcium generally is the dominant cation and bicarbonate generally is the dominant anion. Concentrations of nutrients generally were less than 1 milligram per liter. The concentration of many trace elements in shallow ground water was below or slightly above 1 microgram per liter and there was little variation in the concentrations. Barium, iron, manganese, molybdenum, and uranium were the only trace elements analyzed for that had median concentrations greater than 5 micrograms per liter. Volatile organic compounds were detected in 5 of 24 samples. Cis-1,2-dichloroethene and 1,1-dichloroethane were the most commonly detected volatile organic compounds (detected in two samples each). Pesticides were detected in 8 of 24 samples. Prometon was the most commonly detected pesticide (detected in 5 of 24 samples). Concentrations of volatile organic compounds and pesticides detected were much smaller than any U.S. Environmental Protection Agency standards that have been established.

Infiltration of surface water and the evaporation or transpiration of this water, which partially is the result of past and present agricultural land use, seem to affect the concentrations of common constituents in shallow ground water in the study area. The small excess chloride in shallow ground water relative to surface water that has been affected by evaporation or transpiration could be due to mixing of shallow ground water with small amounts of precipitation/bulk deposition or septic-system effluent.

Infiltration of septic-system effluent (residential land use) has affected the shallow ground-water composition in parts of the study area on the basis of the small dissolved oxygen concentrations, large dissolved organic carbon concentrations, and excess chloride. Despite the loading of nitrogen to the shallow ground-water system as the result of infiltration of septic-system effluent, the small nitrogen concentrations in shallow ground water probably are due to the small dissolved oxygen concentrations and relatively large dissolved organic carbon concentrations.

The small concentrations and lack of variation of most trace elements indicate that land use has not substantially affected the concentrations of most trace elements in shallow ground water. The relatively large dissolved iron and manganese concentrations in shallow ground water might be the result of the reduced state of shallow ground water, which is due to residential land use. There is no direct evidence that land use has affected radionuclide concentrations or activities.

The presence of synthetic organic compounds (volatile organic compounds and pesticides) in shallow ground water in the study area indicates that human activities have affected shallow ground-water quality. Determining the type of land use associated with the presence of a particular synthetic organic compound in shallow ground water was not possible.

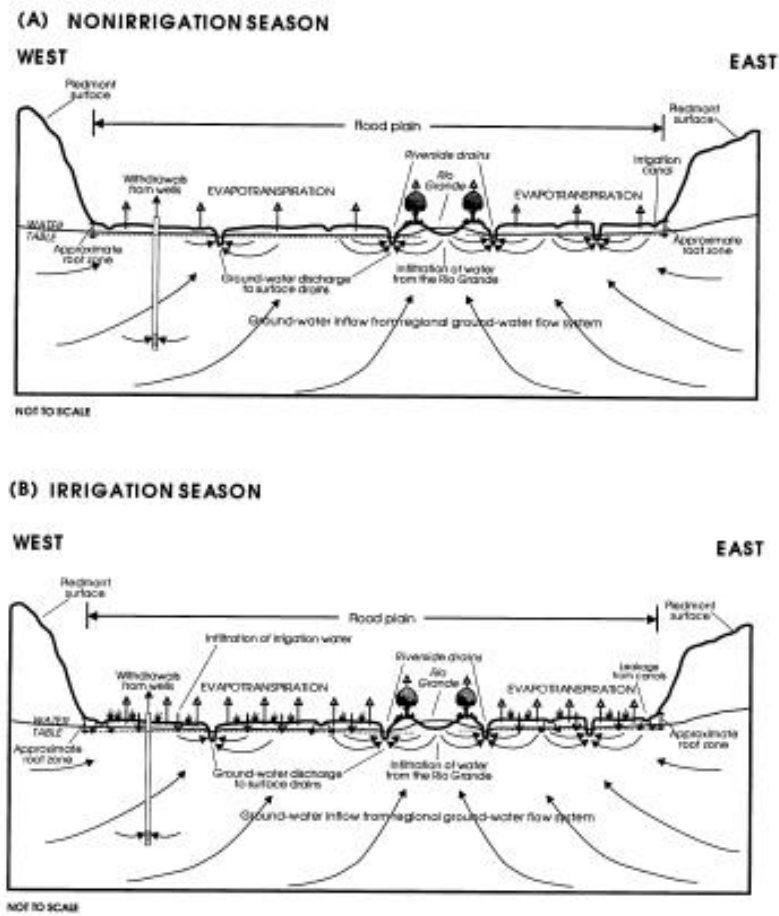


Figure 4. -- Generalized hydrologic sections showing local and regional ground-water flow along the Rio Grande flood plain during the (A) nonirrigation season and (B) irrigation season (modified from Anderholm, 1987).

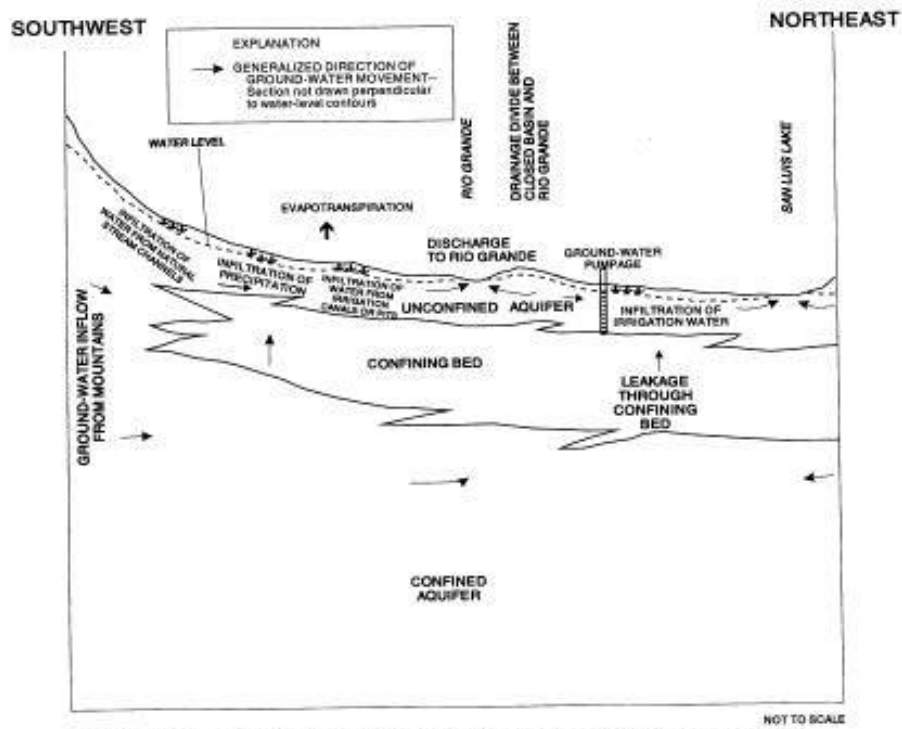


Figure 2.--Generalized geohydrologic section illustrating the hydrologic system of the San Luis Valley.

Geochemical characterization of ground-water flow in the Santa Fe Group aquifer system, Middle Rio Grande Basin, New Mexico

By L. Niel Plummer, Laura M. Bexfield, Scott K. Anderholm, Ward E. Sanford, and Eurybiades Busenberg

U.S. GEOLOGICAL SURVEY

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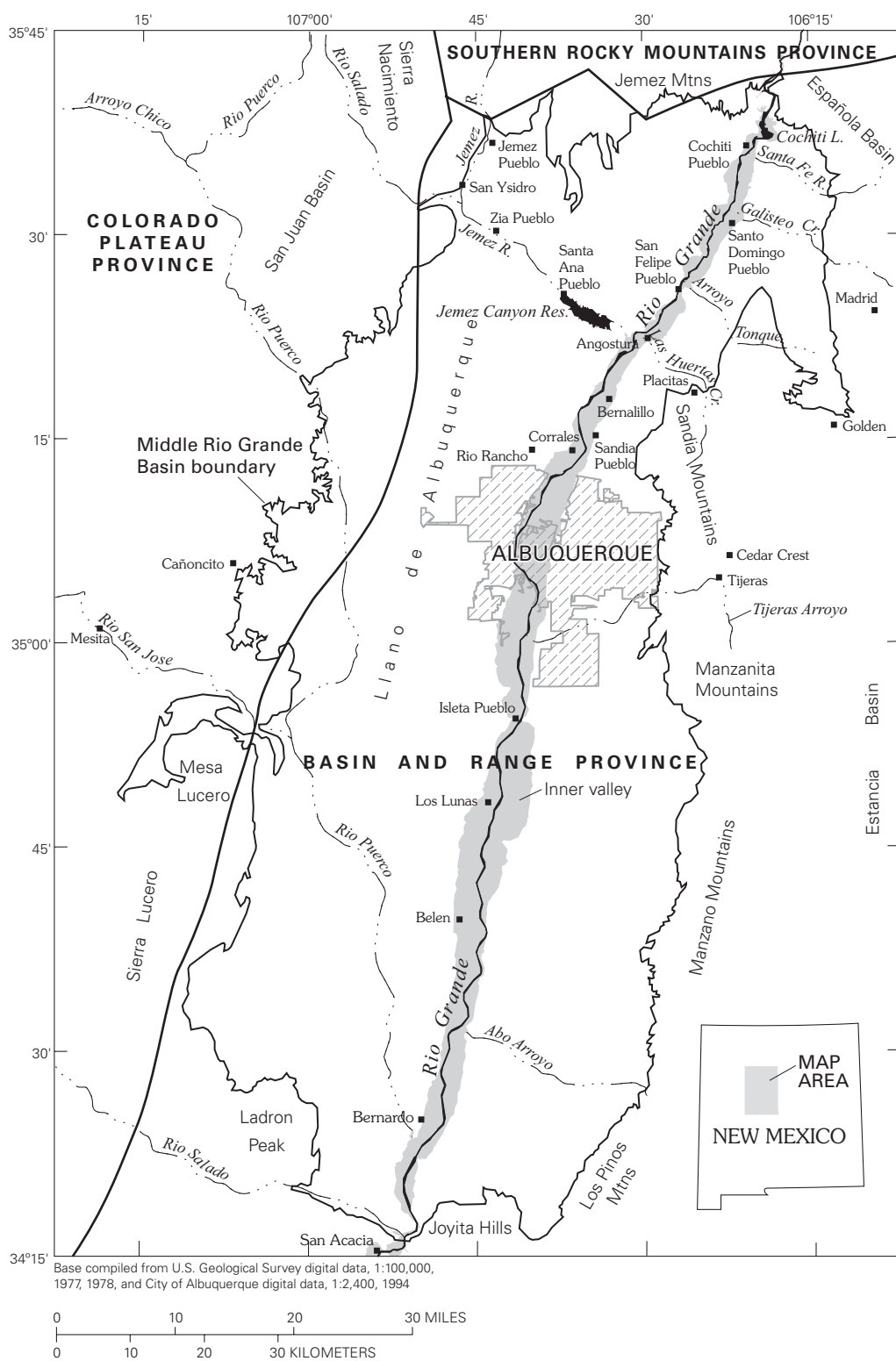


Figure 1. Selected physiographic features of the Middle Rio Grande Basin and vicinity, New Mexico.

Ground-Water-Flow System

The ground-water-flow system of the MRGB between Cochiti and San Acacia is complex and in some areas has not been well characterized because of a lack of data. Multiple sources of recharge to the aquifer system are present across the basin. Land use, particularly the presence of irrigation and septic systems, has added to the potential sources of recharge. Characterization of the flow system also has been complicated by drawdown because of sustained ground-water pumping, particularly in the vicinity of Albuquerque, which has altered directions of ground-water flow and probably changed the rates of recharge resulting from various sources. Faults that juxtapose relatively permeable deposits with impermeable units also appear to affect directions and rates of ground-water flow. These faults also have been proposed as

possible conduits for the upward flow of relatively deep (from depths of thousands of feet) ground water (Bexfield and Anderholm, 2002a).

A map of predevelopment water levels compiled by Bexfield and Anderholm (2000) indicates that ground-water movement through the central part of the basin has historically been oriented primarily north to south (fig. 9). Near the basin margins, ground-water flow has historically been oriented primarily toward the central part of the basin. Maps published by Bjorklund and Maxwell (1961) and Titus (1961) of pre-1960 water levels show similar patterns, although they indicate a greater east-to-west component of ground-water flow east of the Rio Grande in the vicinity of Albuquerque.

On various predevelopment water-level maps of the region (Bexfield and Anderholm, 2000; Bjorklund and Maxwell, 1961; Meeks, 1949; Titus, 1961; Titus, 1963), a depression in the water-level surface, commonly referred to as the “ground-water trough”, is apparent west of the Rio Grande, from just south of the Jemez River south to the area of Los Lunas (fig. 9). This trough is located on the east side of the Sand Hill and Cat Mesa faults. Previous investigators have theorized that the presence of the trough indicates that there is a thicker sequence of more permeable material in the area of the trough than in areas on either side (Kernodle and others, 1995). However, lithologic logs of wells drilled in the trough area generally do not appear to support this hypothesis (Hawley, 1996; Stone and others, 1998). Further discussion of the origin of the ground-water trough is given in the section “Summary of Implications from Geochemical and Isotopic Data for the Conceptual Model of the Aquifer System”.

There has been a steady increase in ground-water pumping in the MRGB since about the mid-1940’s (Thorn and others, 1993; Bartolino and Cole, 2002). Estimates of urban, rural, commercial, and industrial ground-water withdrawals in the MRGB were 97,000, 131,000, and 152,700 acre-feet for the years of 1970, 1980, and 1990 (Thorn and others, 1993, and references within). Estimates of total ground-water withdrawals for the 7 counties that extend over at least part of the MRGB totaled nearly 310,000 acre-feet in 1995 (Wilson and Lucero, 1997; Bartolino and Cole, 2002). Over the period 1974-92, 72 percent of the total basin ground-water withdrawals were made by the City of Albuquerque (Thorn and others, 1993). Pumping in the vicinity of Albuquerque has resulted in substantial

declines in water levels, as indicated by water-level data for 1999-2002 (Bexfield and Anderholm, 2002b) (fig. 10). The largest and most widespread declines compared to predevelopment levels are east of the Rio Grande, where water levels have fallen in excess of 120 feet; declines of as much as 100 feet have been observed west of the Rio Grande (Bexfield and Anderholm, 2002b). Water-level declines have resulted in ground-water movement being directed into the major pumping centers on the east and west sides of the Rio Grande (fig. 10). These large-scale shifts in the directions of ground-water flow during the past 40 to 50 years locally may affect interpretation of some geochemical data, although ground-water flow velocities are low enough that regionally significant changes in water-quality patterns are not likely to have occurred during this time period. Smaller-scale changes in ground-water flow directions also likely have occurred as a result of ground-water pumping in the vicinity of other communities, such as Bernalillo, Los Lunas, and Belen.

Bexfield and Anderholm (2002a) investigated water levels in deep nested piezometers in the Albuquerque area and found that the direction and magnitude of vertical hydraulic gradients differed substantially around the city. Vertical gradients in piezometer nests located in the Rio Grande flood plain and west of the river were directed primarily downward. In piezometer nests located outside of the flood plain to the east of the Rio Grande, vertical gradients were directed primarily upward, except in the two shallowest completions of a piezometer nest located near the mountain front. The largest downward gradients appeared to exist near the mountain front and west of the Rio Grande. However, seasonal differences in ground-water pumping affected both the direction and magnitude of gradients in some piezometer nests. Because data are not available from deep nested piezometers prior to sustained ground-water pumping, it is not known how well these vertical gradients represent predevelopment conditions.

Many investigators have attempted to identify and quantify the major sources of recharge to the aquifer system of the basin. Estimates of the quantity of recharge contributed by various sources have been compiled in reports describing ground-water models of the basin, such as Kernodle and others (1995). Mountain-front recharge probably is one of the most important sources of recharge to the basin (Thorn and others, 1993; Anderholm, 2001). Mountain-front

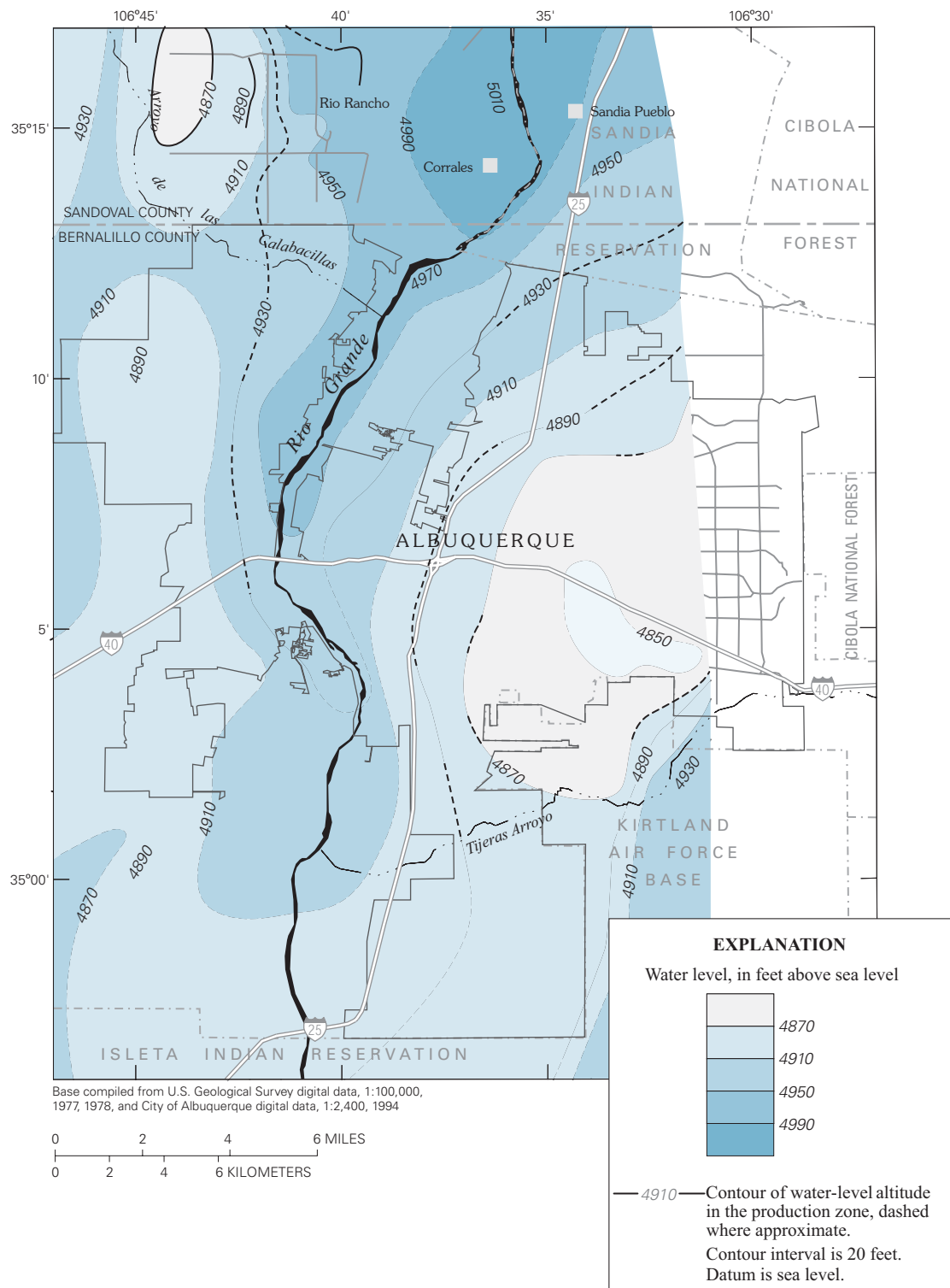


Figure 10. Water levels that represent 1999-2002 conditions in the production zone of the Santa Fe Group aquifer system in the Albuquerque area, New Mexico (modified from Bexfield and Anderholm, 2002a).

recharge is the general term used to describe recharge that occurs parallel to mountainous areas along the margins of a regional aquifer system, and usually refers more specifically to infiltration through mountain stream channels and to shallow subsurface inflow (Anderholm, 2001). Estimates of the quantity of mountain-front recharge occurring along the east side of the basin range from about 11,000 to 38,000 acre-feet per year (Anderholm, 2001). Mountain-front recharge could also be substantial along the Jemez Mountains in the northern part of the basin, and perhaps along the Ladron Mountains in the southwest, although studies have not been performed to establish the quantities of recharge occurring to the MRGB from these areas.

Subsurface ground-water inflow from adjacent basins also occurs along the margins of the MRGB, typically at fairly substantial depths. Along the northern margin of the basin, inflow occurs from the basin-fill deposits of the upgradient Española Basin and possibly from the Jemez volcanic deposits. Ground-water inflow along the entire northern margin has been estimated at about 14,300 acre-feet per year (Kernodle and others, 1995). Along the western margin, inflow probably occurs from Mesozoic rocks of the San Juan Basin toward the north and from Precambrian and Paleozoic rocks toward the south. Ground-water inflow along the entire western margin has been estimated at about 8,800 acre-feet per year (Kernodle and others, 1995). Deep ground-water inflow may also occur from Precambrian and Paleozoic rocks along the eastern margin of the basin, and from Paleozoic and Mesozoic rocks in the area of the Hagan Embayment (fig. 5), but the quantity of inflow occurring in these areas is not well known.

As discussed above, infiltration is known to occur from the Rio Grande and its tributary streams and arroyos, as well as from the irrigation systems in the Rio Grande flood plain. The quantity of water from these sources that contributes to ground-water recharge is affected by evapotranspiration. Also, in the Rio Grande flood plain, the quantity of recharge is dependent upon the complex interactions between the river and the canals and drains of the associated irrigation system. Kernodle and others (1995) estimated through use of their ground-water model that about 79,000 acre-feet of water was contributed to the aquifer system from the Rio Grande and associated canals during the year ending in March 1994. Substantial quantities of recharge from the Jemez

River, the Santa Fe River, Galisteo Creek, the Rio Puerco, Tijeras Arroyo, Abo Arroyo, and the Rio Salado (fig. 1) were also assumed in model simulation. Other sources that probably contribute to recharge, particularly in the Rio Grande flood plain (where depths to water typically are less than about 25 feet), include infiltrating irrigation water and septic systems. Residents of most areas that fall outside of the incorporated boundaries of the City of Albuquerque or selected other cities, such as Los Lunas and Belen, have individual septic systems. Although the direct infiltration of precipitation within the basin is a potential source of recharge to the aquifer system, any such infiltration is probably fairly localized and the quantity of recharge it contributes generally is believed to be minor (Anderholm, 1988).

Ground water discharges from the MRGB to the Socorro Basin near San Acacia (fig. 1). Ground-water discharge also occurs within the MRGB through evapotranspiration (particularly in the Rio Grande flood plain), ground-water pumpage, and discharge of ground water into drains and some reaches of the Rio Grande. The Kernodle and others (1995) ground-water model indicated that under predevelopment conditions, ground-water discharge occurred primarily through evapotranspiration. However, ground-water pumpage, estimated to be about 152,700 acre-feet in 1990 for all uses (Thorn and others, 1993), has substantially reduced the amount of ground-water discharge that occurs through evapotranspiration.

METHODS

Data Used in the Investigation

The data used in this investigation include both new data collected specifically for this USGS MRGB study and historical ground- and surface-water data from the USGS NWIS database and the City of Albuquerque. Details about site characteristics, sample collection, and sample analysis for each data source are discussed below.

New Middle Rio Grande Basin Study Data

Site Characteristics

For the MRGB study, more than 300 sets of samples were collected at 288 ground-water sites (wells and springs) across the basin (fig. 11, and appendix A) between June 1996 and August 1998. Sampling sites were selected primarily on the basis of location in an attempt to attain the best possible areal coverage of the basin. Efforts also were made to locate wells with discrete sampling intervals (in other words, short screened intervals) and groups of wells that allowed samples to be obtained from a variety of depths within the aquifer system at a given location. However, in most areas of the basin except in the vicinity of Albuquerque, so few wells were available for sampling that well construction was not a primary consideration. About 100 of the wells are located in or near Albuquerque.

Ground-water sampling sites consisted of 280 wells and 8 springs (table A1). Of the wells that were sampled, 116 were classified as monitoring wells (wells from which water is not obtained for any purpose other than monitoring of water level and ground-water quality), 82 were classified as production wells (wells used to supply water to more than 3 households or to industrial operations), 34 were classified as domestic wells (wells used to supply water to fewer than 3 households), 45 were classified as windmills (wells having a piston mechanism to lift water, which is used primarily to water livestock), and 3 were classified as stock wells (wells with submersible pumps, where water is used primarily for livestock). Well depths ranged from about 23 to 2,020 feet, with a median of about 500 feet. Screen lengths ranged from 5

to 1,270 feet, with a median of 20 feet. Casing material was steel in at least 167 wells and polyvinylchloride (PVC) in at least 108 wells; the material was not noted for 5 wells. Construction information for each category of well type is summarized in table 2, which shows that production wells typically were deepest but also had the longest screened intervals, whereas monitoring wells typically provided the most discrete sampling intervals.

All data from each ground-water source (wells and springs) are listed in appendix A. A unique 3-digit site number preceded by “S” (designated “Syyy”) was assigned to each ground-water site. Each water sample was assigned a unique 3-digit sample number preceded by “NM” (designated “NMxxx”). In some cases, multiple samples were collected at a particular site.

Based on the chemical and isotopic data of this investigation, source areas of recharge were identified and each water sample in appendix A was assigned to a primary hydrochemical zone, indicating the primary source area for recharge; in some cases, samples were identified as containing water from a secondary hydrochemical zone. Thirteen hydrochemical zones were recognized and numbered 1-13. The basis and definition of the various hydrochemical zones is discussed in the section “Tracing Sources of Water in the Middle Rio Grande Basin-- Definition of Hydrochemical Zones and Water Sources”. Data from some of the ground-water sites sampled for the MRGB study have not been included in the main body of data used for most analysis. These data are marked in the tables of appendix A with an “E” (for “exotic”) as the primary hydrochemical zone. Data for 14 sites (S009, S023, S038, S057, S063, S067, S070, S202, S249-251, S256, S258, and S273) were removed from the main data set because the sites fell outside the boundaries of the MRGB and may have produced water from an aquifer system other than that of the Santa Fe Group. Data for a few other selected sites were removed from the main data set because they were determined not to be representative of regional water quality. Examples are data from wells that were believed to produce water from a perched system not in hydraulic connection with the Santa Fe Group aquifer system (S094), wells that were believed to have been substantially affected by local contamination and/or evapotranspiration (S099, S129, S182, S225, and S282), wells that were believed to produce water associated primarily with geothermal systems (S028, S054, S112, and S211), and wells that

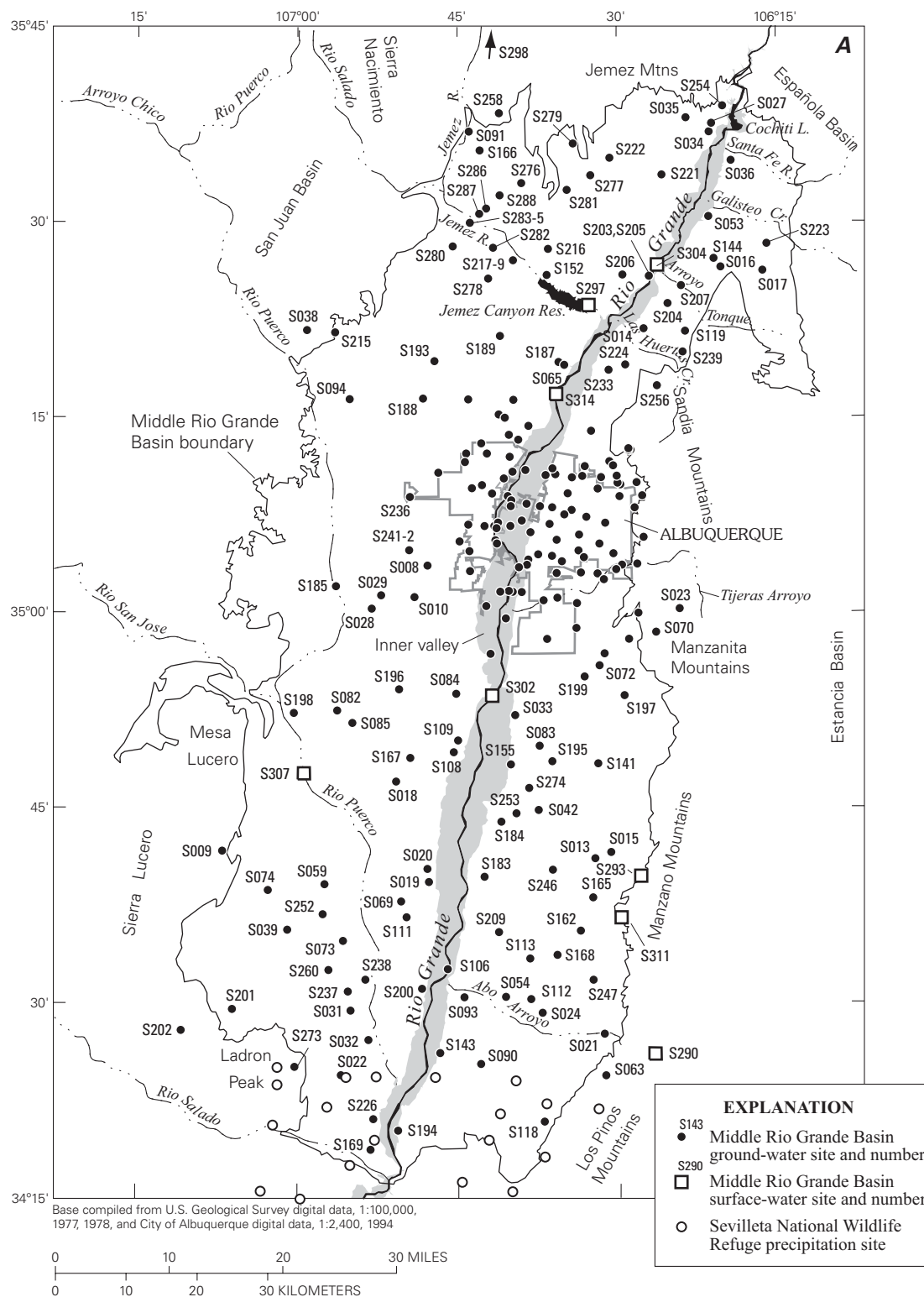


Figure 11a. Locations of ground-water and surface-water samples collected for the Middle Rio Grande Basin study outside the Albuquerque area, and locations of precipitation sites on Sevilleta National Wildlife Refuge (Moore, 1999).

Table 2. Statistical summary of well-construction information by well type, Middle Rio Grande Basin, New Mexico

[Length of sample interval is in feet; all other data are in feet below land surface; nd, not determined]

Parameter	Number of wells	Minimum value	Median value	Maximum value
<u>Domestic wells</u>				
Depth of well	34	55	379	985
Depth to top of sample interval	27	120	390	965
Depth to bottom of sample interval	27	130	400	980
Length of sample interval	27		20	40
Depth to water	32	8	252	530
<u>Monitoring wells</u>				
Depth of well	116	23	394	1,805
Depth to top of sample interval	115	10	349	1,634
Depth to bottom of sample interval	115	20	415	1,795
Length of sample interval	115	5	10	270
Depth to water	115	6	98	887
<u>Production wells</u>				
Depth of well	81	81	1,000	2,020
Depth to top of sample interval	79	19	425	1,355
Depth to bottom of sample interval	79	81	950	2,000
Length of sample interval	79	18	400	1,270
Depth to water	81	4	269	1,101
<u>Stock wells</u>				
Depth of well	3	120	192	460
Depth to top of sample interval		nd	nd	nd
Depth to bottom of sample interval		nd	nd	nd
Length of sample interval		nd	nd	nd
Depth to water	2	107	139	171
<u>Windmills</u>				
Depth of well	44	42	291	1,109
Depth to top of sample interval	7	125	269	715
Depth to bottom of sample interval	7	135	279	725
Length of sample interval	7	5	10	40
Depth to water	33	13	207	991

were believed to have been affected only locally and at relatively shallow depths by surface-water bodies (S091 and S152). Some of these data are nevertheless discussed in certain sections of this report; the *exotic* waters discussed in such sections are explicitly identified.

Each ground-water site, the site number, site name, location, altitude, depth, water level, and other well construction information is listed in table A1. Also listed are the primary and secondary hydrochemical zone numbers, assigned as a part of this investigation.

All the chemical and isotopic data for ground-water samples collected as a part of this study are listed in tables A2-A12. The site number, sample number, site name, date of collection, and primary and secondary hydrochemical zone is listed for each sample (except for those in table A10 that contain isotopic data specific for City of Albuquerque production wells that were sampled only for ^2H and ^{18}O isotopic composition). The water-quality parameters measured in the field (temperature, dissolved oxygen, pH, and specific conductance), and the major-element chemical composition (Ca, Mg, Na, K, Cl, Br, SO_4 , and HCO_3) are

listed in table A2. The concentrations of selected minor constituents (Sr, SiO₂, Fe, Mn, NO₃ (as N), and F) are listed in table A3. The concentrations of selected trace elements (Al, B, Ba, Li, Zn, Pb, Cu, Rb, V, Cr, Co, Mo, As, Se, and U) for each ground-water sample are listed in table A4. Not all constituents were measured for each water sample.

The concentrations of selected dissolved gases (N₂, Ar, O₂, CO₂, and CH₄) are listed in table A5 for each ground-water sample. Additional values given in table A5 are discussed in a later section of this report and include the assigned altitude of recharge, a calculated concentration of excess N₂ from denitrification, and the recharge temperature and excess air calculated at the recharge altitude. The results of a sensitivity calculation of the recharge temperature and excess air as a function of recharge altitude, calculated with and without consideration of denitrification, are summarized in table A6.

The concentrations of the chlorofluorocarbons (CFCs), CFC-11, CFC-12, and CFC-113, in picograms of CFC compound per kilogram of water (pg/kg) are given in table A7 for each ground-water sample. The CFC concentrations are also given as the calculated atmospheric partial pressure and percentage of modern concentrations. For uniformity of results, and because the presence of most CFCs indicate anthropogenic sources rather than recharge conditions, the calculated atmospheric partial pressures and percentage of modern concentrations of CFCs in ground-water samples assume an altitude of 5,000 feet above sea level and temperature of 13.5°C. For some water samples containing recent recharge (water recharged within the past 50 years), the atmospheric partial pressure and percent modern concentration should be re-calculated at the actual recharge altitude and temperature, if recharge date is desired (Plummer and Busenberg, 1999). Because of uncertainties in recharge conditions resulting from infiltration of water through relatively deep unsaturated zones and/or contamination from anthropogenic sources, ground-water age based on CFCs was not evaluated as a part of this report. Instead, the CFC data were used to recognize water samples that contained at least a fraction of post-1940's water.

The concentrations of dissolved sulfur hexafluoride and dissolved helium gas determined by gas chromatography at the U.S. Geological Survey Dissolved Gas Laboratory, Reston, Virginia, and by

mass spectrometric methods at the Noble Gas Laboratory, Lamont-Doherty Earth Observatory of Columbia University, Palisades, New York, respectively, are summarized in table A8.

The values of the stable H and O isotopic compositions of water, $\delta^2\text{H}$ and $\delta^{18}\text{O}$, and stable S isotopic composition of dissolved SO₄, $\delta^{34}\text{S}$, are given for each ground-water sample in table A9. The stable isotopic composition of water from City of Albuquerque production wells analyzed by the U.S. Geological Survey Stable Isotope Laboratory, Reston, Virginia are given in table A10. Water samples from City of Albuquerque wells collected as a part of this study (1996-97), archived samples collected by C. Yapp from the early 1980's and re-analyzed by the USGS as a part of this study, and additional water samples collected by City of Albuquerque personnel from all City of Albuquerque production wells that were in operation during July-August 1997 are included in the table. Values of $\delta^2\text{H}$ reported by Yapp (1985) and values of $\delta^2\text{H}$ and $\delta^{18}\text{O}$ reported by Lambert and Balsley (1997) are included for comparison.

The tritium concentration and uncertainty in the tritium measurement, the CFC-12 concentration, the stable carbon isotopic composition of DIC, $\delta^{13}\text{C}$, the measured ¹⁴C activity and standard deviation of the measurement, and the unadjusted radiocarbon age in years are given for each ground-water sample in table A11. The inclusion of CFC-12 and tritium concentration is redundant with other tables but useful in recognizing water samples that may contain fractions of post-nuclear detonation (post-bomb) ¹⁴C activities.

Data collected for the determination of the ³H/³He age of water are summarized in table A12. The data were measured at the Noble Gas Laboratory, Lamont-Doherty Earth Observatory (LDEO) of Columbia University, Palisades, New York. Values of tritium concentration, $\delta^3\text{He}$, ³He/⁴He isotope ratio, the dissolved He and Ne concentration, the calculated values of $\Delta^4\text{He}$ and ΔNe in percent of solubility equilibrium, the uncorrected ³H/³He age (uncorrected for terrigenous helium sources) and the corrected ³H/³He age (corrected for presence of terrigenous helium assuming an ³He/⁴He ratio of 2×10^{-8} for terrigenous helium), and associated uncertainties are summarized in table A12. Although the tritium and helium data are referred to in this report, the ³H/³He ages are not, because the emphasis of the present study concerns

SUMMARY OF IMPLICATIONS FROM GEOCHEMICAL AND ISOTOPIC DATA FOR THE CONCEPTUAL MODEL OF THE AQUIFER SYSTEM

The chemical and isotopic data collected for ground water in the MRGB as a result of this study have enabled evaluation and refinement of the conceptual model of the aquifer system of the basin. Specifically, the data have been used to examine the following aspects of the system: (1) the major locations of recharge to the basin, (2) the long-term historic direction of ground-water flow, and relation of that flow to the predevelopment water levels, (3) possible origins of the ground-water trough, (4) the effects of major structural and stratigraphic features of the basin on ground-water flow, and (5) implications for the mechanism of recharge and timing of paleo-environmental conditions in the basin. The radiocarbon ages and locations of the hydrochemical zone boundaries from this study also have been used to help calibrate a ground-water-flow model for the basin (Sanford and others, 2004). Modifications to the conceptual model based on geochemical data now available for the basin also have been incorporated into a recent revision of the USGS flow model (McAda and Barroll, 2002). This section summarizes some of the major implications of the geochemical and isotopic data for the conceptual model of the flow system and, where appropriate, describes how the previous conceptual model has changed as a result of this study. As was stated earlier in this report, the geochemical data used in this study were collected from wells completed in the upper few thousand feet of the Santa Fe Group aquifer system, so that the findings presented here may not all apply throughout the entire depth of the basin.

Tracing Sources of Water Through the Basin

This study has shown that ground water of the MRGB can be grouped into zones of distinctly different chemical “signatures” that typically have well-defined boundaries. These zones enable the delineation of waters having separate recharge sources, and their extents facilitate determination of the relative importance of each recharge source. In most cases, ground water having separate recharge sources differs in both major-/minor-element chemistry and isotopic

composition. In other cases, the use of isotopic data, particularly ^2H , ^{18}O , and ^{14}C , is essential to differentiating waters of separate origin. The individual chemical and isotopic “signatures” within the basin can be associated with the observed composition of recharge water from particular sources, or with the expected composition based on the geology and physiography of a potential recharge area. This study has demonstrated that, because the sediments of the Santa Fe Group aquifer system are relatively unreactive, the chemical composition of ground water in the basin generally changes little along a flow path. This consistency in composition allows source-water associations to be traced over long distances through the aquifer system.

The distribution and extent of the 13 hydrochemical zones defined for the MRGB (the chemical and isotopic “signatures” of which have been discussed in the section “Tracing Sources of Water in the Middle Rio Grande Basin-- Definition of Hydrochemical Zones and Water sources”) indicate the presence of multiple sources of recharge to the aquifer system of the basin. Mountain-front recharge sources appear to dominate over large areas of the basin, particularly along the northern and eastern margins of the basin, as well as in the southwestern corner. Mountain-front recharge was previously known to be important along the Sandia, Manzanita, and Manzano Mountains of the eastern basin margin (Kernodle and others, 1995). Mountain-front recharge was also suspected to contribute water to the aquifer system along the Ladron Mountains in the southwest (Kernodle and Scott, 1986), but no previous studies presented direct evidence of the importance of this source. The importance of mountain-front recharge water as opposed to ground-water inflow along the Jemez Mountains in the north also does not appear to have been well characterized previously. Although mountain-front recharge appears to occur along the entire northern boundary of the basin that is bordered by the Jemez Mountains, the chemical composition of recharge water along the boundary is not uniform, which indicates that the recharge mechanisms and/or characteristics of the flow paths may vary. In particular, water of the West-Central zone, which originates in the western part of the Jemez Mountains and extends southward through most of the length of the basin (fig. 78), differs considerably in chemical and isotopic composition from water recharging in the eastern part of the mountains. Most water of the West-Central zone may actually enter the

basin as underflow even though the water apparently originated at high altitude, as indicated by ^2H and ^{18}O . This water may have infiltrated in the area of the Valles Caldera and migrated to a substantial depth before reaching the basin.

Another important source of recharge to the aquifer system of the MRGB, particularly in the area of Albuquerque, is infiltration from the Rio Grande. This study has helped to define both the extent and age of water sourced at the river. The ^2H and ^{18}O compositions of ground water have been particularly useful in delineating the river's area of influence, as indicated by the boundaries of the Central zone, which has been shown to extend from about San Felipe on the north to near Abo Arroyo on the south (fig. 78). Ground water of the Central zone extends 2 to 3 miles beyond the flood plain of the Rio Grande—particularly to the east—and has been identified at substantial depth (on the order of 2,000 feet). This study indicates that water has been entering the aquifer system from the Rio Grande for tens of thousands of years and has historically flowed primarily north to south under the Albuquerque area (figs. 64, 74, and 78). Therefore, movement of infiltration from the Rio Grande beyond the flood plain has not occurred primarily as the result of water-level declines associated with intense pumping of ground water for municipal supply. The long-term contribution of this source of recharge to a large part of the central area of the basin is probably of greater magnitude than previously thought.

This study has confirmed that infiltration from ephemeral streams can be an important source of recharge in some areas of the MRGB. In particular, recharge from the Rio Puerco appears to be one of the primary sources of water to the aquifer system in the western part of the basin. The chemical and isotopic composition of ground water associated with the Rio Puerco can be identified across a large area extending both east and west of the present stream channel. Such a large area of influence does not appear to have been previously recognized, and may result partly from a lack of other substantial recharge sources in the western part of the basin. Similarly, infiltration through Abo Arroyo has been shown to have a large area of influence in the southeastern part of the basin. Ground water that has infiltrated from Tijeras Arroyo also can be identified by its chemical and isotopic composition, but has extended over only a fairly limited area compared to infiltration from the Rio Puerco and Abo Arroyo, possibly because of a high quantity of moun-

tain-front recharge in the area. Infiltration from the Jemez River appears to be limited primarily to a relatively narrow and shallow area located directly along the river, which differs from the previous assumption that the Jemez River is an important source of recharge to the aquifer system in the northern part of the basin (Kernodle and others, 1995).

The underflow of ground water from adjacent basins has been confirmed to contribute water to the aquifer system of the MRGB along its western margin, as well as in limited areas of the eastern margin. Ground water from Paleozoic and Mesozoic rocks to the west appears to leak into the basin and mix with a greater quantity of water that infiltrates locally, possibly from ephemeral streams. Ground water also leaks into the northeastern part of the MRGB from the Mesozoic rocks of the Hagan Basin. The area that shows evidence of this underflow is not extensive, which indicates that the quantity of ground water entering the MRGB from the Hagan Basin probably is relatively small. In the area of the Tijeras Fault Zone, mineralized water associated with the fault system appears to leak into the basin. This water then mixes with local mountain-front recharge over a fairly broad area, and perhaps migrates to depth below the more plentiful and more dilute mountain-front water. Overall, the results of this study have shown that the quantity of underflow entering the MRGB along the western and eastern margins is probably less than previously determined (Kernodle and others, 1995). However, as mentioned above, underflow from the area of the Jemez Mountains may contribute a substantial quantity of recharge to the basin.

Direction of Ground-Water Flow

Both conservative geochemical tracers and boundaries between hydrochemical zones indicate that ground-water flow in the MRGB has historically been directed primarily north-south through the center of the basin, with a greater east-west component near the basin margins. As discussed in the section "Chemical and Isotopic Composition of Ground Water in the Middle Rio Grande Basin", values of specific conductance, Cl, and the stable isotopes of water indicate that the dominant component of ground-water flow in the basin is from north to south. Similarly, the orientations of the Central, West-Central, and Rio Puerco hydro-

chemical zones in particular demonstrate the importance of north-south flow (fig. 78).

Although predevelopment hydraulic head maps for the MRGB (Titus, 1961; Bjorklund and Maxwell, 1961; Bexfield and Anderholm, 2000) are broadly consistent with the primary direction of ground-water flow indicated by geochemical data, there are some apparent inconsistencies. In particular, predevelopment hydraulic heads indicate greater east-west components of flow through the center of the basin than is indicated by the orientation of the hydrochemical zones. The differences in flow direction indicated by hydraulic heads and geochemical tracers likely are associated with the different time horizons and depths represented by the two types of information.

Carbon-14 data from this study of the MRGB indicate that most ground water resides in the aquifer system for thousands to tens of thousands of years. Therefore, the hydrochemical zones defined on the basis of current ground-water chemistry in the basin probably reflect aquifer-system conditions (including ground-water-flow directions) present on the order of thousands of years into the past, and may not yet reflect the current hydraulic-head distribution. In addition, predevelopment hydraulic-head maps for the basin were based on conditions primarily in the shallow part of the aquifer system (to a depth of at most a few hundred feet), whereas geochemical data were collected from wells reaching greater depths, where ground-water flow directions may differ.

As a result of this study, the conceptual model of how ground water moves through the aquifer system of the MRGB has been refined. The geochemical data indicate that ground water in the Albuquerque area has not flowed almost directly west from the eastern mountain front to the inner valley of the Rio Grande, as previously reported (Kernodle and Scott, 1986). Instead, the water chemistry beneath Albuquerque indicates that ground water flows mostly north to south through the city area, and this flow has occurred over at least the past 20,000 years. The rather distinct boundary between water sourced from the Rio Grande and mountain-front recharge water to the east (as most clearly indicated by a substantial difference in ^2H and ^{18}O across a short distance, 1 to 2 miles) indicates that similar flow conditions prevailed in the area for many thousands of years, relatively unaffected by dispersion or mixing. However, recharge from the Rio Grande does not appear to have moved a substantial distance toward the ground-water trough that is shown by

hydraulic-head maps present in the western part of the basin. Previous conceptual models assumed that much of the ground water in the area of the trough was sourced at the river (Yapp, 1985; Logan, 1990), or possibly even the mountain front (Logan, 1990).

Origin of the Ground-Water Trough

The predevelopment hydraulic-head maps of Meeks (1949), Titus (1961), Bjorklund and Maxwell (1961), and Bexfield and Anderholm (2000) have all indicated the presence of a ground-water trough in the western part of the MRGB, extending from near the Jemez River on the north to Belen on the south, and from a series of major faults on the west to near the Rio Grande on the east (fig. 9). Water levels in the trough can be as much as 50 feet lower than the level of the Rio Grande directly to the east. Geochemical data from this study show that ground water flowing along the axis of the trough occurs within the West-Central hydrochemical zone and likely originated in the area of the Jemez Mountains during the last glacial period. Although hydraulic heads indicate that water sourced both at the western margin of the basin and at the Rio Grande should be flowing into the trough, the geochemical data indicate that this flow has not occurred. The lack of evidence that these waters have reached the axis of the trough appears to contradict one conceptual model that the trough is present primarily as the result of a large thickness of permeable materials near its axis as compared to along its flanks (though still quite impermeable compared with other parts of the basin) (Kernodle and others, 1995). Such a configuration would cause the area to act as a sort of “drain” for the aquifer system in the western part of the basin. Recent geohydrologic data for the area do not support this conclusion (Hawley, 1996).

One possible explanation for the presence of the trough that would be consistent with the observed geochemical data is that the trough is a transient feature of the aquifer system in the MRGB. As mentioned above, changes with time in the quantity and distribution of recharge around the basin could result in a change in the distribution of hydraulic head. If the trough developed relatively recently as a result of such changes, water sourced at the western margin of the basin and at the Rio Grande may not yet have moved into the trough area. Radiocarbon ages of waters sourced from the western margin that are located today

just west of the trough, and Rio Grande waters just east of the trough, are approximately 10 ka. Waters younger than 10 ka are observed farther to the west and east toward the Rio Puerco and Rio Grande sources, respectively. This suggests that water has been flowing toward the trough from the Rio Puerco and Rio Grande since the beginning of the Holocene. Using a ground-water-flow model for the MRGB, Sanford and others (2001) found that the trough could be simulated merely by lowering the quantities of recharge from certain known sources of water to the basin.

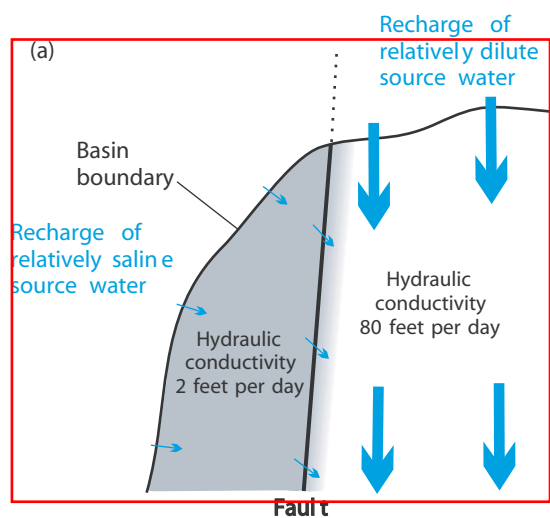
Another factor that could contribute to the presence of the trough could be the existence of features that limit ground-water flow into the trough from the east and west. Such features could include major faults that are relatively impermeable or low-conductivity geologic strata. Most major faults within the MRGB trend north to south; therefore, cementation of any of several major faults could be consistent with the restriction of east-west ground-water flow. Also, the presence of a dipping geologic layer of low permeability could retard east-west flow. For example, Connell and others (1998) have shown that the fine-grained Atrisco Member of the Arroyo Ojito Formation dips downward from west to east beneath the Albuquerque area. This layer could restrict flow of infiltration from the Rio Grande westward toward the trough. The same or a similar layer of low-permeability material might possibly also restrict the eastward flow of water infiltrating through the Rio Puerco. Further research would be necessary to conclude whether these types of restrictions are present in such a configuration that would contribute to the presence of the trough. Such restrictions would act only to limit flow toward the trough and not to prevent it entirely. Therefore, even if flow restrictions are important, the absence of water sourced from the Rio Grande and from the western margin near the axis of trough indicates that the trough is likely a transient rather than a long-term feature.

Ground-Water Flow in Relation to Faults and Other Structural Features

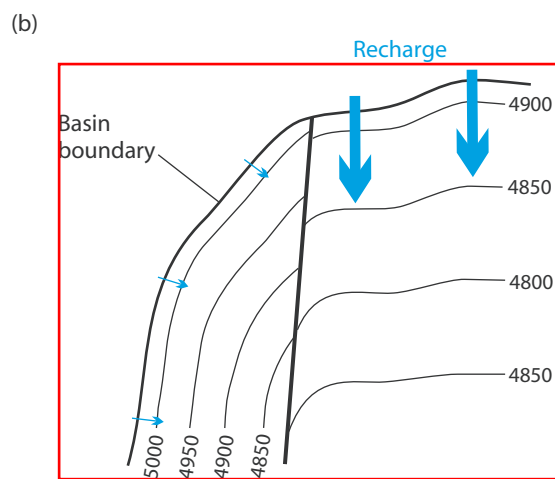
Because ground-water chemistry in the MRGB enables delineation both of waters from particular recharge sources and of ground-water-flow directions, chemistry can be used to investigate the possible effects of faults on regional ground-water flow in the basin. In

particular, correspondence of a boundary between hydrochemical zones with the location of a fault could be an indication that horizontal ground-water flow is restricted across that fault. Flow across a fault can be restricted by reduced permeability and hydraulic conductivity in the fault zone resulting from cementation and/or sediment deformation, or by the juxtaposition of relatively high-conductivity aquifer materials on the upgradient side of the fault against relatively low-conductivity materials on the downgradient side. Alternatively, correspondence of a chemical boundary with a fault could indicate that recharge rates and hydraulic properties of the aquifer materials differ substantially on either side of the fault. For example, if recharge rates and hydraulic conductivity are much greater on one side of the fault than the other, the quantity of water from the low-recharge source that can mix with water from the high-recharge source will be relatively small and will probably have little effect on the water chemistry on the high-conductivity, high-recharge side of the fault (fig. 118). Where either horizontal flow across a fault is restricted, or recharge rates and hydraulic conductivity differ greatly on either side of a fault, a hydraulic discontinuity may result, as would be indicated by a substantial difference in water-level altitudes across the fault (fig. 118).

In the MRGB, a few boundaries between hydrochemical zones appear to correspond fairly closely in places with “major” faults identified by Mark Hudson and Scott Minor (U.S. Geological Survey, written commun., 1999). Such regional differences in ground-water chemistry in the vicinity of faults, indicating possible effects of faults on ground-water flow, do not appear to have been previously identified in the basin. Two examples are the boundary between the Rio Puerco and West-Central zones and the boundary between the Tijeras Fault Zone and Eastern Mountain Front zones. In the northern half of the basin, the boundary between the Rio Puerco and West-Central zones corresponds fairly well in places with the Tenorio and Sand Hill faults identified by Kelley (1977). The boundary between the Tijeras Fault Zone and Eastern Mountain Front zones corresponds fairly well in places with the Tijeras and Colorada faults, also identified by Kelley (1977). In the areas of both the Tenorio/Sand Hill faults and the Tijeras/Colorada faults, the predevelopment water-level map of Bexfield and Anderholm (2000) shows the occurrence of hydraulic discontinuities that indicate the likely effects of these faults on ground-water flow.



NOT TO SCALE



NOT TO SCALE

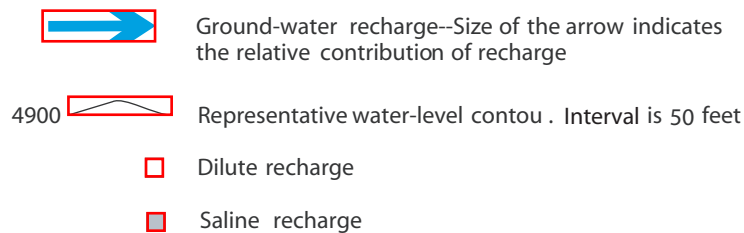


Figure 118. Schematic diagrams of (a) representative change in water chemistry across a fault in the Middle Rio Grande Basin, New Mexico, with greater recharge rates and hydraulic conductivity on one side of the fault relative to the other, and (b) representative discontinuity in the water levels across the fault.

The general correspondence of differences in ground-water chemistry with particular fault locations may indicate that horizontal ground-water flow is restricted across those faults—flow being primarily from west to east in the case of the Tenorio/Sand Hill faults, and from east to west in the case of the Tijeras/Colorada faults. The area surrounding the Sand Hill fault has been shown to include a zone of deformation that might reduce horizontal flow across the fault (Rawlings and Goodwin, 2001); also, the Sand Hill fault zone has been shown to be preferentially cemented by calcite (Mozley and Goodwin, 1995; Heynekamp and others, 1999). The Tijeras/Colorada faults offset different geologic units by hundreds to thousands of feet, and a contrast in hydraulic properties between the juxtaposed units (as opposed to cementation or deformation in the fault zone) could be the primary explanation for any restriction to horizontal ground-water flow in the area of these faults. However, in the case of both the Tenorio/Sand Hill faults and the Tijeras/Colorada faults, the aquifer materials on the downgradient side of the fault zone are much thicker and more conductive than materials on the upgradient side. Also, in both cases, recharge to the hydrochemical zone on the downgradient side (the West-Central zone or Eastern Mountain Front zone) is believed to be much greater than recharge to the zone on the upgradient side (the Rio Puerco zone or Tijeras Fault Zone zone). Therefore, the chemical differences observed across the faults at the boundaries between the hydrochemical zones could result from mixing of a relatively small quantity of water from the upgradient zone with a much higher quantity of water from the downgradient zone (as described earlier in this section).

Whereas differences in ground-water chemistry appear across sections of the “major” faults mentioned above, differences in chemistry are not evident in other areas where important—even basin-bounding—faults are present. For example, the water-level map of Bexfield and Anderholm (2000) indicates the presence of a hydraulic discontinuity along the Sandia fault adjacent to the front of the Sandia Mountains. However, the data available in the area from this study indicate that the distinct chemical boundary between water sourced from the Rio Grande and water sourced along the eastern mountain front occurs several miles west of the Sandia fault and is apparently unrelated to the location of the fault. The data do not show a difference in ground-water chemistry across the fault itself,

indicating that ground-water flow is not restricted across the fault. The hydraulic discontinuity in this area is probably the result of the large saturated thickness of conductive materials on the west side of the fault relative to the small saturated thickness of materials of similar hydraulic conductivity on the east side of the fault. This configuration results in a much greater transmissivity and a correspondingly flatter hydraulic gradient on the west side of the fault. Therefore, not all “major” faults in the basin appear to have characteristics that appreciably limit ground-water flow across them. Similarly, there appears to be little indication from either chemistry or water levels indicating that “minor” faults with small offset of similar sediments substantially limit horizontal ground-water flow, at least on a regional scale.

As discussed in earlier sections, the geochemical data from this study have provided new information that the presence of major faults may actually enhance or cause vertical ground-water flow in some areas of the MRGB. Elevated water temperatures and Cl and As concentrations, in particular, usually are in close proximity to “major” faults in the basin, especially at locations in the northern part of the basin, the northeastern part of Albuquerque, and the southeastern part of the basin. These elevated concentrations appear to be indicative of ground water from a deep source that is moving upward to mix with water at shallower depths of the aquifer system. The association of these elevated concentrations with the locations of faults indicates that vertical movement of ground water could be enhanced in fault zones by increased vertical conductivity resulting from deformation. Elevated Cl and As concentrations also are common in the vicinity of the structural highs located near the northern and southern extents of Albuquerque that separate the subbasins within the MRGB, as well as near the structural high/constriction that separates the MRGB from the Socorro Basin to the south. The thinning of Santa Fe Group deposits (which are relatively conductive compared to basement rocks) across structural highs could result in upward movement of ground water, both along faults that bound these highs and within the entire aquifer system. Ground water would tend to move to shallower depths of the aquifer system across structural highs in order to maintain flow within the more conductive sediments. Upward movement of ground water along faults and near structural highs/constrictions was not previously a part of the conceptual model for the basin.

Stratigraphic Controls on Ground-Water Flow

The geochemical data collected for this study indicate that stratigraphy may be important in controlling ground-water flow in some areas of the basin. Several deep piezometer nests show substantial variability in ground-water chemistry (including ground-water age) with depth. In some cases, water from different depths in the same piezometer nest belong to different hydrochemical zones, and, therefore, have different recharge sources and flow paths. Such variability indicates a lack of vertical mixing in some areas of the aquifer system, which could be the result of anisotropy and/or differences in hydraulic properties of aquifer materials with depth. Therefore, evidence from this study suggests that stratigraphic controls may act to substantially limit vertical ground-water movement on a regional scale in the basin. However, as mentioned in the previous section on ground-water flow related to faults, it also appears that stratigraphy/structure may actually enhance vertical ground-water movement in areas where structural highs cause relatively impermeable materials to be present closer to the land surface and/or in areas where major faults are present.

As mentioned above in relation to the ground-water trough west of Albuquerque, extensive layers of particularly fine-grained material may affect not only the vertical, but also the horizontal direction of ground-water flow in the MRGB on a regional scale. For example, because the fine-grained Atrisco Member dips to the east across the center of the basin and is present at depth below more conductive materials, it could limit both downward and westward movement of water sourced at the Rio Grande. Similarly, extensive coarse-grained layers also may affect ground-water flow. For example, in the vicinity of Albuquerque, Hawley and Haase (1992) identified the presence of a thick section of north-south trending coarse-grained ancestral Rio Grande deposits extending several miles east from the Rio Grande. Ground-water chemistry data and analysis from this study indicate that water sourced at the Rio Grande has historically flowed primarily north to south through this general area, which may be largely attributed to the presence of these high-conductivity sediments.

Source, Mechanism, and Timing of Recharge to the Middle Rio Grande Basin

The stable isotope data, in conjunction with the dissolved gas recharge temperatures and radiocarbon ages, have been useful in interpretation of source, mechanism and timing of recharge to the aquifer system. Recharge temperatures calculated from measured concentrations of dissolved N_2 and Ar vary widely throughout the MRGB. The minimum in the calculated N_2 -Ar recharge temperatures for a particular hydrochemical zone appears to be near the mean annual temperature for the assumed recharge area for that zone, and the maximum recharge temperature approaches that of ground water beneath the relatively deep unsaturated zones. Apparently, the dissolved gas recharge temperatures demonstrate cases of both focused (cold recharge temperatures) and diffuse flow (warm recharge temperatures) recharge mechanisms in semiarid regions such as the MRGB.

Some interesting patterns in paleo-recharge temperatures emerge when the recharge temperatures are examined spatially in the MRGB. Relatively warm recharge temperatures are found for most waters recharged along the northern margin of the basin (western parts of the Northern Mountain Front zone and throughout the Northwestern zone), indicating recharge primarily by diffuse infiltration through relatively deep and warm unsaturated zones. Waters from the northeast part of the Northern Mountain Front zone have cold recharge temperatures indicative of relatively shallow and/or focused recharge. Many of the waters from the West-Central zone have cold recharge temperatures, as cold as the expected 3-4°C for waters recharged at 8,000 feet along the flanks of the Jemez Mountains during the LGM. The waters from the West-Central zone with cold recharge temperatures were found beneath Northwestern zone waters in the northern part of the basin and traced south throughout the basin along the eastern-most half of the West-Central zone. Waters with relatively warm recharge temperatures also were measured in the West-Central zone and can be traced along the western-most half of the West-Central zone. The axis of the ground-water trough approximately divides West-Central zone waters with warm recharge temperatures (on the west side of the trough) from West-Central zone waters with cold recharge temperatures (on the east side of the trough).

During the last glacial period, water recharged to the West-Central zone varied widely in stable isotope composition and recharge temperature, indicating both diffuse and focused recharge of low- and high-altitude precipitation. The range in $\delta^2\text{H}$ of West-Central zone waters from the LGM suggests recharge that occurred over a 4,000-foot range in altitude.

Ground water in the Central zone was recharged by direct infiltration from the Rio Grande, and apparently records surface-water temperature at the time of recharge. The indication from the dissolved $\text{N}_2\text{-Ar}$ data is that the long-term average temperature of the Rio Grande that infiltrated into the Central zone varied only $\pm 1^\circ\text{C}$ from the modern mean annual temperature at Albuquerque over the past 27 ka, even though the regional mean annual temperature was likely some 5°C colder during the LGM than today. The stable isotope composition of paleo Rio Grande water was relatively depleted in ^2H during the mid-Holocene warm period and relatively enriched in ^2H in the pluvial following the LGM. These observations may indicate that during the past, the timing of the spring runoff of northern New Mexico and southern Colorado snowmelt varied, coming late into early- to mid-summer during cold periods and overlapping, in part, with the summer monsoon season (currently July-October). During warm periods, such as modern times, the peak discharge of the Rio Grande occurred in mid- to late-spring, in advance of the summer monsoon season. These observations indicate that the Central zone is recharged primarily during the season of peak runoff of mountain snowmelt and that the season of peak runoff has shifted by as much as 2 months between the LGM and modern times.

During the last 5 ka, the $\delta^2\text{H}$ isotopic composition of Eastern Mountain Front recharge has decreased about 7 per mil, suggesting an average cooling of about 1.4°C following the mid-Holocene warm period. Over the same time span, the $\delta^2\text{H}$ isotopic composition of Rio Grande water has increased approximately 6 per mil, consistent with a shift in the season of peak snowmelt into the beginnings of the summer monsoon season.

SUMMARY

A large number of measurements of the chemical and isotopic composition of ground water and surface water were used to refine the conceptual model of ground-water flow in about the upper 2,000 feet of the Santa Fe Group aquifer system of the MRGB, and improve understanding of the water resources of the basin, as discussed in the previous section of this report. The extensive regional coverage, supplemented with data from the USGS National Water Information System and City of Albuquerque water-quality database, permitted definition and mapping of the spatial extents of 12 regional sources of water to the MRGB, and estimation of radiocarbon ages.

The chemical and isotopic analyses indicate that the composition of most ground water in the basin changed little after recharge, and can be traced over large distances (tens of miles) through the primarily siliciclastic sediment of the aquifer system. Radiocarbon ages adjusted for geochemical reactions in the aquifer system are nearly identical to the unadjusted radiocarbon ages and range from less than 1 ka to more than 30 ka.

Predominant sources of water to the basin include (1) recharge from mountains along the northern, eastern and southwestern basin margins (median age 5-9 ka); (2) seepage from the Rio Grande and Rio Puerco (median age 4-8 ka), and from Abo and Tijeras Arroyos (median age 3-9 ka); (3) inflow of saline water along the southwestern basin margin (median age 20.4 ka); and (4) inflow along the northern basin margin (median age 19.9 ka) that probably represents recharge from the Jemez Mountains during the last glacial period.

Piezometer nests provided critical information on variations in chemical and isotopic composition with depth to nearly 1,500 feet below the water table, particularly in the vicinity of Albuquerque. In some cases, the data from piezometers and other wells permitted characterization of the three-dimensional aspects of water composition. For example, the data indicate that paleowater (approximately 20 ka), present at the water table through the west-central part of the basin, extends beneath more recent mountain-front recharge in northern parts of the basin, and beneath water of Rio Grande origin under western parts of Albuquerque.

Age gradients from all available piezometer nests ranged from 0.1 to 2 yr cm⁻¹. Age gradients from two piezometer nests indicate "modern" (past few thousand years) recharge rates of approximately 3 cm yr⁻¹ for infiltration from the Rio Grande at Albuquerque and recharge along the eastern mountain front at Albuquerque. Historically, there have been higher amounts of recharge to the Eastern Mountain Front zone at Albuquerque than to vicinities either north or south of Albuquerque along the eastern mountain front.

Chlorofluorocarbon and/or tritium concentrations were measured in water samples at all wells where ¹⁴C samples were taken. These measurements permitted identification of samples that might be contaminated with nuclear bomb-era ¹⁴C or drilling fluid, and that might consequently have a young bias in radiocarbon age. Gradients in ¹⁴C activity of dissolved inorganic carbon with depth were determined from the piezometer nests and used to estimate uncertainty in radiocarbon age in mixtures of ground water pumped from wells that are open to hundreds of feet of the aquifer system.

Detailed study of the ²H and ¹⁸O isotopic composition of source waters to the basin helped to delineate flow directions and boundaries between water sources. The stable isotope data for ground water in the vicinity of Albuquerque indicate little movement of ground water in response to withdrawals from production wells during the past 20 years, even though, in places, the modern water table has fallen as much as 140 feet below the predevelopment potentiometric surface. Small shifts over the past 20 years in stable isotope composition of water discharged from public supply wells along the boundaries between the West-Central zone (paleowater) and Central zone (Rio Grande water) west of the Rio Grande, and along the boundary between the Central zone and Eastern Mountain Front zone east of the Rio Grande, indicate local areas where paleowater of Rio Grande origin likely is beginning to spread west and east in response to ground-water pumping in the vicinity of Albuquerque.

The geochemical data show that ground water flowing along the axis of a ground-water trough in the western part of the MRGB, extending from near the Jemez River on the north to Belen on the south, and from a series of major faults on the west to near the Rio Grande on the east, originated in the area of the Jemez Mountains during the last glacial period. Although hydraulic heads indicate that water sourced both at the western margin of the basin and at the Rio Grande

should be flowing into the trough, the geochemical data indicate that this flow has not occurred. The absence of water sourced from the Rio Grande and from the western margin near the axis of ground-water trough indicates that the trough is probably a transient rather than a long-term feature of the aquifer system.

Over the past 5 ka, the isotopic composition of water recharged along the eastern mountain front has become depleted in ^2H by about 7 per mil, consistent with an average cooling of about 1.4°C . Over the same period, $\delta^2\text{H}$ of ground water recharged from the Rio Grande increased by nearly 6 per mil from a minimum near -98 per mil. Seasonal shifts in the timing of peak discharge and/or historical changes in the amounts of low-altitude precipitation along the basin margins may account for the long-term trends in the stable isotopic composition of Rio Grande water. The $\delta^2\text{H}$ of water recharged from the Rio Grande to the Central ground-water zone (ground water of Rio Grande origin) was low at approximately 5 ka B.P., a maximum at about 15 ka B.P., and a minimum at about 22 ka B.P.

The recharge temperatures calculated from dissolved N_2 and Ar concentrations in paleowater recharged from the Rio Grande indicate that the long-term average water temperature at the time of predominant infiltration was nearly constant for the past 27 ka, and within about $\pm 1^\circ\text{C}$ of the modern mean annual temperature. This observation is consistent with seasonal shifts in the timing of peak discharge on the river. Where infiltration occurs through deep (greater than 300 feet deep) unsaturated zones, the dissolved-gas recharge temperatures are biased warm by as much as 10°C relative to the mean annual temperature (13.6°C at Albuquerque).

The $\delta^{13}\text{C}$ isotopic composition of dissolved inorganic carbon in ground water is remarkably constant throughout most of the basin, indicating a nearly constant historical predominance of C_4 over C_3 plants in recharge areas for the aquifer system, and little effect of geochemical reactions on radiocarbon age. However, during the past 1 ka, or perhaps even more recently, the $\delta^{13}\text{C}$ isotopic composition of dissolved inorganic carbon of water recharged along the basin margins and at the Rio Grande has become depleted in ^{13}C . This ^{13}C depletion could indicate a recent increase in C_3 plant abundance relative to C_4 plants, as has been observed in other parts of the southwest U.S.

Extensive chemical and isotopic datasets, such as obtained for the MRGB during this and previous

studies, are uncommon in the hydrologic sciences. This study demonstrates the benefits of obtaining a diverse and extensive chemical and isotopic dataset when characterizing hydrochemical processes in ground-water systems, retrieving historical environmental information from ground water, and/or refining conceptual models of ground-water systems.