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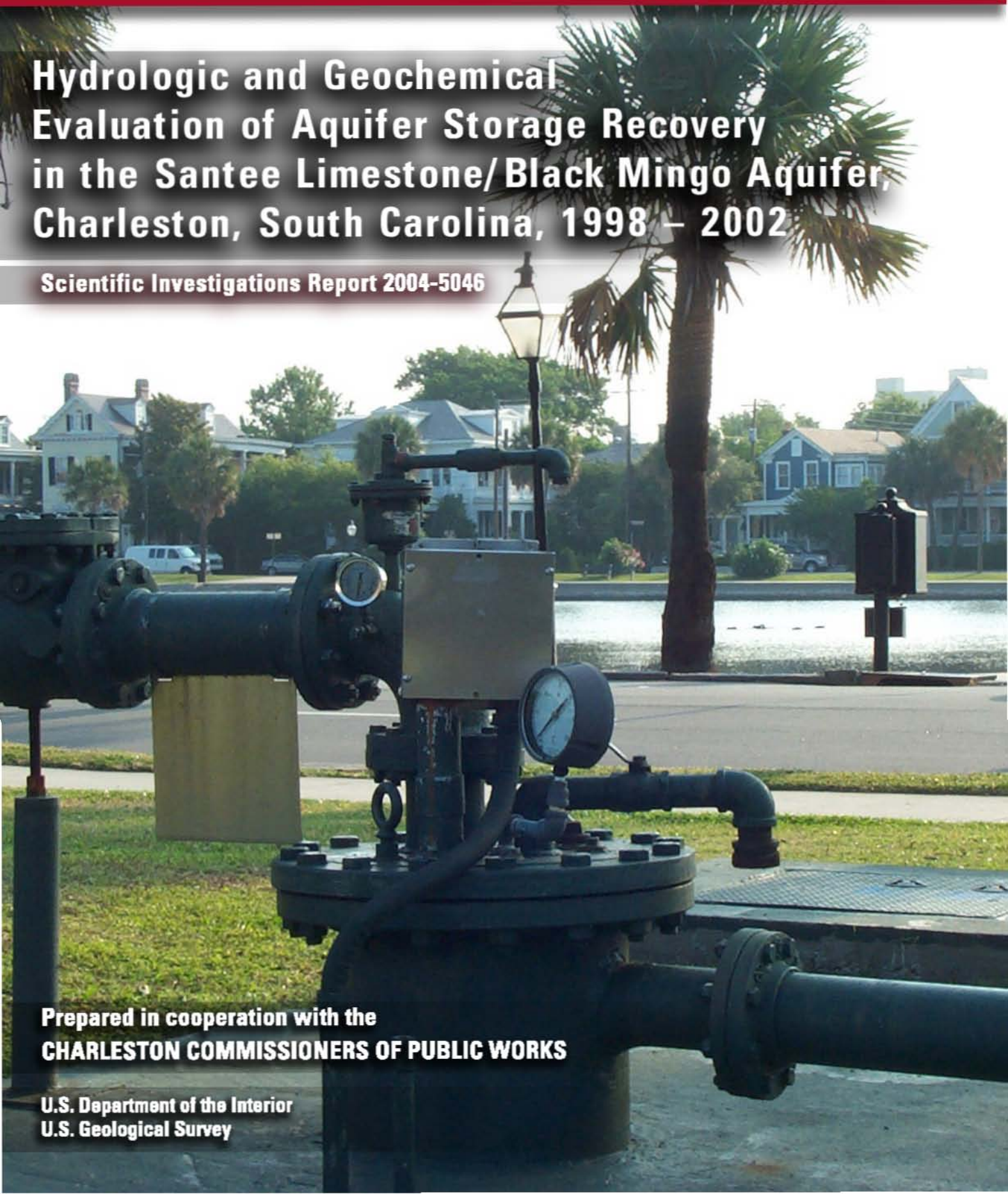
# Hydrologic and Geochemical Evaluation of Aquifer Storage Recovery in the Santee Limestone/Black Mingo Aquifer, Charleston, South Carolina, 1998 – 2002

Scientific Investigations Report 2004-5046



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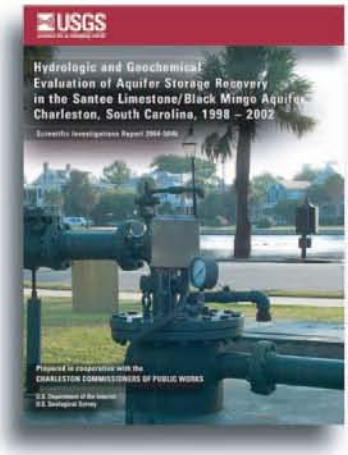


***COVER PHOTO:*** Production well CHN-812 (324618080560900) located near Colonial Lake in Charleston, South Carolina.

*Photograph by Kevin J. Conlon*

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# Figures

# TABLES

# Conversion Factors and Datum

| Multiply | By | To obtain |
|---|---|---|
| | **Length** | |
| centimeter (cm) | 0.3937 | inch (in.) |
| meter (m) | 3.281 | foot (ft) |
| kilometer (km) | 0.6214 | mile (mi) |
| | **Volume** | |
| cubic meter ($m^3$) | 35.31 | cubic foot ($ft^3$) |
| liter (L) | 0.2642 | gallon (gal) |
| | **Flow rate** | |
| cubic meter per minute ($m^3$/min) | 35.31 | cubic foot per minute ($ft^3$/min) |
| liter per second (L/s) | 15.85 | gallon per minute (gal/min) |
| million liters per day (ML/d) | 0.2642 | million gallons per day (Mgal/d) |
| | **Mass** | |
| gram (g) | 0.03527 | ounce avoirdupois (oz) |
| kilogram (kg) | 2.205 | pound avoirdupois (lb) |
| | **Pressure** | |
| kilopascal (kPa) | 0.1450 | pound-force per square inch (lbf/$in^2$) |
| | **Hydraulic conductivity** | |
| meter per day (m/d) | 3.281 | foot per day (ft/d) |
| | **Transmissivity*** | |
| meter squared per day ($m^2$/d) | 10.76 | foot squared per day ($ft^2$/d) |

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

$$°F = (1.8 \times °C) + 32$$

**Datums:**

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29). To convert to the North American Vertical Datum of 1988 (NAVD 88), subtract 0.3 meter from altitudes reported herein.

Horizontal coordinate information is referenced to the North American Datum of 1927 (NAD 27).

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

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

**Acronyms and Abbreviations:**

| | |
|---|---|
| ANC | acid-neutralizing capacity |
| ASR | aquifer storage recovery |
| bls | below land surface |
| $CaCO_3$ | calcium carbonate |
| CCPW | Charleston Commissioners of Public Works |
| CEC | cation exchange capacity |
| $CO_2$ | carbon dioxide |
| $d^{-1}$ | per day |
| DOC | dissolved organic carbon |
| EMFM | electromagnetic flow meter |
| eq/L | equivalents per liter |
| g/mol | gram per mole |

| | |
|---|---|
| > | greater than |
| HDPE | high-density polyethylene |
| < | less than |
| LPZ | lower production zone |
| $m^{-1}$ | per meter |
| Ma | million years before present |
| MCL | maximum contaminant level |
| meq | milliequivalents |
| m/m | meter per meter |
| microgram per liter | µg/L |
| milliequivalents per liter | meq/L |
| milligram per liter | mg/L |
| millimole per liter | mmol/L |
| mol/L | mole per liter |
| $O_2$ | dissolved oxygen |
| pH | negative logarithm of hydrogen-ion activity |
| PHREEQC | pH-REdox-EQuilibrium equations in the C programming language |
| PVC | polyvinyl chloride |
| SI | saturation index |
| SL/BM | Santee Limestone/Black Mingo |
| SSE | sum of squares error |
| TDS | total dissolved solids |
| THM | trihalomethanes |
| UPZ | upper production zone |
| USEPA | U.S. Environmental Protection Agency |
| USGS | U.S. Geological Survey |

# Hydrologic and Geochemical Evaluation of Aquifer Storage Recovery in the Santee Limestone/Black Mingo Aquifer, Charleston, South Carolina, 1998–2002

By Matthew D. Petkewich[1], David L. Parkhurst[2], Kevin J. Conlon[1], Bruce G. Campbell[1], and June E. Mirecki[3]

## Abstract

The hydrologic and geochemical effects of aquifer storage recovery were evaluated to determine the potential for supplying the city of Charleston, South Carolina, with large quantities of potable water during emergencies, such as earthquakes, hurricanes, or hard freezes. An aquifer storage recovery system, including a production well and three observation wells, was installed at a site located on the Charleston peninsula. The focus of this study was the 23.2-meter thick Tertiary-age carbonate and sand aquifer of the Santee Limestone and the Black Mingo Group, the northernmost equivalent of the Floridan aquifer system.

Four cycles of injection, storage, and recovery were conducted between October 1999 and February 2002. Each cycle consisted of injecting between 6.90 and 7.19 million liters of water for storage periods of 1, 3, or 6 months. The volume of recovered water that did not exceed the U.S. Environmental Protection Agency secondary standard for chloride (250 milligrams per liter) varied from 1.48 to 2.46 million liters, which is equivalent to 21 and 34 percent of the total volume injected for the individual tests. Aquifer storage recovery testing occurred within two productive zones of the brackish Santee Limestone/Black Mingo aquifer. The individual productive zones were determined to be approximately 2 to 4 meters thick, based on borehole geophysical logs, electromagnetic flow-meter testing, and specific-conductance profiles collected within the observation wells. A transmissivity and storage coefficient of 37 meters squared per day and $3 \times 10^{-5}$, respectively, were determined for the Santee Limestone/Black Mingo aquifer.

Water-quality and sediment samples collected during this investigation documented baseline aquifer and injected water quality, aquifer matrix composition, and changes in injected/aquifer water quality during injection, storage, and recovery. A total of 193 water-quality samples were collected and analyzed for physical properties, major and minor ions, and nutrients. The aquifer and treated surface water were sodium-chloride and calcium/sodium-bicarbonate water types, respectively. Forty-five samples were collected and analyzed for total trihalomethane. Total trihalomethane data collected during aquifer storage recovery cycle 4 indicated that this constituent would not restrict the use of recovered water for drinking-water purposes. Analysis of six sediment samples collected from a cored well located near the aquifer storage recovery site showed that quartz and calcite were the dominant minerals in the Santee Limestone/Black Mingo aquifer. Estimated cation exchange capacity ranged from 12 to 36 milliequivalents per 100 grams in the lower section of the aquifer.

A reactive transport model was developed that included two 2-meter thick layers to describe each of the production zones. The four layers composing the production zones were assigned porosities ranging from 0.1 to 0.3 and hydraulic conductivities ranging from 1 to 8.4 meters per day. Specific storage of the aquifer and confining units was estimated to be $1.5 \times 10^{-5}$ meter$^{-1}$. Longitudinal dispersivity of all layers was specified to be 0.5 meter. Leakage through the confining unit was estimated to be minimal and, therefore, not used in the reactive transport modeling.

Inverse geochemical modeling indicates that mixing, cation exchange, and calcite dissolution are the dominant reactions that occur during aquifer storage recovery testing in the Santee Limestone/Black Mingo aquifer. Potable water injected into the Santee Limestone/Black Mingo aquifer evolved chemically by mixing with brackish background water and reaction with calcite and cation exchangers in the sediment. Reactive-transport model simulations indicated that the calcite and exchange reactions could be treated as equilibrium processes.

Simulations with the calibrated reactive transport model indicated that approximately one-fourth of the total volume of water injected into the aquifer can be recovered as potable

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water, regardless of the length of the injection period. Simulations also indicated that calcite dissolves near the injection well and precipitates where freshwater and brackish water mix during injection. However, the amounts of dissolved and precipitated calcite have a negligible effect on the aquifer porosity for simulated injection periods as long as 100 years and do not affect the flow of water during aquifer storage recovery. Finally, simulations indicated that the regional configuration of the potentiometric surface in the aquifer has a negligible effect on the storage of water.

# Introduction

Charleston, South Carolina, is located at the confluence of the Ashley, Cooper, and Wando Rivers near the Atlantic Ocean in the lower Atlantic Coastal Plain physiographic province (fig. 1). The area is characterized by wide estuaries bordered by extensive salt marshes, which are typical of coastal topography of low relief. The Charleston area is underlain by approximately 760 meters (m) of Quaternary, Tertiary, and Upper Cretaceous sediments that compose five distinct aquifers separated by five confining units (Park, 1985; Campbell and others, 1997).

In 2000, the Charleston Commissioners of Public Works (CCPW ) provided retail and wholesale water to the Charleston metropolitan area, which includes portions of Charleston, Berkeley, and Dorchester Counties, and serves a population of about 420,000. CCPW provided water with an average flow of 189 million liters per day (ML/d) and a peak flow of 340 ML/d (J.B. Cook, Charleston Commissioners of Public Works, written commun., 2003). The primary sources of potable water were treated surface water from the Bushy Park Reservoir and Edisto River (fig. 1). Although the CCPW currently (2004) has a treatment capacity that far exceeds normal demand, there is concern that demand may exceed delivery capacity in the event of damage to the water-distribution system. Hurricanes, earthquakes, and hard freezes potentially can result in such damage. Charleston is vulnerable to hurricanes and coastal flooding, as demonstrated in 1989 during Hurricane Hugo (Purvis, 1989). In 1886, the city was heavily damaged by the largest earthquake (magnitude 7.3) to strike the eastern United States in recorded history (Bollinger, 1977). Occasional hard freezes, such as one that occurred in December 1989, also can cause major disruptions in water-distribution service. One of the consequences of such disasters for the city of Charleston is the loss of potable water-transmission capacity, especially in the historic peninsula section of the city.

Approximately 2,400 kilometers (km) of water mains distribute treated surface water from the water-treatment plant in Hanahan, S.C. (fig. 1B) to retail and wholesale customers throughout CCPW’s water-service area. These water mains are constructed of cast iron, ductile iron, and a small portion consists of polyvinyl chloride (PVC) and high-density polyethylene (HDPE). Some of these water mains are located in the historic Charleston peninsula and consist of original cast iron installed in the early 1900's as well as newer ductile iron (J.B. Cook, Charleston Commissioners of Public Works, written commun., 2003).

As part of the CCPW's overall asset-management program, water mains are replaced on a systematic basis using determining criteria such as breakage rate, location, and age. While most mains in the peninsula area are in structurally good condition, these are the oldest water mains in the distribution system, and the breakage rate is highest in the peninsula area. Therefore, it is reasonable to assume that in the event of extraordinary circumstances, such as hurricanes, hard freezes, and earthquakes, some of these older water mains will break. Water main breaks located between the treatment plant and downtown Charleston, or extensive water main breaks in the downtown area, can substantially reduce the water flow rates to the peninsula. This is known as a result of both historical experience and CCPW’s water system hydraulic modeling (J.B. Cook, Charleston Commissioners of Public Works, written commun., 2003).

In the event of an earthquake of similar magnitude (7.3) as the 1886 Charleston earthquake, it is estimated that 80 percent of the Charleston area homes would be without water for weeks to months because of damaged water pipes (South Carolina Emergency Preparedness Division, 2001). The loss of water-transmission capacity would not only eliminate potable water for domestic purposes, but also would limit fire-fighting capabilities in the areas affected by the ruptured mains. The South Carolina Emergency Preparedness Division (2001) predicts the possibility of more than 250 fires following a 7.3-magnitude earthquake in the Charleston, S.C., area.

The CCPW, in cooperation with the U.S. Geological Survey (USGS), evaluated the hydrologic and geochemical effects of an aquifer storage recovery (ASR) system on the Charleston peninsula from 1998 to 2002. Aquifer storage recovery is the concept of storing injected water in an aquifer for later recovery. The investigation evaluated the effects of injecting freshwater in a brackish aquifer, a process that can be cost effective in areas where surface-water reservoirs or areas for above-ground storage tanks are limited. A typical ASR system consists of at least one production well that is open or screened in the aquifer of interest. The production well is equipped with an injection line to transport water from land surface to the aquifer through the screens or open-hole portion of the well and a pump to transport the water from the aquifer back to land surface. Screened or open-hole observation wells are located near the production well to allow the monitoring of the spatial distribution of injected water and collection of ground-water samples.

## Purpose and Scope

This report describes the hydrogeology and geochemistry of the Tertiary Santee Limestone/Black Mingo (SL/BM) confined aquifer (fig. 2), the northernmost equivalent of the Floridan aquifer system (Park, 1985), and the effect of ASR

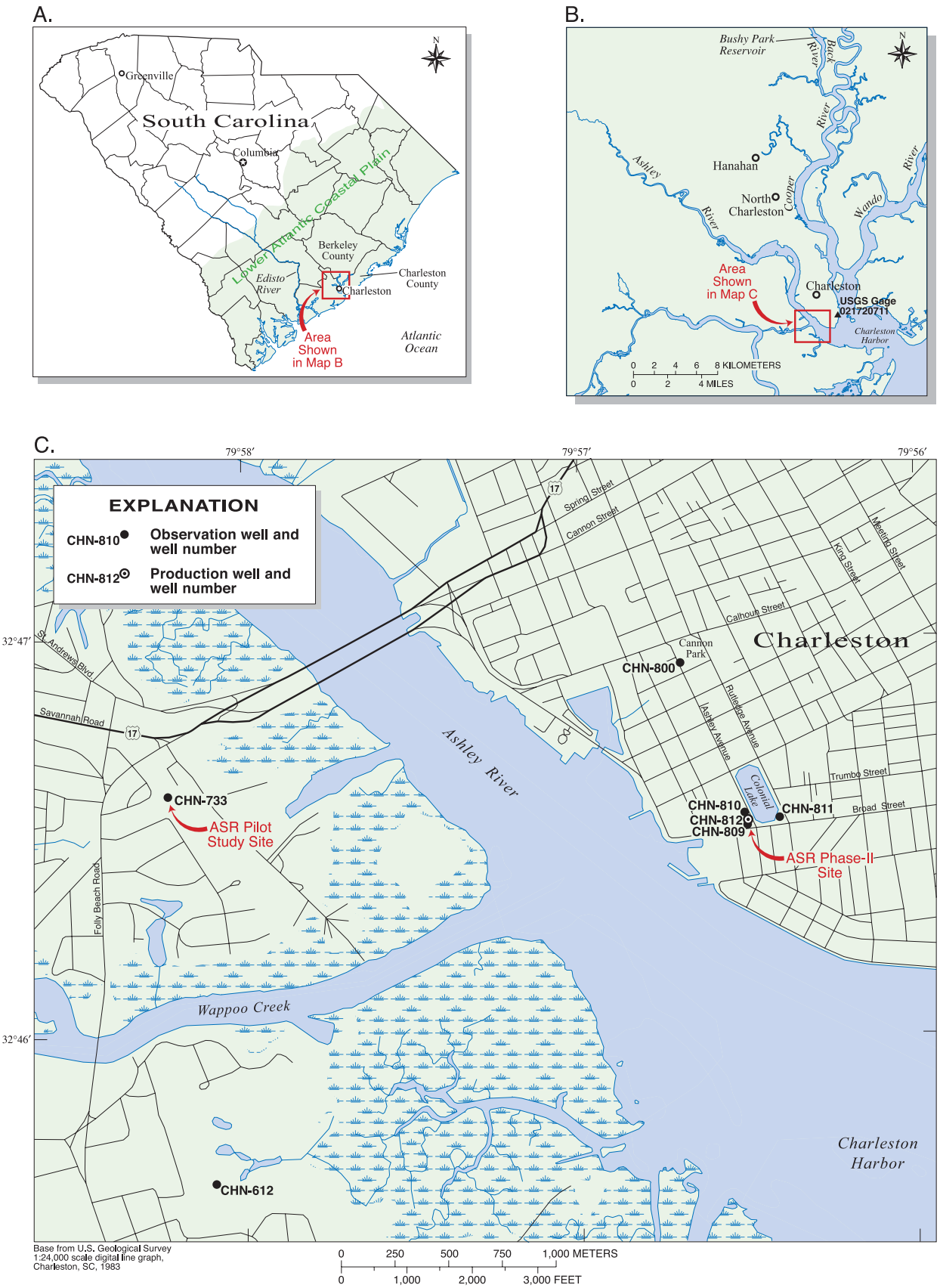


**Figure 1.** Study area and location of the aquifer storage recovery pilot study site and phase-II site, Charleston, South Carolina.

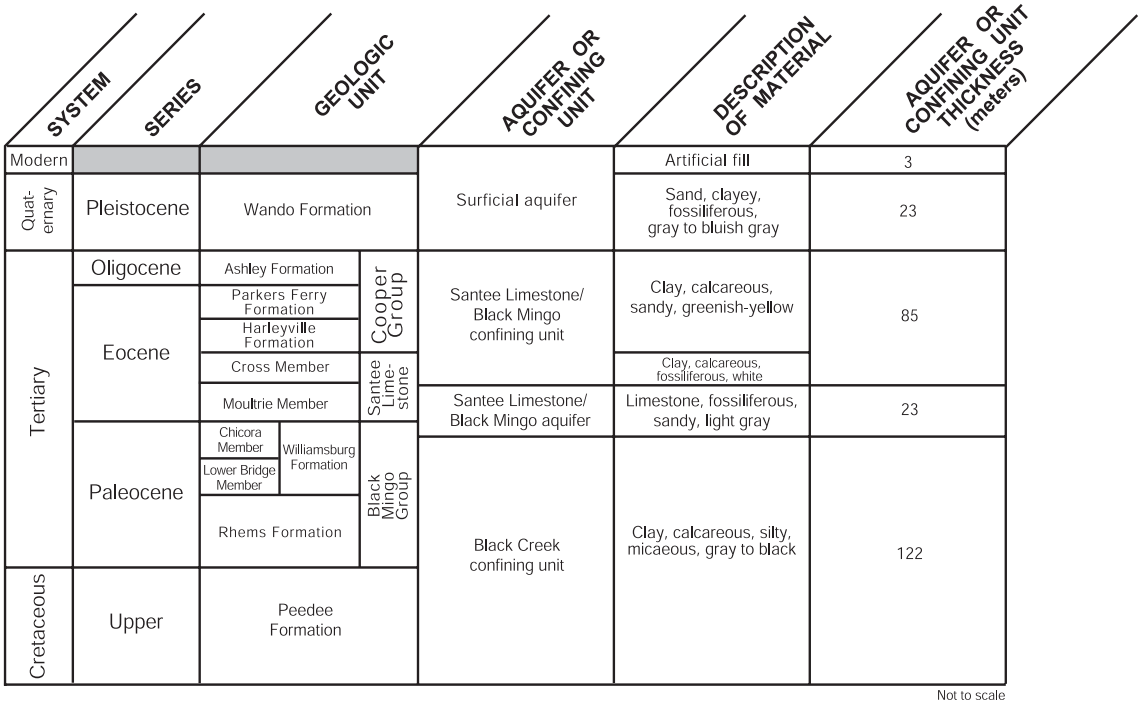


**Figure 2.** Generalized stratigraphic and hydrogeologic column based on core collected from observation well CHN-800 in Charleston, South Carolina.

testing on aquifer hydraulic properties and recovered water quality. The report includes an evaluation of data collected to determine if ASR technology is a technically sound method for supplying downtown Charleston, S.C., with large quantities of potable water during emergencies.

The scope of this report includes discussion of results obtained from geophysical logging, aquifer testing, water-quality sampling, continuous monitoring of water-level and water-quality data, and numerical modeling during four ASR tests completed in Charleston, S.C. Results from geophysical logging and aquifer testing were used to enhance the knowledge of the hydrogeology of the SL/BM aquifer and to evaluate the transport of injected treated surface water in the aquifer. Continuous water-level and water-quality data collected from two observation wells during the ASR tests were used to evaluate the trends in injection/withdrawal rates, recovery efficiency, and long-term storage effects on the injected water quality. Finally, geochemical modeling was used to determine the dominant chemical reactions and hydraulic processes that affect the injected water quality during the ASR tests.

This investigation addressed issues that are part of the USGS mission. The results of this study are applicable to other ASR investigations in similar hydrogeologic environments and investigations of the Floridan aquifer system. The study advanced the knowledge of ASR in brackish-water aquifers and the hydrologic processes that occur in carbonate/sand aquifer systems. Recently developed numerical modeling techniques were used in this investigation to study solute transport and chemical reactions in the ground-water system. Results of this investigation can be used by local and State water managers to address issues of increased demand for water resources in coastal areas of increasing population.

## Previous Investigation

A pilot study was conducted during 1993–95 in which the USGS tested the feasibility of ASR technology in storing potable water at a pilot site (fig. 1) in Charleston, west of the Ashley River (Campbell and others, 1997; Mirecki and others, 1998). During the pilot investigation, nine successive ASR cycles (each cycle consisting of injection, storage, and recovery) were conducted to evaluate hydrologic and water-quality changes resulting from injection of treated surface water into the SL/BM aquifer.

Results of the pilot study indicated that ASR implementation in the SL/BM aquifer in the Charleston area is feasible, with recovery of potable water (recovery efficiency) ranging between 38 and 61 percent of the total volume injected (Campbell and others, 1997). Recovery efficiency is the percentage of stored water that meets a target water-quality criterion after retrieval from the aquifer (Pyne, 1995). During

the pilot investigation, the period of storage typically was short, with durations ranging from 8 hours to less than 6 days, and the recovery of injected/stored water was discontinued prior to complete retrieval of all the injected water. The scope of the pilot investigation did not include investigating water-quality changes during long-term storage or completely characterizing the hydraulic properties of the SL/BM aquifer. Although the pilot ASR site was in Charleston, it was not located on the Charleston peninsula, which is the area that could be isolated from water supply during emergency situations. Lithologic comparison of core collected at the ASR pilot site (CHN-733) to core collected on the Charleston peninsula (CHN-800; fig. 1) indicated that the top of the SL/BM aquifer was about 7.3 m higher on the Charleston peninsula than at the ASR pilot site. Additionally, the aquifer portion of core collected from the pilot site consisted of more carbonate and less clastic material than the aquifer portion of core collected from the peninsula. Because further investigation was needed to address these issues, an ASR system (phase II) was installed in the downtown Charleston area (fig. 1).

## Description of Study Area

The phase-II ASR site, located immediately southwest of Colonial Lake in downtown Charleston, S.C. (fig. 1), was selected for this study because of access to potable water mains and the storm sewer, and also because it was possible to locate three new observation wells at selected distances and directions surrounding a newly installed production well. The production well was constructed to allow the injection of treated surface water from the city water mains. Recovered water was discharged to the storm sewer.

## Acknowledgments

The support received from John Cook and Randy Hoagland of the Charleston Commissioners of Public Works during all phases of this investigation has been invaluable. The Charleston Department of Parks and Recreation provided land for the study site. The interest and support from the Charleston Department of Parks and Recreation and from Fire Chief Russell Thomas are gratefully acknowledged. The analysis of numerous water-quality samples by Dixie Fanning and the Commissioners of Public Works laboratory is greatly appreciated. Keith Halford and Francis Riley of the U.S. Geological Survey assisted in aquifer-test planning and analysis. The interest and positive feedback from the residents of the neighborhoods surrounding Colonial Lake also are appreciated.

# Hydrogeologic Framework

The phase-II ASR site in the lower Atlantic Coastal Plain physiographic province is underlain by Quaternary, Tertiary, and Upper Cretaceous sediments that regionally have a total combined thickness of about 760 m (Campbell and others, 1997). These depositional units are composed of terrigenous and carbonate sediments that unconformably overlie Precambrian and Paleozoic basalt (Gohn and others, 1977). The subject of the ASR study was the SL/BM aquifer within Tertiary Coastal Plain sediments. A generalized description of part of the Upper Cretaceous and all of the Tertiary and Quaternary stratigraphy and lithology at the ASR site (fig. 2) was based on a continuous core obtained during the drilling of observation well CHN-800 in Cannon Park in downtown Charleston (fig. 1). The Tertiary section beneath the ASR site was subdivided on the basis of lithology, paleontology, and geophysical logs (Bybell and others, 1998).

## Tertiary Stratigraphy

The Black Mingo Group of Tertiary age is composed of two upper Paleocene formations—the Williamsburg Formation and the underlying Rhems Formation (Sloan, 1908; fig. 2). These sediments were deposited in inner shelf and marginal-marine depositional environments about 55 million years before present (Ma). The dominant lithology of the Black Mingo Group consists of interbedded sequences of greenish-gray mudstones and dark-gray to black laminated clays (Bybell and others, 1998). The total thickness of the Black Mingo Group in the area penetrated at the Cannon Park core site is approximately 117 m. The Rhems Formation is approximately 75 m thick and lies between unconformable contacts at 227 and 152 m below land surface (bls) in the Cannon Park core. The Rhems section consists of bioturbated, moderately calcareous silty clays, clayey silts, and muddy, very fine sands. The Lower Bridge Member of the Williamsburg Formation is 21 m thick and lies between unconformable contacts at 152 and 131 m bls. The Lower Bridge is a homogeneous section of bioturbated, moderately calcareous, clayey quartz silts and muddy, very fine quartz sands. The Chicora Member of the Williamsburg Formation is 21.3 m thick at the Cannon Park site and extends from 131 to 109 m bls. The Chicora Member consists of interbedded sequences of gray, bioturbated, muddy limestone; carbonate- and silica-cemented sandstones; moldic, quartz-bearing pelecypod limestone; macrofossiliferous quartz sands; and white to pale-gray argillaceous sands and silts.

The middle Eocene Moultrie Member of the Santee Limestone unconformably overlies the Chicora Member of the Williamsburg Formation of the Black Mingo Group. The Moultrie Member, only 2.1 m thick at the Cannon Park site, is present from 109 to 107 m bls. The Moultrie Member consists primarily of a light-gray, quartz-rich, moldic, bryozoan-pelecypod biosparrudite. The top of the Moultrie Member is extensively bioturbated with quartz-, phosphate-, and

glauconite-filled burrows. The upper surface of the Moultrie Member is partially coated with a green phosphatic crust that marks the contact with the overlying late middle to upper Eocene Cross Member of the Santee Limestone. The Moultrie Member was deposited in a shallow, open marine-shelf environment about 45 Ma (Bybell and others, 1998).

The middle to upper Eocene Cross Member of the Santee Limestone is a white, dense, partially silicified calcilutite containing abundant foraminifers, echinoid spines, and ostracods (Fronabarger and others, 1995). The Cross Member is 11.9 m thick and is present from 107 to 95.4 m bls at the Cannon Park site. The sediments were deposited in an outer continental-shelf environment about 41 Ma.

The Cross Member of the Santee Limestone is unconformably overlain by the Cooper Group, which consists of the upper Eocene Harleyville and Parkers Ferry Formations, and the upper Oligocene Ashley Formation (Ward and others, 1979; Weems and Lemon, 1984). The Harleyville Formation is a compact, phosphatic, light-gray calcilutite containing abundant foraminifers (Bybell and others, 1998). The Harleyville Formation is 3.66 m thick and present between unconformable contacts at 95.4 and 91.7 m in the Cannon Park core. Both contacts are defined by extensively bioturbated phosphate- and glauconite-filled burrows. The overlying Parkers Ferry Formation is a dense, pale-yellow to light-gray calcilutite containing abundant echinoid spines and sand-size foraminifers. The Parkers Ferry Formation is 28.3 m thick and lies between 91.7 and 63.4 m bls in the Cannon Park core. The Harleyville and Parkers Ferry Formations were deposited in an outer-continental-shelf environment about 38 Ma. The upper Oligocene Ashley Formation of the Cooper Group unconformably overlies the Parkers Ferry Formation. The Ashley Formation is present from 63.4 to 22.6 m in the Cannon Park core. This 40.8 m thick section consists of a massive to bioturbated, pale-olive, fine-grained, quartz-rich, glauconite and phosphatic calcarenite containing abundant sand-size foraminifers (Fronabarger and others, 1995). The Ashley Formation was deposited in outer continental-shelf to marginal marine environments about 30 Ma. The Miocene Marks Head Formation is absent in the Cannon Park core.

## Quaternary Stratigraphy

The upper Pleistocene Wando Formation unconformably overlies the upper Oligocene Ashley Formation. The Wando Formation, approximately 20 m thick at the Cannon Park site, is present from 22.6 to 2.74 m bls. The Wando Formation consists of quartz sand to shell-rich, clayey sand; organic-rich clays; and fine-grained, fossiliferous sand overlying a phosphate-pebble lag deposit (Fronabarger and others, 1995). The Wando Formation has a complex depositional history related to sea-level changes during the late Quaternary and is about 130,000 years old (McCartan and others, 1980; Wehmiller and Belknap, 1982). The sediment overlying the Wando Formation is 2.74 m thick and predominantly fill material.

## X-ray Diffraction Analysis

Six sediment samples from the Cannon Park (CHN-800) core were analyzed to determine the dominant mineralogy in the SL/BM aquifer. Samples were analyzed at the USGS laboratory in Boulder, Colorado, using quantitative, whole rock, X-ray diffraction analysis methods described in Srodon and others (2001). The samples were collected from high and low permeable sections of the SL/BM aquifer at depths that ranged from 108 to 130 m bls. Four samples representing the high permeable zones of the aquifer were collected from sections of the aquifer referred to as the upper (UPZ) and lower (LPZ) production zones (fig. 3). Two UPZ samples were collected at depths of 109 and 112 m bls, and two LPZ samples were collected at depths of 126 and 129 m bls. Two samples were collected from less-permeable material between the UPZ and the LPZ at depths of 117 and 122 m bls. Results of the X-ray diffraction analysis are listed in table 1.

Calcite and quartz are the dominant minerals identified in the six sediment samples. Pyrite, an iron sulfide, is present in all the zones sampled but represents less than 3 percent by weight in all samples. Ankerite, a calcium carbonate and member of the dolomite group, contains iron, magnesium, and manganese (Ford, 1949). Ankerite is present only in sample CP-384 and represents 9 percent of the sample by weight (table 1). Clinoptilolite, a high silica zeolite, is an alterated by-product of volcanic ash (Heron, 1969). Clinoptilolite is present in the LPZ and less-permeable material and ranges from 2 to 13 percent by sample weight. Opal, a hydrated silica, is derived from both volcanic ash and from the decomposition of clinoptilolite under near-surface weathering conditions. Heron (1969) described several investigations that report the presence of zeolites and(or) opal in the lower Tertiary System in the southeastern States of Mississippi, Alabama, Georgia, South Carolina, North Carolina, and in core holes off the eastern coast of Florida. Opal is not present in the UPZ samples (CP-358 and CP-369) but represents between 0 and 23 percent (by weight) of the samples collected in the less permeable zone (CP-384 and CP-401) and LPZ (CP-412 and CP-423) (table 1).

The clay minerals of the aquifer consist of ferruginous illite-smectite and aluminous illite-smectite. Smectite is a group of clay minerals with large surface areas and high cation-exchange capacities (Pyne, 1995). Smectite-illite clay minerals form through the alteration of silicate minerals, such as feldspars, micas, and volcanic ash (Schlumberger Limited, 2003). Clays are not present in the samples collected from the UPZ (CP-358 and CP-369) but are present in the LPZ and lower permeable material sampled and range from 9 to 12 percent by sample weight (table 1).

Cation exchange capacity (CEC), expressed in milliequivalents per 100 grams (meq/100 g), is an indicator of the potential for ion exchange to occur in certain minerals and clays (Pyne, 1995). The estimated CEC of the sediment samples from the core are based on estimated CEC for clinoptilolite of 180 and for illite-smectite of 100 meq/100 g. Estimated CEC

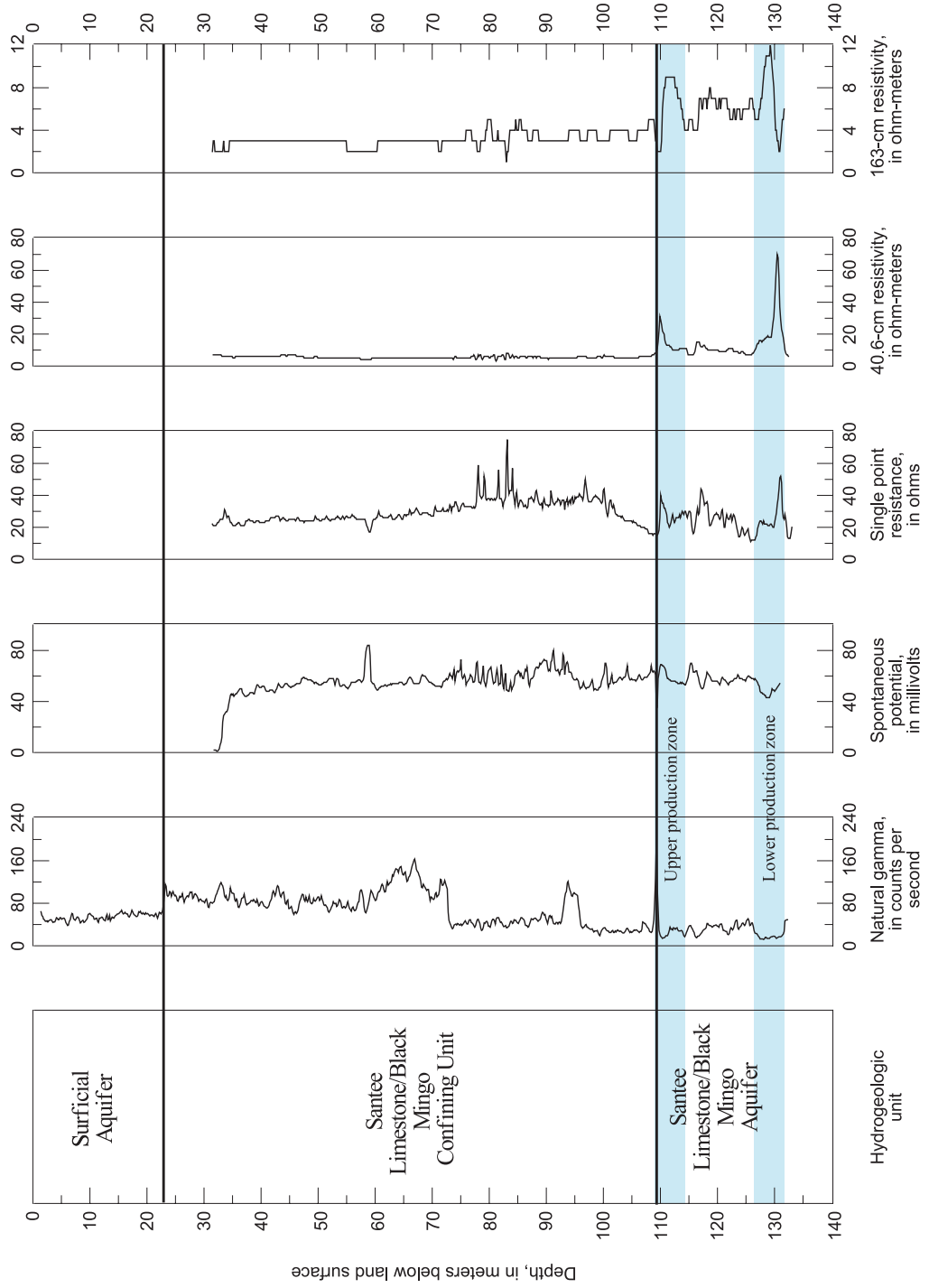


**Figure 3.** Borehole geophysical logs for observation well CHN-811 at the aquifer storage recovery site, Charleston, South Carolina.

**Table 1.** Mineral abundances measured by quantitative X-ray diffraction microscopy and cation exchange capacity.

[UPZ, upper production zone; LP, less permeable zone; LPZ, lower production zone; CEC, cation exchange capacity; meq/100 g, milliequivalents per 100 grams]

| Mineral | Sample identification (zone) | | | | | |
|---|---|---|---|---|---|---|
| | CP-358 (UPZ) | CP-369 (UPZ) | CP-384 (LP) | CP-401 (LP) | CP-412 (LPZ) | CP-423 (LPZ) |
| | Weight percent | | | | | |
| **Non-clays** | | | | | | |
| Quartz | 16 | 49 | 37 | 56 | 24 | 19 |
| Microcline feldspar | 2 | 2 | 1 | 1 | 1 | 0 |
| Plagioclase feldspar | 2 | 1 | 0 | 1 | 1 | 0 |
| Calcite | 77 | 48 | 32 | 19 | 23 | 66 |
| Ankerite | 0 | 0 | 9 | 0 | 0 | 0 |
| Pyrite | 0 | 2 | 1 | 1 | 1 | 0 |
| Opal | 0 | 0 | 4 | 10 | 23 | 0 |
| Clinoptilolite | 0 | 0 | 7 | 7 | 13 | 2 |
| Total non-clays[a] | 97 | 102 | 91 | 95 | 86 | 87 |
| **Clays** | | | | | | |
| Ferruginous illite plus smectite | 0 | 0 | 9 | 6 | 9 | 9 |
| Aluminous illite plus smectite | 0 | 0 | 0 | 3 | 3 | 0 |
| Total clays | 0 | 0 | 9 | 9 | 12 | 9 |
| Total[a] non-clays and clays | 97 | 102 | 100 | 104 | 9 | 96 |
| Estimated CEC[b] (meq/100 g) | 0 | 0 | 22 | 22 | 36 | 12 |

[a]Total weight percent may not equal 100 percent because of rounding.

[b]Based on estimated CEC for clinoptilolite of 180 and for illite plus smectite of 100 meq/100 g.

ranges from 12 to 36 meq/100 g (table 1) in the lower section of the aquifer (CP-384, CP-401, CP-412, and CP-423).

## Hydrogeology

The South Carolina Coastal Plain strata can be divided into a series of aquifers and confining units based on their relative permeabilities. Aucott and Speiran (1985a) described five major Coastal Plain aquifers in the Charleston area. From youngest to oldest, these aquifers are the surficial, Floridan, Black Creek, Middendorf, and Cape Fear aquifers. The surficial aquifer is composed of Quaternary unconsolidated sands of the Wando Formation (fig. 2) and 2.74 m of overlying sediment/fill material. In Charleston, the Floridan aquifer system is composed of the Tertiary limestones and sands of the Moultrie Member of the Santee Limestone and the Williamsburg Formation of the Black Mingo Group, respectively, and is referred to as the SL/BM aquifer in Park (1985) and this report. The confined Black Creek, Middendorf, and Cape Fear aquifers are composed of unconsolidated sands within the respective Upper Cretaceous formations (Campbell and Gohn, 1994).

The focus of this study is the Tertiary limestone and sand aquifer of the Santee Limestone and the Black Mingo Group (fig. 2). Park (1985) and Meadows (1987) indicated that the two geologic units (the Santee Limestone and the upper 30.5 m of the Black Mingo Group) respond hydraulically as a single hydrogeologic unit. The combined units are characterized by a significant degree of hydraulic connection and little difference in potentiometric levels. The permeable zones of the Santee Limestone and Black Mingo Group are approximately 23 m thick at the ASR site. The SL/BM aquifer is confined above by the Cooper Group and the Cross Member of the Santee Limestone (fig. 2). Transmissivity of the SL/BM aquifer varies regionally from about 12 to 344 meters squared per day ($m^2/d$; Park, 1985; Aucott and Newcome, 1986; Newcome, 1993). A storage coefficient of $1.0 \times 10^{-4}$ was reported for this aquifer in Berkeley County (Newcome, 1993).

Predevelopment flow (prior to 1960) in the SL/BM aquifer was from northwest to southeast, generally perpendicular to the

coastline (fig. 4). Predevelopment water-level altitudes in the SL/BM aquifer in the Charleston area were approximately 7.6 m above NGVD 29. Ground-water recharge entered the aquifer at its outcrop area near Orangeburg and Lake Marion and flowed toward the southeast (fig. 4). Large-scale development of the aquifer began during the 1960's, especially in the area approximately 30 km northwest of Charleston. Water-level measurements collected in 1982 indicate a cone of depression in the SL/BM aquifer potentiometric surface (Aucott and Speiran, 1985b). By the early 1990's, extensive development combined with poor aquifer hydraulic characteristics resulted in large depressions in the potentiometric surface, and the lowest water-level altitudes (approximately -19.8 m NGVD 29) occurred in southern Berkeley County (Campbell and others, 1997). In 1998, the regional ground-water flow direction in the SL/BM aquifer was the reverse of the predevelopment flow direction in the Charleston area (fig. 5) and was toward the cones of depression (Hockensmith, 2001).

# Hydrogeologic Methods

This investigation included installation of a production well and 3 observation wells, geophysical logging of wells, aquifer testing, monitoring long-term water levels and water-quality characteristics, and ASR testing. The methods used allowed characterization of the hydraulic properties of the SL/BM aquifer, evaluation of the movement of the injected water during ASR testing, and management of the ASR tests.

## Well Construction and Instrumentation

A production well (CHN-812) and three observation wells (CHN-809, CHN-810, and CHN-811; fig. 1) were drilled at the phase-II ASR site using hydraulic rotary drilling techniques. The locations of the observation wells were selected to allow relatively rapid breakthrough of injected water at the two nearest observation wells (CHN-809 and CHN-810) during ASR testing, and to create a spatial distribution that optimizes the ability to characterize hydraulic properties, such as aquifer anisotropy and confining-unit leakage. All wells installed during this investigation were screened because of unsuccessful attempts at drilling, developing, and maintaining open-hole well construction. Prior to screening, unconsolidated sands in the SL/BM aquifer would collapse into the open hole and block part of the LPZ. Screen locations (table 2) were based on interpretation of natural gamma geophysical logs collected at well CHN-811 and lithologic core collected from well CHN-800 (fig. 1). After installation, the wells were developed by air injection and pumping until discharged water was free of drilling fluid and aquifer material.

The production well was constructed of 40.6-centimeter (cm) galvanized steel casing to 100 m bls and 16.8-cm galvanized steel casing from 100 to 134 m bls (table 2). The well also contained a 10.2-cm galvanized injection pipe and 25-horsepower pump. Inline totalizing meters were installed on the injection and discharge pipes to measure the volume of water injected and recovered during each test. A 5.1-cm PVC pipe also was placed in the production well to facilitate water-level measurements.

Observation wells CHN-809 and CHN-810 were constructed of 10.2-cm PVC casing and screened at the same intervals as the production well (table 2). Observation wells CHN-811A and CHN-811B were 5.1-cm PVC wells installed in the same 15.2-cm borehole and screened in the SL/BM aquifer and overlying confining unit, respectively. Well CHN-811A was screened from 112 to 115 m and 130 to 134 m bls. Well CHN-811B was screened from 102 to 104 m bls. Screens of the two wells were separated by a bentonite seal located in the borehole between 108 and 109 m bls.

Wells CHN-809 and CHN-810 were equipped with pressure transducers, water-quality probes, and data loggers that recorded data at 15-minute intervals. Pressure transducers and data loggers were used to monitor water levels in the observation wells during all phases of testing. Water-quality probes and data loggers were used to monitor changes in pH, water temperature, specific conductance, and dissolved-oxygen concentrations in the observation wells at specific depths during ASR testing. Both wells contained two separate specific-conductance probes located at depths that corresponded to the two most productive sections of the SL/BM aquifer. Initially, the specific-conductance probes were located at 116 and 131 m bls. On June 6, 2000, the 116-m and 131-m probes were moved to 112 and 129 m bls, respectively, because specific-conductance profiles collected in the wells indicated that these depths were more permeable. Water-quality data were used to determine when to sample the observation wells during injection and also facilitated sample collection. The water-quality probes also were used to collect intermittent specific-conductance profiles during ASR testing.

## Geophysical Logging

Borehole geophysical logs were collected from all wells installed during this investigation and were used to help interpret site hydrogeology and select well-screen placement. Logs collected include natural gamma, spontaneous potential, single-point resistance, 40.6-cm short normal resistivity, 163-cm long normal resistivity, lateral resistivity, formation resistivity, temperature, heat-pulse flow meter, and electromagnetic flow meter (EMFM). The natural gamma logs were collected prior to installing well casings and were used to determine the placement of well screens. Fluid resistivity, temperature, and flow-meter logs were used to determine the locations of the predominant ground-water flow zones in the screened portion of the SL/BM aquifer.

Borehole geophysical logs collected from well CHN-811 are the most complete set of logs collected at the ASR site (fig. 3). Low natural gamma counts on this geophysical log

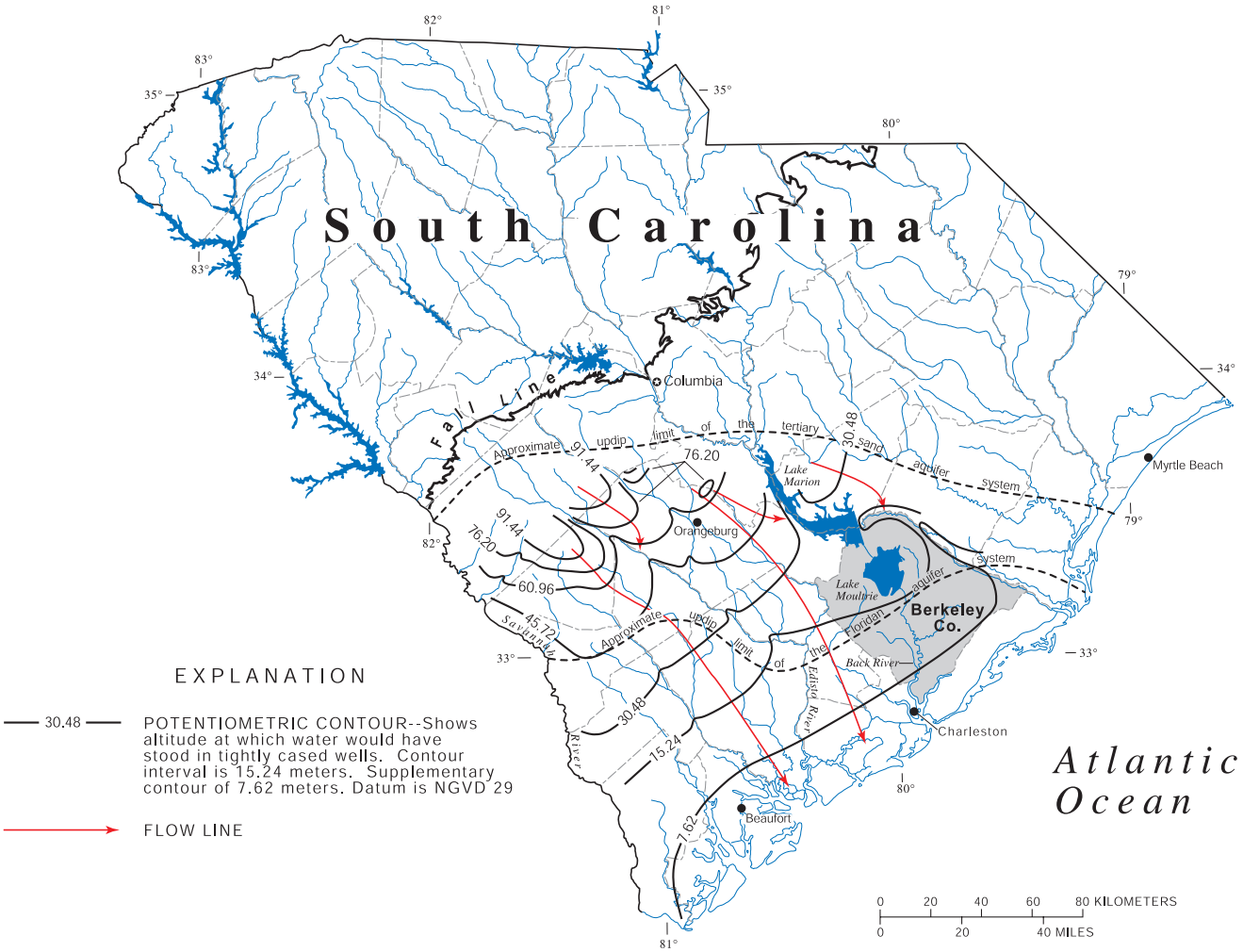


**Figure 4.** Approximate potentiometric surface of the Floridan (Santee Limestone/Black Mingo) aquifer system prior to development in South Carolina (modified from Aucott and Speiran, 1985a).

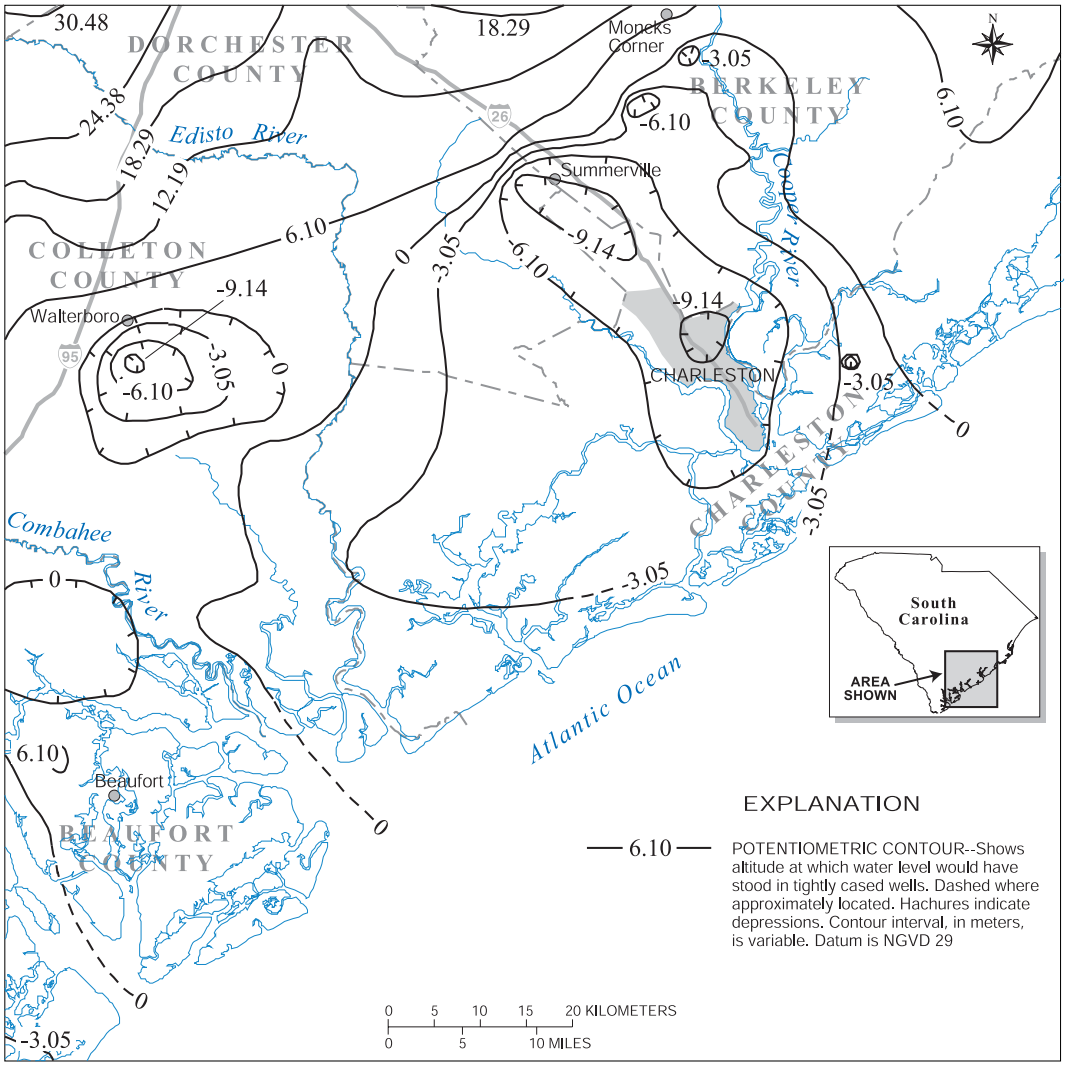


**Figure 5.** 1998 potentiometric surface of the Floridan (Santee Limestone/Black Mingo) aquifer system and Tertiary sand aquifer in South Carolina (modified from Hockensmith, 2001).

**Table 2.** Construction information for wells monitored during the aquifer storage recovery study in Charleston, South Carolina.

[m, meters; cm, centimeters; SS, stainless steel; diam., diameter; PVC, polyvinyl chloride; @, at; N/A, not applicable]

| Well identification (fig. 1) | Site identification | Latitude Longitude | Distance from production well CHN-812 (m) | Casing type | Borehole diameter (cm) | Grouted interval (meters below land surface) | Filter pack (meters below land surface) | Screened or open interval (meters below land surface) | Screen type |
|---|---|---|---|---|---|---|---|---|---|
| CHN-812 | 324618080560900 | 32° 46′ 32.42″<br>79° 56′ 30.30″ | 0 | 16.8-cm<br>304 SS Riser | 38.7 | 0 – 104 | 100 – 134 | 112 – 118<br>121 – 124<br>127 – 134 | 15.2-cm wire wrapped, 304 SS, 0.051-cm slot |
| CHN-809 | 324619080560900 | 32° 46′ 31.67″<br>79° 56′ 30.36″ | 23 | 10.2-cm<br>SCH 80 PVC | 14.9 | 0 – 32.6 | 109 – 134 | 112 – 118<br>121 – 124<br>127 – 134 | 10.2-cm diam. PVC, 0.025-cm slot |
| CHN-810 | 324620080561000 | 32° 46′ 33.52″<br>79° 56′ 30.91″ | 37 | 10.2-cm<br>SCH 80 PVC | 14.9 | 0 – 35.1 | 109 – 134 | 112 – 118<br>121 – 124<br>127 – 134 | 10.2-cm diam. PVC, 0.025-cm slot |
| CHN-811A | 324632079562400 | 32° 46′ 32.73″<br>79° 56′ 24.60″ | 148 | 5.1-cm SCH 40 PVC | 14.9 | 0 – 29.7<br>seal @ 109 m | 109 – 134 | 112 – 115<br>130 – 134 | 5.1-cm diam. PVC, 0.038-cm Vee Wire |
| CHN-811B | 324632079562401 | 32° 46′ 32.73″<br>79° 56′ 24.60″ | 148 | 5.1-cm SCH 40 PVC | 14.9 | 0 – 29.7<br>seal @ 101 m | 101 – 108 | 102 – 104 | 5.1-cm diam. PVC, 0.038-cm Vee Wire |
| CHN-800 | 324658079562500 | 32° 46′ 56.20″<br>79° 56′ 42.19″ | 795 | Open hole | 10.2 | 0 – 32.6 | N/A | 32.6 – 127<br>(Native material filled in from 127 to 168 m) | N/A |
| CHN-733 | 324637079581400 | 32° 46′ 36.55″<br>79° 58′ 14.08″ | 2,704 | Open hole | 10.2 | 0 – 15.2 | N/A | 15.2 – 161 | N/A |
| CHN-612 | 324541079580600 | 32° 45′ 38″<br>79° 58′ 06″ | 3,002 | Open hole | 15.2 | 0 – 40.2 | N/A | 40.2 – 174 | N/A |

represent coarse sand or limestone, as determined from lithologic data compiled from a core collected at well CHN-800. Well screens were placed in each borehole within the sand and limestone sections of the aquifer.

Formation resistivity logs were collected in wells CHN-809 and CHN-810 during the injection phase of ASR cycle 2 (May 8–June 16, 2000); logs were collected on May 19, June 1, and June 5 (fig. 6). Comparison of the logs indicates that two

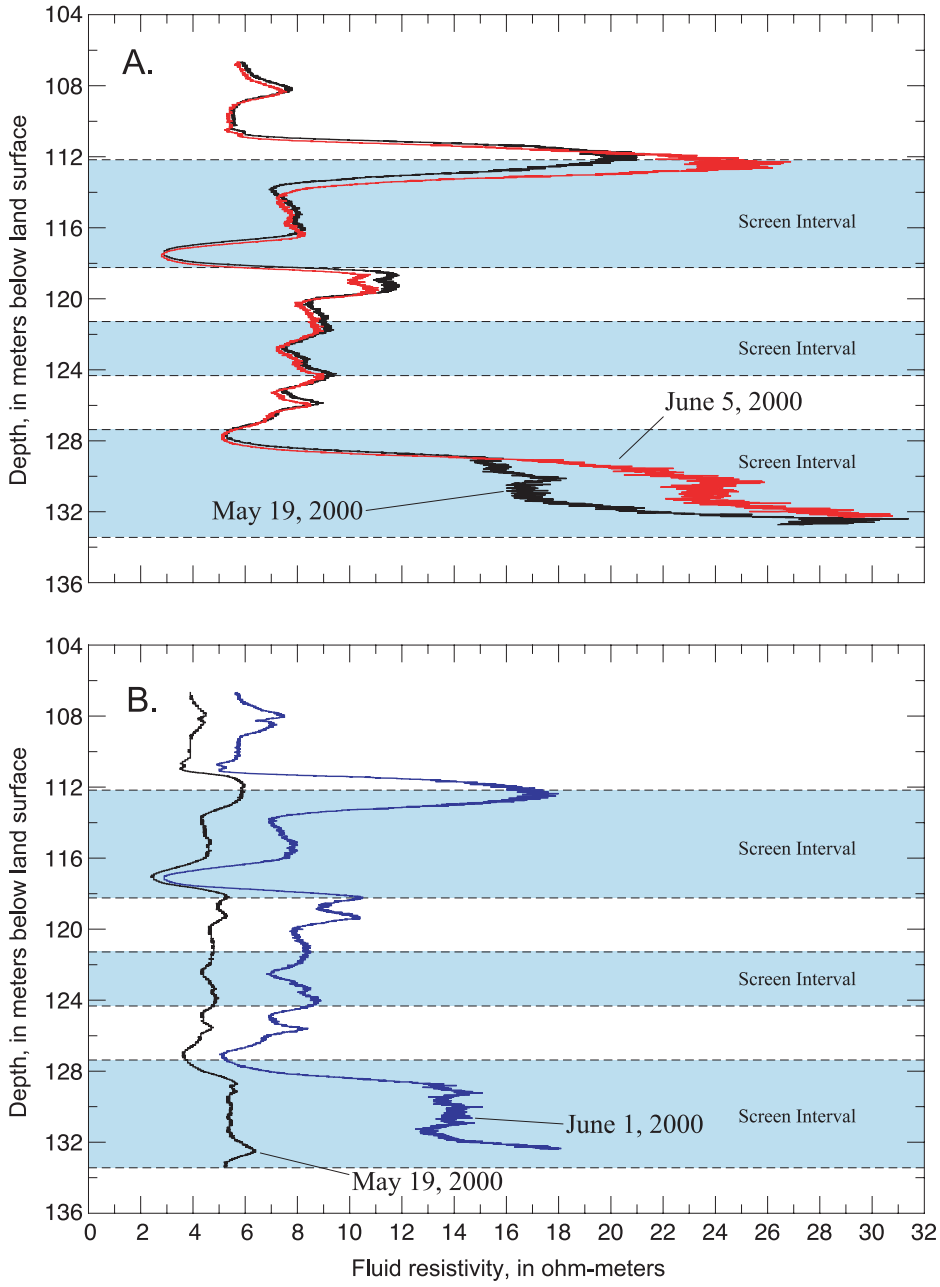


**Figure 6.** Formation resistivity profiles collected in observation wells (A) CHN-809 and (B) CHN-810 during the injection phase of cycle 2 of the aquifer storage recovery study in Charleston, South Carolina.

zones of high resistivity develop as low specific-conductance-injected water flowed through the SL/BM aquifer. The top zone is centered on about 112 m bls and is approximately 2 m thick; this zone includes a portion of the aquifer that is not screened. The lower zone is centered on about 130 m bls. Because the sensor of the resistivity probe was near the top of the geophysical tool, the full section of the lower screen could not be logged. Therefore, the total thickness of the lower zone (as defined by the higher resistivity measurements) could not be determined. Results of the resistivity profiles depict a lower zone thickness of at least 4 m.

Stationary borehole electromagnetic flow-meter data were collected to delineate changes in vertical flow rates in the screened sections of well CHN-809 (fig. 7), thereby providing a means for estimating the most productive portions of the SL/BM aquifer. Borehole EMFM results, measured in units of liters per second (L/s), represent the volume of water flowing through a hollow portion of the flow meter. The EMFM was equipped with a rubber skirt (diverter) that surrounded the EMFM and extended to the well casing. In the cased portion of the well bore, essentially all vertical flow at a given depth is directed through the flow meter. Within the screened portion of the well bore, however, vertical flow can occur through the gravel pack, resulting in an underestimation of flow at the given depth in the well bore.

Because ambient flow conditions in the SL/BM aquifer were similar to the quantifiable limits of the EMFM tool (0.006 L/s), the aquifer was stressed to induce flow within the well bore. Treated surface water was injected into the well bore at an average rate of 0.372 L/s. Borehole flow-meter flow rates were measured at various depths within and above the screened zones of the observation well. Measured flow rates were converted to percentages of average injection rate to delineate the areas of the screen where the vertical flow was lost to the aquifer. Dominant horizontal flow zones were assumed to be located where the greatest changes in the percentage of total flow occurred.

Borehole EMFM results indicate two major flow zones in the screened portion of well CHN-809 (fig. 7). An upper zone, located between 112 and 115 m bls, represented 49 percent of total flow into the well. A lower zone, located between 130 and 134 m, represented 40 percent of the total flow. The remaining

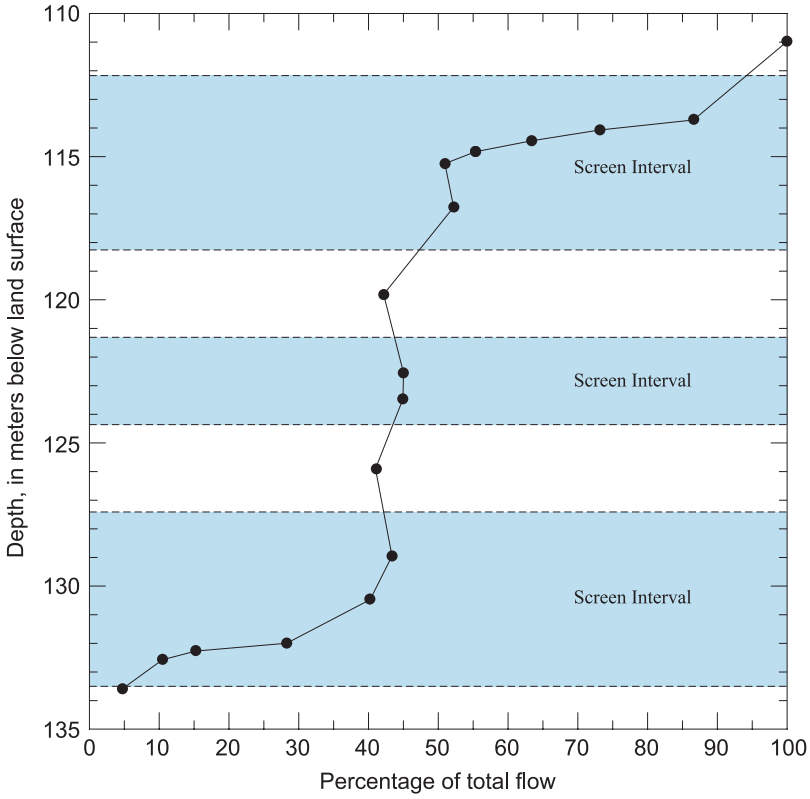


**Figure 7.** Electromagnetic flow-meter results while injecting freshwater into observation well CHN-809 during the aquifer storage recovery study in Charleston, South Carolina.

11 percent was received from the lower portion of the upper screen (about 9 percent), the middle screen (1 percent), and the upper portion of the lower screen (1 percent). These results indicate that the upper, middle, and lower screened sections represent total flow percentages of 58, 1, and 41, respectively.

## Determination of Aquifer Hydraulic Properties

Two aquifer tests were conducted to determine the hydraulic characteristics of the SL/BM aquifer and whether these characteristics change as a result of ASR testing. The initial aquifer test was conducted in June 1999 prior to the injection of any surface water. The second aquifer test was conducted in November 2001 during the recovery phase of the final ASR test.

The aquifer tests consisted of pumping production well CHN-812 at a constant discharge rate for several days while measuring water-level changes in wells CHN-612, CHN-733, CHN-800, CHN-809, CHN-810, CHN-811, and CHN-812 (fig. 1). All of these wells are screened in or open to the SL/BM aquifer. Water-level changes in the production and observation wells were measured with pressure transducers and data-logging equipment. Barometric pressure was measured by using the data-logging equipment, and tidal stage data were collected at a USGS streamgage less than 1.6 km from the ASR site on the Cooper River (station number 021720711 [fig. 1]), which allowed correction of aquifer water-level data for barometric or tidal effects, as necessary.

Because the SL/BM aquifer is potentially anisotropic and(or) affected by leakage from the adjacent confining units, the desired length of the pumping period was greater than 1 week. The plan for the aquifer test was to pump the production well for as long as reasonable. In both tests, the pump shut off prior to the planned shutdown of the aquifer tests. The durations of the June 1999 and November 2001 aquifer tests were 9.3 and 7.3 days, respectively (table 3). Average discharge rates of 10 and 7.78 L/s were determined by using an inline totalizing flow meter for the June 1999 and November 2001 aquifer tests, respectively (table 3).

The Moench (1985) method was selected to analyze the aquifer-test data because this method predicts water-level response to pumping a large-diameter well screened or open to a leaky confined aquifer. The method takes into account well-bore storage, skin effects, and the assumption that the confining unit is overlain or underlain by a constant-head boundary. The assumptions of the Moench (1985) method are reasonable for the Charleston ASR aquifer tests. Analytical results for the two aquifer tests are listed in table 3. Anisotropy was not required to match the water-level data measured during these two aquifer tests.

Analytical results using the Moench (1985) method were somewhat insensitive to the leakage factor. The leakage factor reported in table 3 is the maximum value allowed given the corresponding transmissivity and storage coefficient while maintaining the analytical method's approximate sum of squares error (SSE; change in SSE less than 10 percent). A larger leakage factor increased the SSE for the given set of data. A lower leakage factor maintained the originally calculated SSE. Analytical results using the maximum leakage factor produced predicted time-drawdown curves that became flat by the end of the tested time interval. However, the measured water levels declined throughout the duration of the aquifer test. The contrast between the measured data and the time-drawdown curves indicates that the leakage factor was substantially less than the maximum value. The SSE calculated from analytical results that did not include leakage equaled the SSE calculated from the best-fit results, indicating that leakage into the SL/BM aquifer was minimal.

Results of the two aquifer tests indicate that aquifer properties did not change as a result of ASR cycling (table 3). Transmissivity values for the two tests are equal. Differences in the storage coefficient and the maximum leakage values are minimal and could be due to the insensitivity of the analytical

**Table 3.** Estimated hydraulic characteristics of the Santee Limestone/Black Mingo aquifer based on the results of the June 1999 and November 2001 aquifer tests at the phase-II aquifer storage recovery site in Charleston, South Carolina.

[L/s, liters per second; m, meters; (L/s)/m, liters per second per meter; $m^2$/d, meter squared per day; ---, not applicable]

| Test dates | June 28–July 7, 1999 | November 8–15, 2001 | Composite results |
|---|---|---|---|
| Average pumping rate (L/s) | 10.0 | 7.78 | --- |
| Maximum drawdown in CHN-812 (m) | 79.9 | 84.7 | --- |
| Specific capacity [(L/s)/m] at the given pumping rates | 0.12 | 0.092 | --- |
| Transmissivity ($m^2$/d) | 37 | 37 | 37 |
| Storage coefficient (dimensionless) | $2 \times 10^{-5}$ | $4 \times 10^{-5}$ | $3 \times 10^{-5}$ |
| Maximum leakage factor ($m^{-1}$) | $3 \times 10^{-4}$ | $7 \times 10^{-4}$ | $7 \times 10^{-4}$ |

method to the leakage factor. The consistent results of the two aquifer tests indicate that physical changes in the SL/BM aquifer as a result of calcite dissolution or other geochemical reactions have little effect on the hydraulic properties of the SL/BM aquifer during the time span tested.

Specific capacity, a measure of the well yield per unit of drawdown in the well, decreased over time during this investigation. Specific capacity depends on the well construction and hydraulic characteristics of the aquifer, decreases with increased discharge rate, and decreases with time during the pumping period (Heath, 1983). The maximum drawdown in the production well was greater during the November 2001 aquifer test (84.7 m) than during the June 1999 aquifer test (79.9 m), even though the average pumping rate was lower for the November 2001 test (7.78 L/s) than for the June 1999 test (10.0 L/s, table 3). Specific-capacity values for the June 1999 and November 2001 aquifer tests were 0.12 and 0.092 (L/s)/m, respectively (table 3).

Well plugging is defined as increased resistance to flow during artificial recharge (Pyne, 1995). Residual plugging, defined as plugging that remains during recovery, increases drawdown during recovery. The greater drawdown and coincident lower specific capacity for the November 2001 test compared with the June 1999 test indicate that plugging occurred at the site. Additional evidence of plugging was the decrease in injection rates for ASR cycles 2 through 4 (table 4), even though the same average injection pressure (145 kilopascals [kPa]) was maintained at the production well.

The introduction of foreign water to an aquifer may lead to many physical, biological, or chemical processes that degrade the quality of the injected water, the productivity of the aquifer, or the performance of the recharge well. The failure of an artificial recharge project in Arkansas identified several potential causes for reduction in recharge capability, including deposition of suspended material from water injected into the aquifer, air entrainment, rearrangement of aquifer materials, bacterial growth, biochemical or chemical precipitation, and swelling or dispersion of formation clays (Sniegocki, 1963). An artificial recharge investigation in Virginia identified clay dispersion as the major cause of hydraulic conductivity reduction during injection of freshwater into a brackish-water sand aquifer (Brown and Silvey, 1977).

**Table 4.** Recovery efficiencies during the phase-II aquifer storage recovery study in Charleston, South Carolina.

[L, liters; L/s, liters per second]

| ASR cycle | Dates | Volume injected[a] (L) | Storage period (days) | Volume of potable[b] water recovered (L) | Total volume recovered (L) | Recovery efficiency (percent) | Injection rate[a] (L/s) | Withdrawal rate (L/s) |
|---|---|---|---|---|---|---|---|---|
| 1 | 10/26/99 – 04/10/00 | 7,190,000 | 30 | 2,462,975 | 31,672,422 | 34 | 0.82[c]<br>1.3 | 8.83 |
| 2 | 05/08/00 – 09/11/00 | 6,900,000 | 34 | 1,922,901 | 33,953,168 | 28 | 2.0 | 8.07 |
| 3 | 09/11/00 – 04/02/01 | 7,010,000 | 99 | 1,513,027 | 39,439,068 | 22 | 1.9 | 7.57 |
| 4 | 04/02/01 – 02/15/02 | 6,905,248 | 168 | 1,478,508 | 52,013,489 | 21 | 1.57 | 7.78 |

[a]Volume injected and injection rates were estimated for cycles 1–3 using water-level data collected at well CHN-809 and the Moench (1985) analytical method for aquifer tests.

[b]Potable water in this report was based on the secondary standard for chloride of 250 milligrams per liter (U.S. Environmental Protection Agency, 2000).

[c]Injection rate varied for cycle 1.

Typically during the injection period of ASR cycles, production wells are periodically redeveloped by pumping to minimize the effects of plugging (Pyne, 1995). Redevelopment was not planned for this investigation but occurred inadvertently due to intermittent pump failures during the cycle-4 recovery phase (fig. 8). Specific-capacity values improved after each pump failure during this recovery phase, indicating that periodic redevelopment during the injection cycles should reduce the overall reduction in specific capacity during ASR implementation at the study site.

## Aquifer Storage Recovery Testing

Four cycles of injection, storage, and recovery were conducted between October 1999 and February 2002 (figs. 9–12). Test dates, volumes of water injected and recovered, injection and withdrawal rates, and recovery efficiencies for each ASR cycle are listed in table 4. Each test consisted of injecting between 6.90 and 7.19 million liters (L) of water for storage periods of 1, 3, or 6 months. A similar length of storage was selected for cycles 1 and 2 to determine if ASR cycling had an effect on recovery efficiency, stored and recovered water quality, and overall cycle performance. Storage period lengths of increasing duration were chosen for the last two cycles to evaluate residence-time effects on the water quality of stored and recovered water.

Total volumes of injected water determined from the inline totalizing flow meter during ASR cycles 1-3 were determined to be inaccurate. While the CCPW records contain no evidence that the meter was malfunctioning during the first three cycles, the volumes of injected water recorded for these cycles were significantly different from the volume recorded for ASR cycle 4. However, injection pressure measured at the production well and changes in water levels at the observation wells during the cycle-4 injection period were similar to the previous three tests. Because aquifer properties did not change during the ASR testing as determined from the two aquifer tests, the significant difference in total volume injected must have been a result of a malfunctioning flow meter. The meter was serviced during the storage period of ASR cycle 3 (December 2000) to replace the dial face plate, which was scratched and unreadable. During ASR cycle 4, the injection rate was higher than that recorded for the three previous cycles. It is assumed, therefore, that the service in December 2000 repaired the meter. No problems were experienced with the inline totalizing flow meter that measured discharge.

The total volume of water injected during ASR cycles 1–3 was estimated by using the Moench (1985) analytical method. Average injection rates for the cycles were estimated by using aquifer properties that accurately predicted the maximum measured drawdown at well CHN-809 for the November 2001 aquifer test. Two injection-rate estimates were made for cycle 1 because the average injection rate was adjusted during the cycle. The injection rates for cycles 1–3 were considered good estimates when the maximum predicted water-level change at well CHN-809 matched the maximum measured water-level change for the injection phases of the three ASR cycles. To confirm the appropriateness of this methodology, the Moench (1985) method was used to predict water-level changes during the injection phase of ASR cycle 4 by using the same aquifer property values used for cycles 1–3 and the measured injection rate for cycle 4. The Moench (1985) method predicted a maximum water-level change within 0.06 m of the measured water-level change of 3.16 m. This accuracy indicates that the methodology is appropriate for estimating the injection rates for the cycles when the flow meter was malfunctioning. In addition, numerical results of simulated water-level data for all four ASR cycles using the estimated injection rates compared well with measured data (discussed later in the Reactive-Transport Simulations section). The estimated injected volumes are listed in table 4.

Treated drinking water having an average specific conductance of 260 microsiemens per centimeter at 25 degrees Celsius (µS/cm) was injected into the production well at a constant rate (except cycle 1) until the specific-conductance value from the UPZ water-quality probe at well CHN-809 was less than 2,000 µS/cm for cycle 1 and 1,200 µS/cm for cycles 2–4. The injection rate was increased during cycle 1 to minimize the total time of injection. During cycles 2–4, the injection rate was controlled by maintaining a injection pressure of approximately 138 kPa on the pressure gage located at the top of the casing of production well CHN-812. The 2,000-µS/cm value was chosen as an arbitrary threshold, primarily because this is the approximate value at which the average rate of change in specific conductance decreased significantly from an initially high rate (figs. 9 and 13). For cycles 2–4, a threshold value of 1,200 µS/cm was used (figs. 10–12). This specific-conductance threshold value is less than a specific-conductance value of 1,467 µS/cm, which is approximately equivalent to the U.S. Environmental Protection Agency (USEPA) secondary standard (formerly known as the secondary maximum contaminant level) for chloride (250 milligrams per liter (mg/L), U.S. Environmental Protection Agency, 2000). Specific conductance and chloride are highly correlated at low-chloride concentrations (Hem, 1985). A linear regression of samples collected during this investigation indicates that a chloride concentration of 250 mg/L is equivalent to a specific-conductance value of about 1,467 µS/cm with a correlation coefficient of 0.99. The injection phase ended several days after the specific-conductance value decreased below the threshold level at well CHN-809.

The rates of change from initially high specific-conductance values of about 7,200 µS/cm (representative of the aquifer) to lower specific-conductance values (representative of a mixture of aquifer and injected water) were lowest for cycle 1 and approximately equal in the LPZ for ASR cycles 2–4 (fig. 13). The lower rate of change of specific conductance in the UPZ for most of the cycle-2 injection period was due to placement of the specific-conductance probe in a less permeable zone of the aquifer (116 m) than placement for cycles 3 and 4 (112 m). The specific-conductance probe was

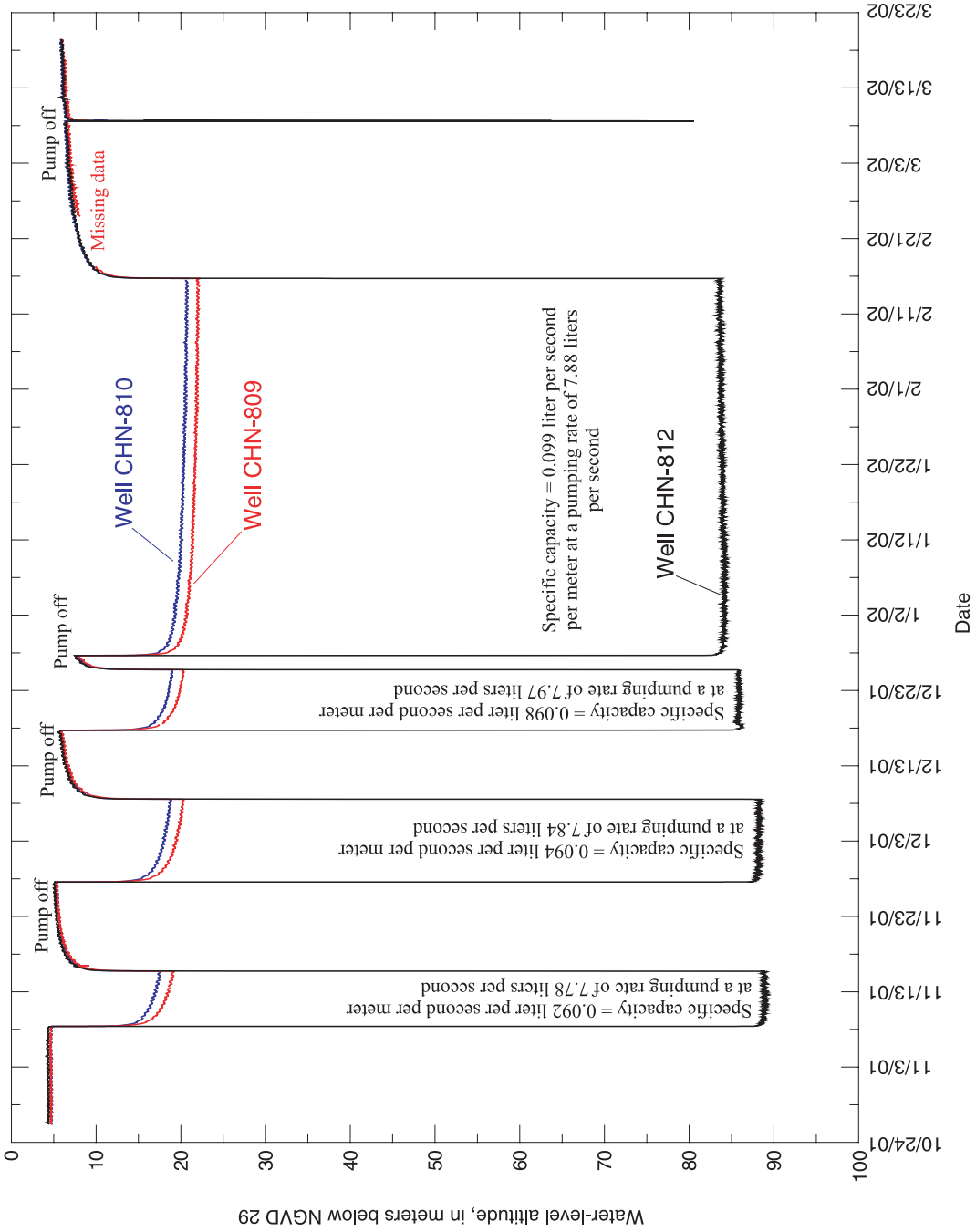


**Figure 8.** Water-level altitudes in wells during the cycle-4 recovery phase of the aquifer storage recovery study in Charleston, South Carolina.

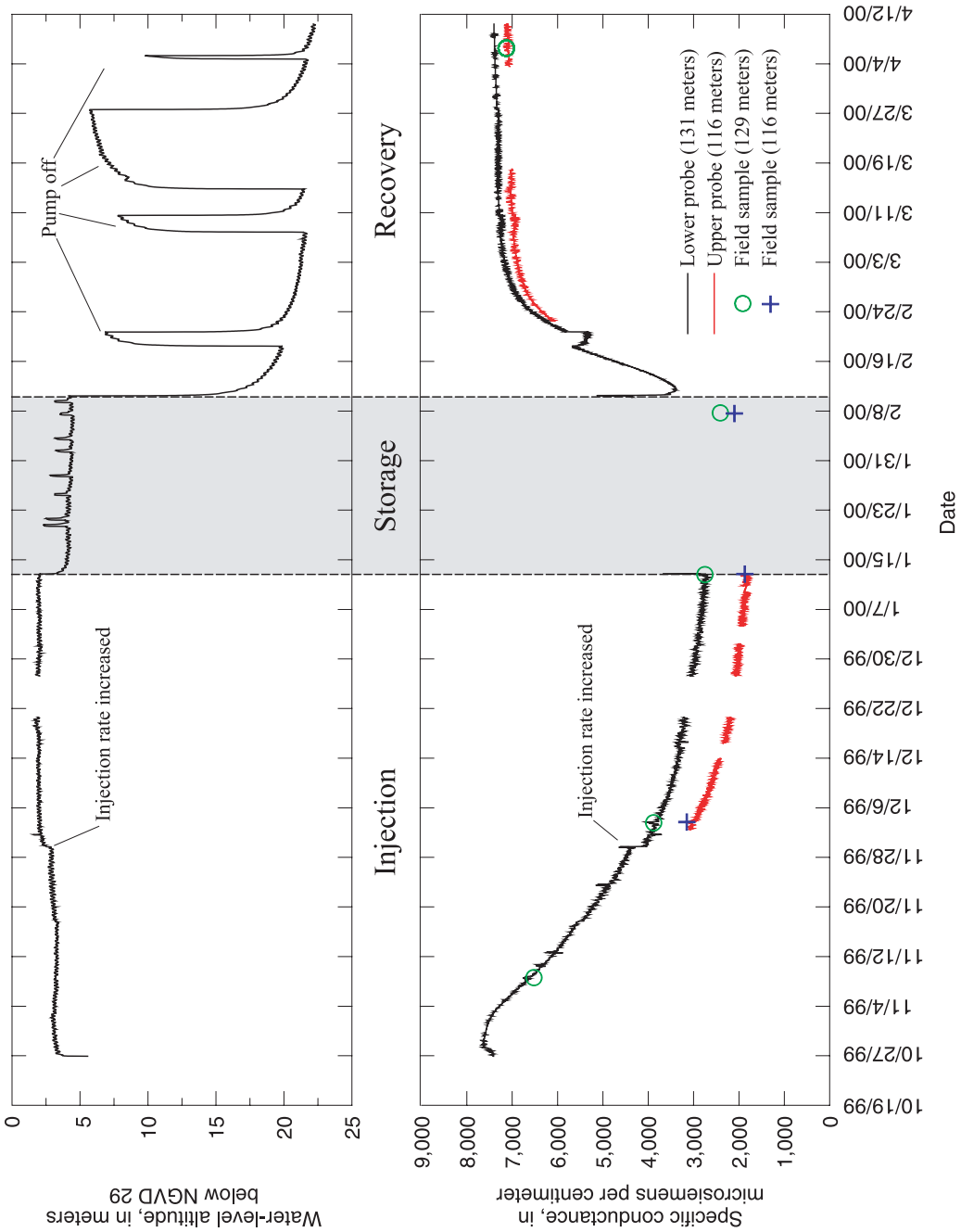


**Figure 9.** Water-level altitude and specific conductance at observation well CHN-809 during cycle 1 of the aquifer storage recovery study in Charleston, South Carolina.

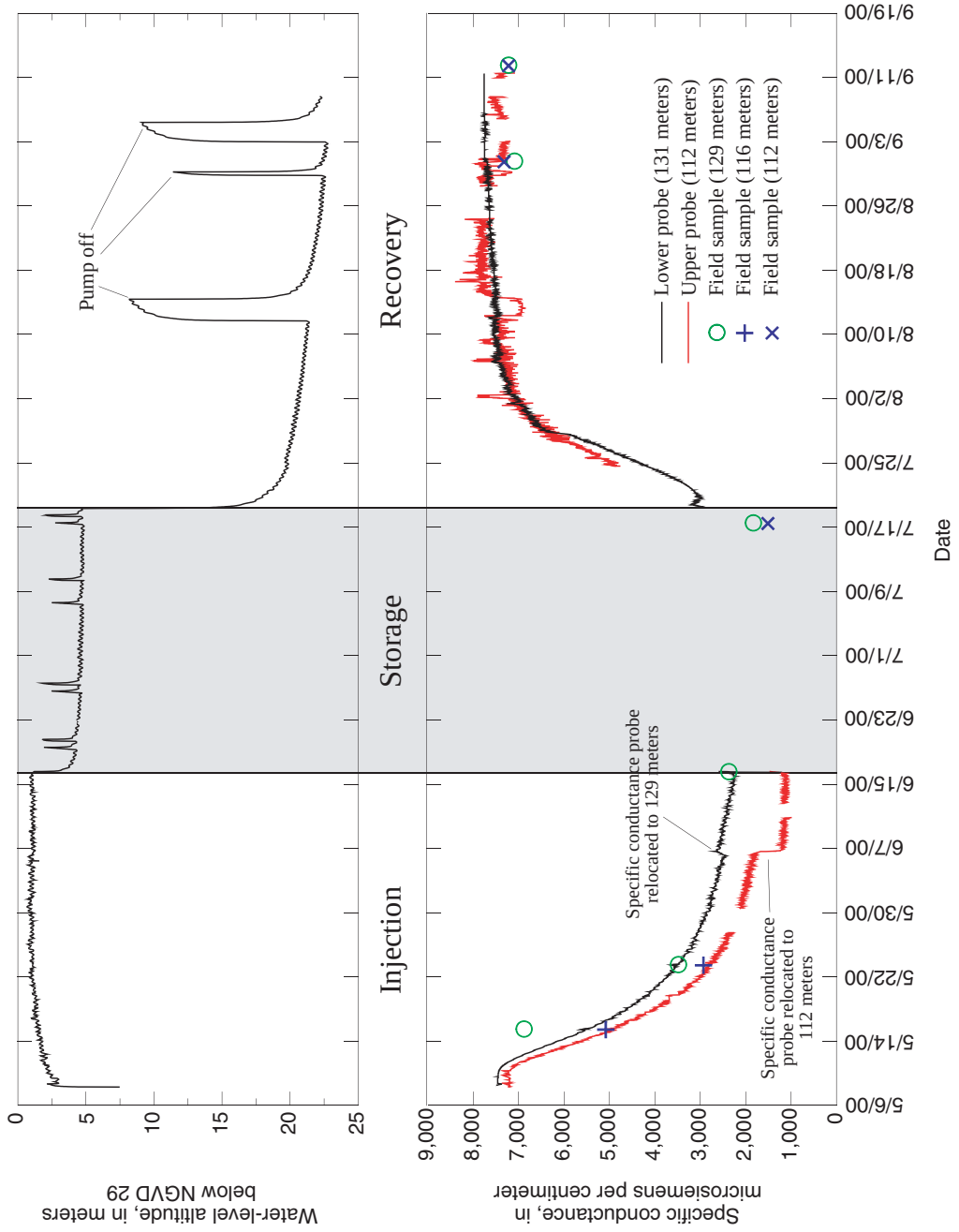


**Figure 10.** Water-level altitude and specific conductance at observation well CHN-809 during cycle 2 of the aquifer storage recovery study in Charleston, South Carolina.

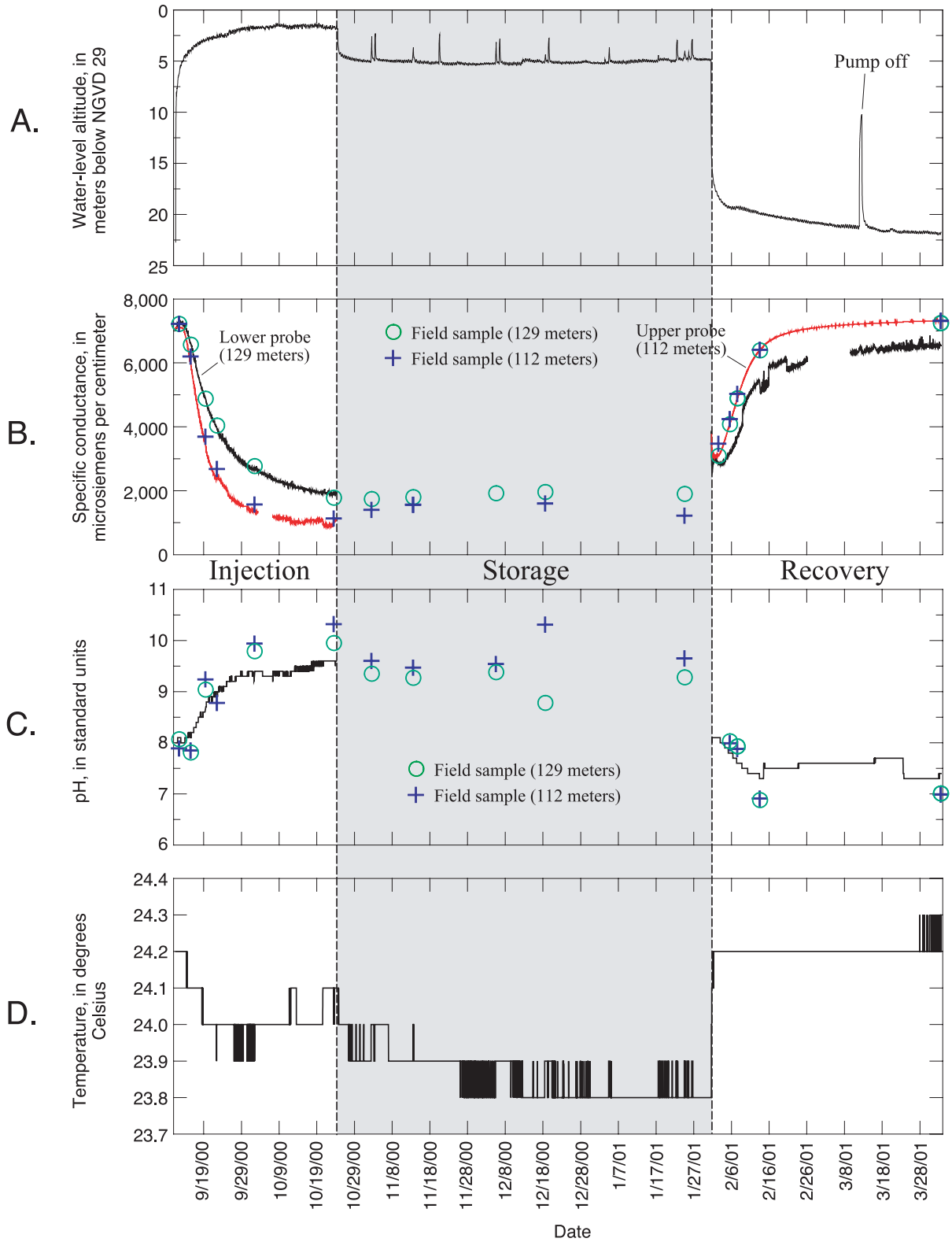


**Figure 11.** (A) Water-level altitude, (B) specific conductance, (C) pH, and (D) water temperature at observation well CHN-809 during cycle 3 of the aquifer storage recovery study in Charleston, South Carolina.

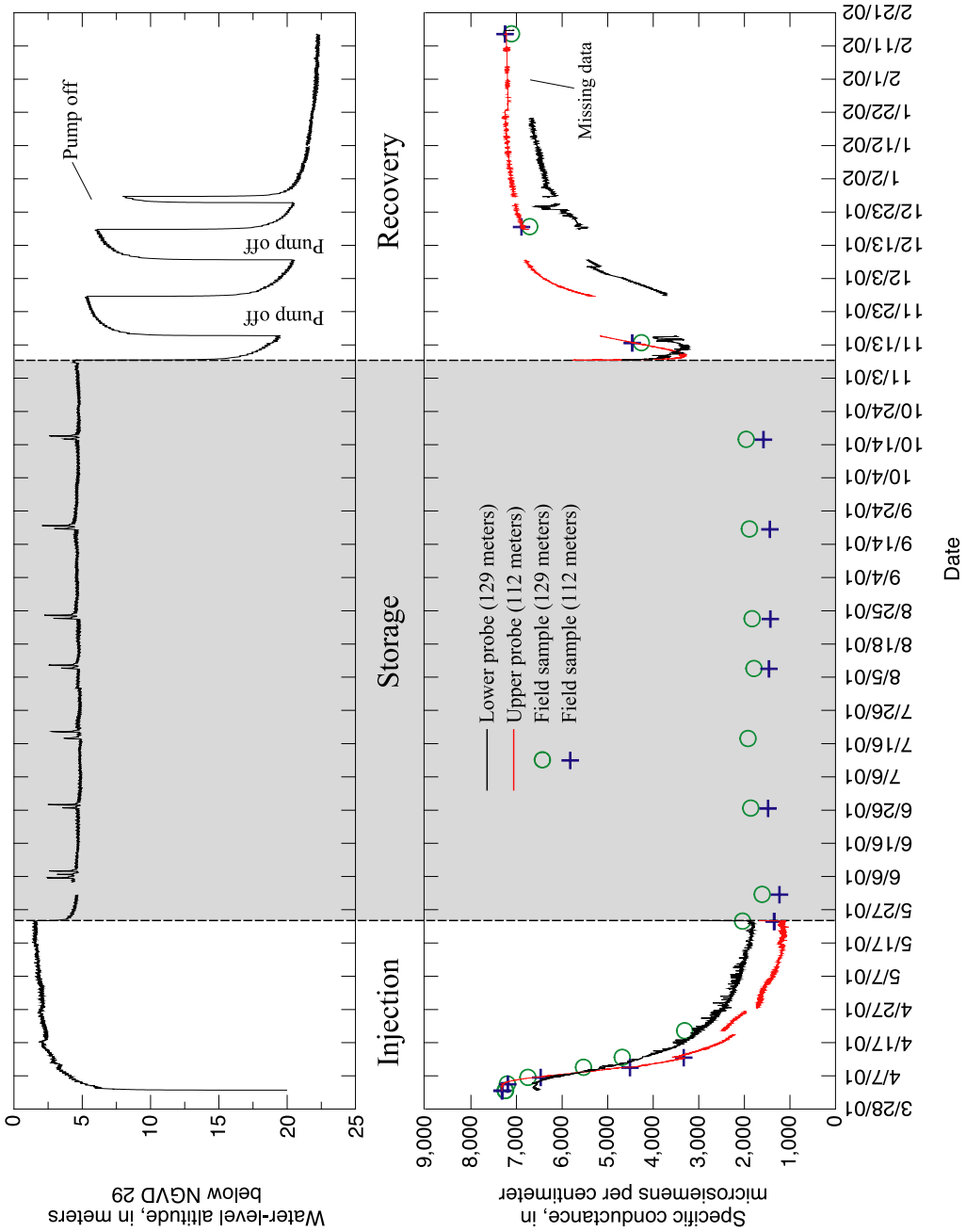


**Figure 12.** Water-level altitude and specific conductance at observation well CHN-809 during cycle 4 of the aquifer storage recovery study in Charleston, South Carolina.

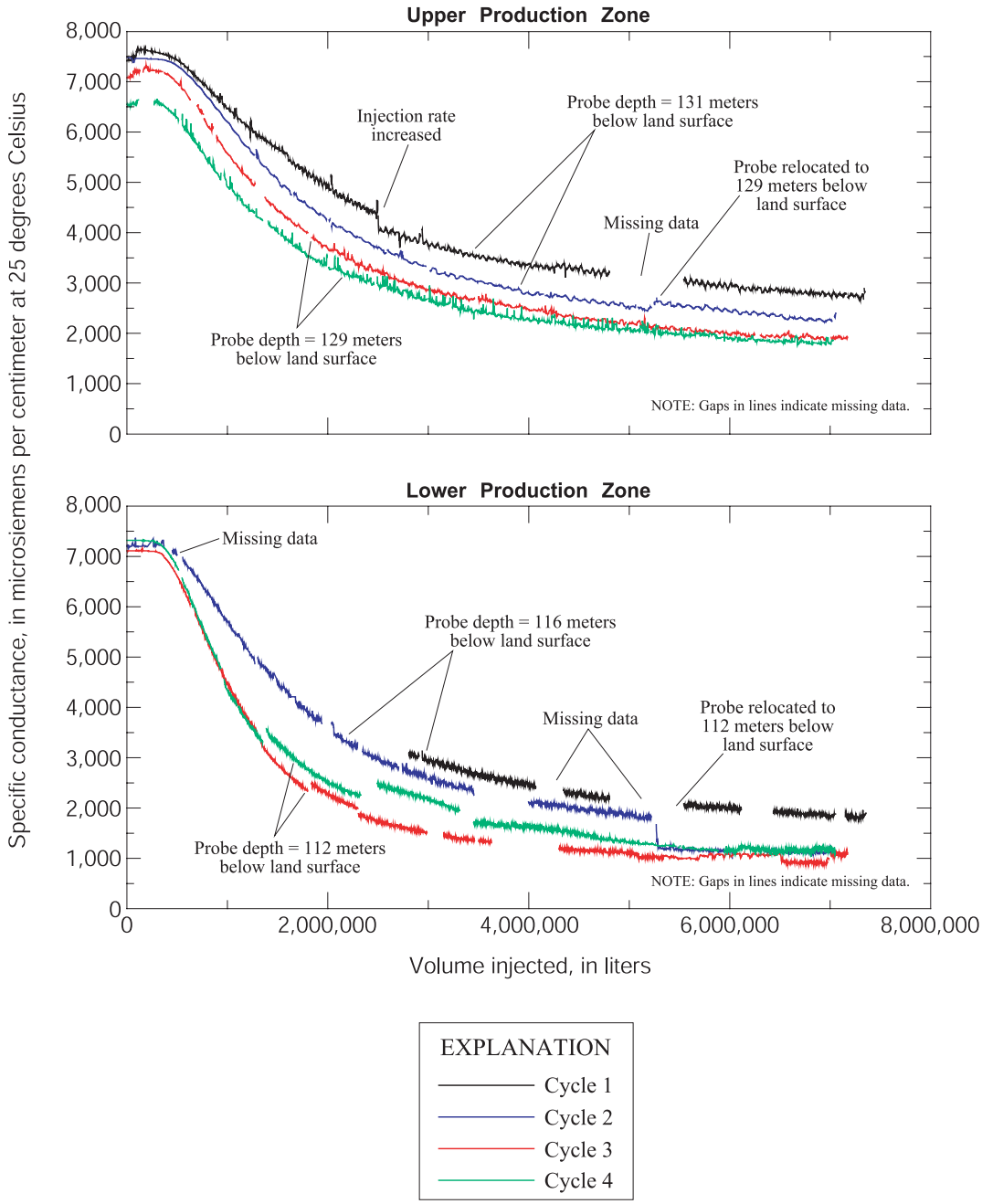


**Figure 13.** Changes in specific conductance at observation well CHN-809 during the injection phases of the aquifer storage recovery study in Charleston, South Carolina.

located at 112 m during cycles 3 and 4, and the rate of change of specific conductance was approximately equal in the UPZ during these cycles.

The length of the recovery phase was determined by the amount of time required to completely pump all of the injected water from the SL/BM aquifer. Recovery was stopped when the specific conductance of the water recovered from the production well equaled pre-injection values of about 7,200 µS/cm (figs. 9–12). Removal of all of the injected water was of utmost importance, so that differences in water-quality evolution (at the observation wells and production well) and recovery efficiency could be attributed to length of storage and not changing water-quality end members. For this reason and because of mixing that occurred during ASR testing, the volume of recovered water always significantly exceeded the volume of injected water (table 4).

Specific-conductance profiles were collected in wells CHN-809 and CHN-810 periodically during cycles 3 and 4 to delineate the UPZ and LPZ (fig. 14). Profiles were collected during injection, storage, and recovery phases. Results from the specific-conductance profiles are less revealing than the fluid resistivity or EMFM logs; however, an upper productive zone is evident in both wells. Specific-conductance profiles collected during injection show the gradual displacement of brackish water in the SL/BM aquifer during the injection phase of cycle 4. The UPZ is a thin zone ranging in depth from 112 to 115 m bls in well CHN-809 and from 112 to 114 m bls in well CHN-810. The LPZ is not apparent from the specific-conductance profiles.

# Geochemical Methods and Processes

During this investigation, 193 water-quality samples were collected and analyzed for physical properties, major and minor ions, nutrients, and total trihalomethane concentrations. Water-quality analytical results coupled with inverse geochemical modeling aided in the evaluation of the dominant geochemical processes that occurred during the ASR tests in the SL/BM aquifer. Water-quality samples collected at the production well aided in estimating the recovery efficiency or volume of potable water that could be recovered during ASR implementation.

## Water-Quality Sampling

Water-quality samples were collected directly from water spigots inline with the CCPW water mains and production-well discharge pipe and by using a portable piston-driven submersible pump. Five treated surface-water samples were collected to document injectant water quality. Thirty-two samples were collected from the production well during the recovery phase of the ASR cycles. A submersible pump was used to collect 151 samples from observation wells CHN-809 and CHN-810. Three samples from observation well CHN-811A and one sample from CHN-800 were collected to document water-quality differences among the observation wells.

Samples collected during this investigation were analyzed for physical properties, major and minor ions, and nutrients (Appendix 1). Physical properties included pH, water temperature, specific conductance, and dissolved-oxygen concentration, which were measured by using a water-quality field meter. Samples were shipped to the USGS water-quality laboratory in Ocala, Fla., and analyzed for major and minor ions, acid-neutralizing capacity (ANC), and nutrients by using standard USGS methods. In addition, 44 samples were collected for analysis of total trihalomethane concentrations from wells CHN-809 and CHN-810 during injection and storage, and from well CHN-812 during recovery of ASR cycle 4 (Appendix 2). One treated drinking-water sample was analyzed to determine the total trihalomethane concentration of the injected water. Water-quality samples were analyzed using USEPA method 502.2 (U.S. Environmental Protection Agency, 1988) for total trihalomethane concentrations at the CCPW laboratory in Hanahan, S.C. The CCPW laboratory is certified by the South Carolina Department of Health and Environmental Control.

Ground-water samples were collected from the production and observation wells using standard or modified USGS methods (Gibs and Wilde, 1998). Ground-water samples were collected from the observation wells by using a submersible pump and the low-flow purging technique to assure collection of aquifer water and not water stored within the well bore. The sampling method differed from the standard USGS method in that instead of installing sample tubing 24 hours prior to sampling, sample tubing was placed in the well the day of sampling (Barcelona and others, 1994; U.S. Environmental Protection Agency, 1995, 1996). Prior to sampling, the pressure transducer and one of the specific-conductance probes were removed from the well. The submersible pump was lowered to a depth adjacent to the screened section of the UPZ (112 or 116 m bls). Pumping began after moving the down-hole water-quality meter as close to the pump as possible. Discharge from the submersible pump was monitored with a second water-quality meter and a flowthrough cell at land surface. The water level was monitored during the low-flow sampling process to ensure that the pumping rate was not causing drawdown in the sampled well. When water-quality properties (pH, temperature, specific conductance, and dissolved-oxygen concentration) at the pump discharge line stabilized and approximated those measured near the pump intake, the water sample was collected. After sample collection from the UPZ, the water-quality probe and submersible pump were lowered to the screened section adjacent to the LPZ (129 m bls), and a ground-water sample was collected using this same method.

During the storage period of all ASR cycles, values of specific conductance in the observation wells increased considerably due to well-bore mixing. Because of this and to diminish the sampling time required, treated surface water was injected in the production well during storage. Injection at the production well forced the high specific-conductance well-bore

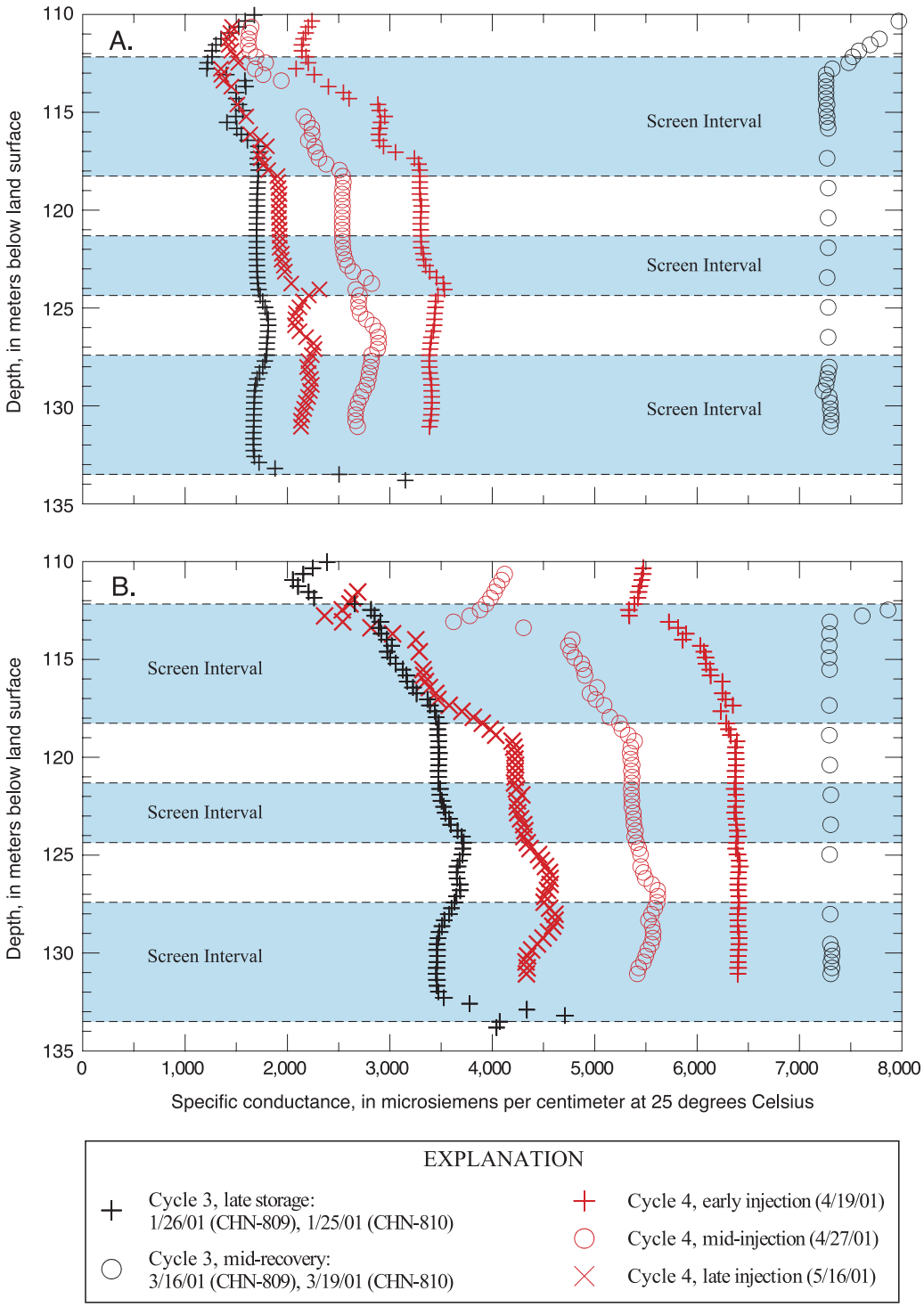


**Figure 14.** Specific-conductance profiles collected in observation wells (A) CHN-809 and (B) CHN-810 during cycles 3 and 4 of the aquifer storage recovery study in Charleston, South Carolina.

water out of the screened zones within the observation wells, which allowed more rapid stabilization of physical properties during the low-flow sampling procedure. The specific conductance in samples collected during storage (figs. 11 and 12) indicates that this sampling method allowed collection of ground-water samples representative of the productive zones of the SL/BM aquifer. The volume of water injected during storage was accounted for when calculating recovery efficiency as discussed below.

Although brackish-water upconing was evident during the pilot ASR investigation, upconing was not evident during the phase-II ASR testing. Well-construction design accounted for the lack of upconing during the phase-II investigation. Wells installed during the pilot investigation (Campbell and others, 1997) were of open-hole construction and were open to the SL/BM aquifer and upper and lower confining units; wells installed in the phase-II investigation were cased in the upper confining unit and screened solely within the SL/BM aquifer.

## Recovery Efficiency

Recovery efficiency is the percentage of stored water that meets a target water-quality criterion after retrieval from the aquifer (Pyne, 1995), and it is used as one measure to evaluate the results of an ASR investigation. The overall magnitude of this diagnostic property and changes in magnitude from one cycle to the next are important factors in determining whether ASR implementation is a worthwhile endeavor for the aquifer under investigation. Recovery efficiencies are controlled by hydrogeologic conditions, well and well-field design, and operational management (Merritt, 1985; Reese, 2002). Hydrogeologic conditions that can reduce recovery efficiency include severe mixing in aquifers that are thick, stratified, heterogeneous, or highly permeable; buoyancy stratification; migration of the freshwater bubble in aquifers with a high regional hydraulic gradient or in dipping aquifers; and migration of freshwater into semiconfining units in poorly confined aquifers (Merritt, 1985; Reese, 2002). Well design and management practices that can reduce recovery efficiency include using partially penetrating injection wells in aquifers in which buoyancy stratification may cause the freshwater bubble to rise above the screened or open portion of the well and well plugging, which can reduce recovery from the affected zones (Merritt, 1985; Reese, 2002). Improvements in recovery efficiencies that have been observed in field investigations and numerical simulations include (1) increasing the volume of water injected, (2) successive cycling after not fully recovering all of the previously injected water, and (3) centering adjacent injection wells around a central well when using multiple injection wells (Merritt, 1985; Reese, 2002).

For this study, the USEPA secondary standard for chloride (250 mg/L; U.S. Environmental Protection Agency, 2000) was selected as the criterion for evaluating recovery efficiency. Chloride was selected because it behaves conservatively in the carbonate aquifer, and chloride concentrations in the treated surface water (19–60 mg/L) and ambient aquifer water (1,800 mg/L) differed significantly (Appendix 1).

During the phase-II investigation, the total volume of recovered water greatly exceeded the total volume injected to assure that all injected water was removed from the aquifer prior to the start of a new ASR test (table 4). During the ASR pilot tests, recovery of injected/stored water was discontinued prior to complete retrieval of all of the injected water. As a result, the pilot investigation results indicated that recovery efficiency was between 38 and 61 percent of the total volume injected (Campbell and others, 1997). Because all of the injected water was not removed during each of the pilot recovery cycles, pilot recovery efficiencies were recalculated by adding the volume of water injected that remained in the aquifer at the end of one cycle to the total volume injected for the subsequent cycle. The volume of injected water remaining in the aquifer was estimated by using specific-conductance values representative of the injected water (174 µS/cm), native ground water (6,530 µS/cm), and recovered water samples collected immediately prior to the end of recovery (3,620 to 5,110 µS/cm). This method assumes that specific conductance is a conservative tracer and that the relation between specific conductance and the percentage of injected water recovered is linear, a relation confirmed by the pilot ASR results (Campbell and others, 1997). A linear regression of specific conductance and chloride data from the pilot investigation resulted in a correlation coefficient of 0.99, indicating that specific conductance behaved conservatively in the pilot ASR investigation. Recalculated recovery efficiencies for the pilot ASR tests of 29 to 39 percent are more appropriate for comparison with the phase-II results.

To calculate recovery efficiency, the volume of water injected during the storage period (to facilitate sampling the observation wells) was added to the total volume injected during the injection period. The total volumes injected were similar for all cycles. The estimated volume of treated surface water injected during the storage periods varied between 276,305 and 442,845 L, which is between 4 and 6 percent of the total volume injected for the individual cycles.

Recovery efficiencies for the phase-II investigation varied between 21 and 34 percent for the four ASR tests (table 4). Recovery efficiencies were essentially equal for the 3-month (cycle 3) and 6-month (cycle 4) storage cycles. The recovery efficiencies are nearly equal even though injection and recovery rates differed for the two tests. Solute transport modeling, discussed below, is consistent with a fixed recovery efficiency of approximately 25 percent, regardless of the length of the injection period.

Recovery efficiencies of 21 to 34 percent during this investigation compare well with the results of 12 ASR investigations of the Floridan aquifer system (Reese, 2002). Recovery efficiencies ranged from 2 to 47 percent during the first cycle of each of these Floridan aquifer system investigations. Of the 10 investigations that had multiple ASR

cycles, 7 showed an improvement in recovery efficiencies during the second cycle. The percent of injected water that was left in the aquifer after completion of the first cycle varied from 23 to 92 percent for each of the Floridan aquifer system investigations (Reese, 2002). A common practice in ASR implementation is to limit recovery to allow some injected water to remain in the aquifer (Pyne, 1995; Reese, 2002). This procedure typically allows greater recovery efficiencies for subsequent cycles, as was the case in the pilot investigation (Campbell and others, 1997). During the phase-II investigation, essentially all injected water was removed from the aquifer prior to initiating the subsequent ASR cycle. The phase-II ASR investigation was designed to determine the amount of water that could be recovered following a single injection period, which is more consistent with the planned use of the stored water.

## Chemical Composition of Waters

In the ASR investigation, treated water with total-dissolved-solids (TDS) concentration of approximately 100 mg/L was injected into the SL/BM aquifer containing brackish water with a TDS of approximately 4,000 mg/L. The dominant cation of the brackish aquifer water was sodium and the dominant anion was chloride (fig. 15A). The dominant cations in the treated water were calcium and sodium and the dominant anion was bicarbonate (fig. 15B). Upon injection into the aquifer, the composition of the injected water changed through mixing with the brackish water and reaction with the aquifer sediments. At the commencement of recovery during ASR cycle 3, water composition in the aquifer varied with distance from the injection and recovery well. Near the recovery well, as indicated by the initially recovered water from CHN-812 (fig. 15D), the water composition was similar to the injected water, although sodium, calcium, chloride, and bicarbonate concentrations were slightly higher in the recovered water than in the injected water. At a distance of 23 m (well CHN-809, UPZ), the composition of the stored water (fig. 15C) was quite different from the injected water (fig. 15B). The stored water had a substantial concentration of sodium (15 millimoles per liter (mmol/L) or milliequivalents per liter (meq/L)) and chloride (8 mmol/L or meq/L, fig. 15C) relative to the injected water (fig. 15B), which indicates that part of the stored water mixed with the brackish aquifer water. In addition, calcium and magnesium concentrations in stored water at well CHN-809 were lower than in the injected water, and bicarbonate concentration was higher relative to chloride than in the brackish aquifer water.

As water continued to be pumped from the aquifer, the water composition changed to that of the brackish aquifer water. During this recovery process, sodium and sulfate concentrations behaved nearly conservatively relative to chloride. However, calcium and magnesium concentrations were slightly lower, and potassium and alkalinity concentrations were slightly higher than would be expected from conservative mixing of initial recovered water and final recovered water (fig. 16). The nonconservative behavior of the cations is indicative of cation-exchange processes that alter the cation composition of the water during ASR.

## Trihalomethanes

Trihalomethanes (THMs) are disinfection by-products that are formed when water containing naturally occurring dissolved organic carbon (DOC), such as water from the Edisto River and Bushy Creek Reservoir, is chlorinated for drinking-water purposes. Trihalomethanes are carcinogenic and, therefore, are regulated by the USEPA. The maximum contaminant level (MCL) for total THMs is 80 micrograms per liter (µg/L; U.S. Environmental Protection Agency, 2002). Total THMs consist of chloroform, bromoform, bromodichloromethane, and chlorodibromomethane.

Ground-water and treated surface-water samples were collected for THM analysis during ASR cycle 4 to determine the fate of THMs during ASR implementation in the SL/BM aquifer. Analytical results for samples collected during cycle 4 are listed in Appendix 2. The highest concentrations of total THMs were measured in the treated drinking water prior to injection (57 µg/L). For this sample, the concentration of chloroform, bromodichloromethane, and chlorodibromomethane were 25, 20, and 12 µg/L, respectively. Bromoform was not detected (<1 µg/L) in this sample. The total THM concentration in the treated drinking-water sample was consistent with the range of total THM concentrations (46 to 97 µg/L, with an annual average of 70 µg/L) reported in Charleston's drinking-water supply for published sample results collected between January and December 2001 (Charleston Commissioners of Public Works, 2003). Chloroform was the only constituent that contributed to the total THM concentration in ground-water samples. During recovery, total THM concentrations decreased from 8.5 to less than the detection limit (4 µg/L) at the production well. At these concentrations, THMs would not restrict the use of the recovered water for drinking-water purposes.

Analytical results collected from observation wells CHN-809 and CHN-810 during the cycle-4 storage period indicate that while bromodichloromethane and chlorodibromomethane decreased to less than individual THM detection limits (<1 µg/L) during injection and storage, chloroform remained above detection limits during about 5.5 months of residence in the aquifer (fig. 17). During this storage period, chloroform concentrations at the two wells ranged from 3 to 28 µg/L (Appendix 2). While chloroform concentrations in well CHN-809 decreased over time, concentrations in well CHN-810 varied little or increased slightly.

The highest chloroform concentration measured at well CHN-809 (28 µg/L) approximately equaled the treated drinking-water sample collected during this cycle (25 µg/L).

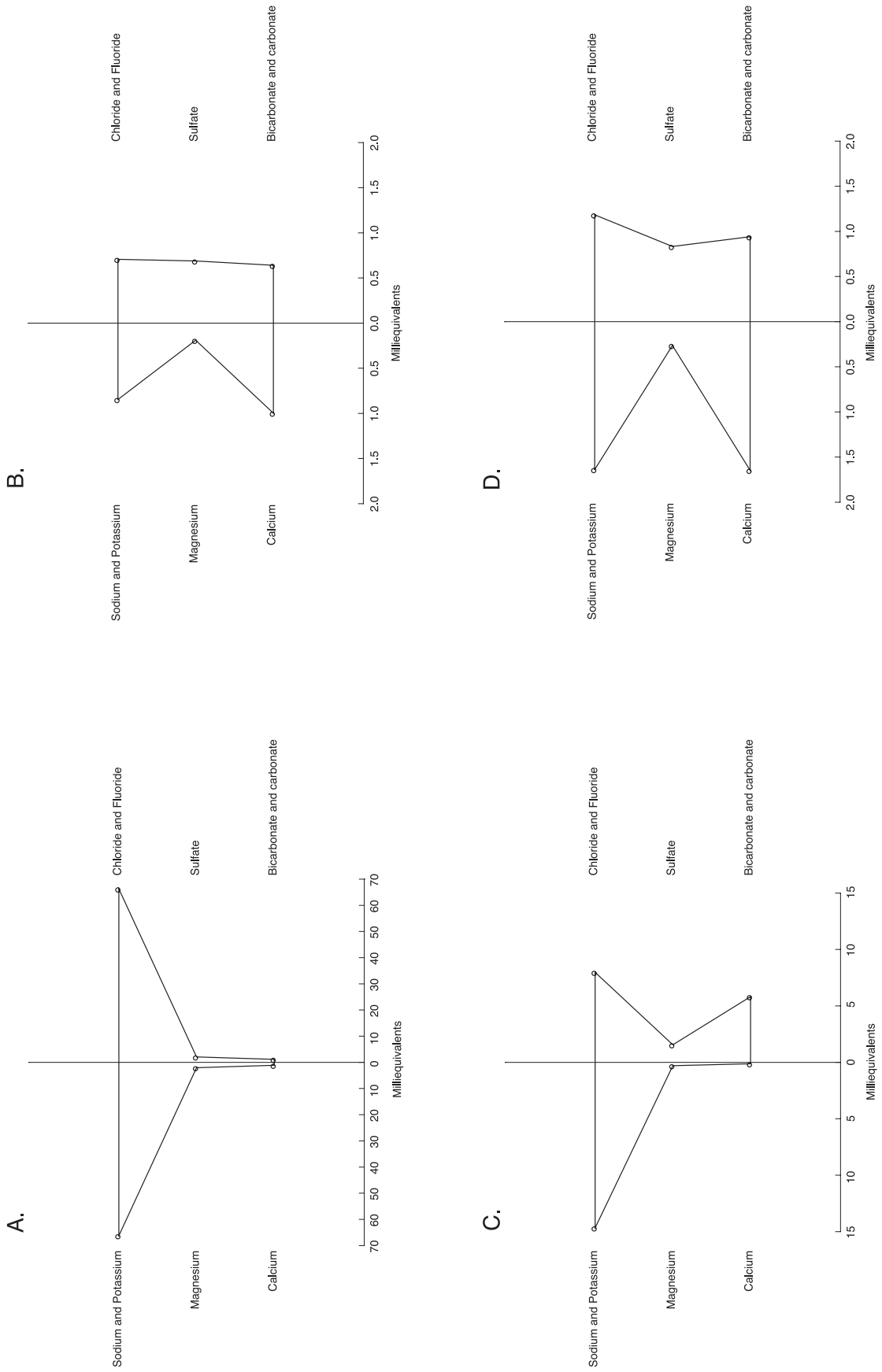


**Figure 15.** Stiff diagrams depicting water quality in (A) native ground water, (B) injected water, (C) late storage from observation well CHN-809, and (D) early recovery from production well CHN-812 during cycle 3 of the aquifer storage recovery study in Charleston, South Carolina.

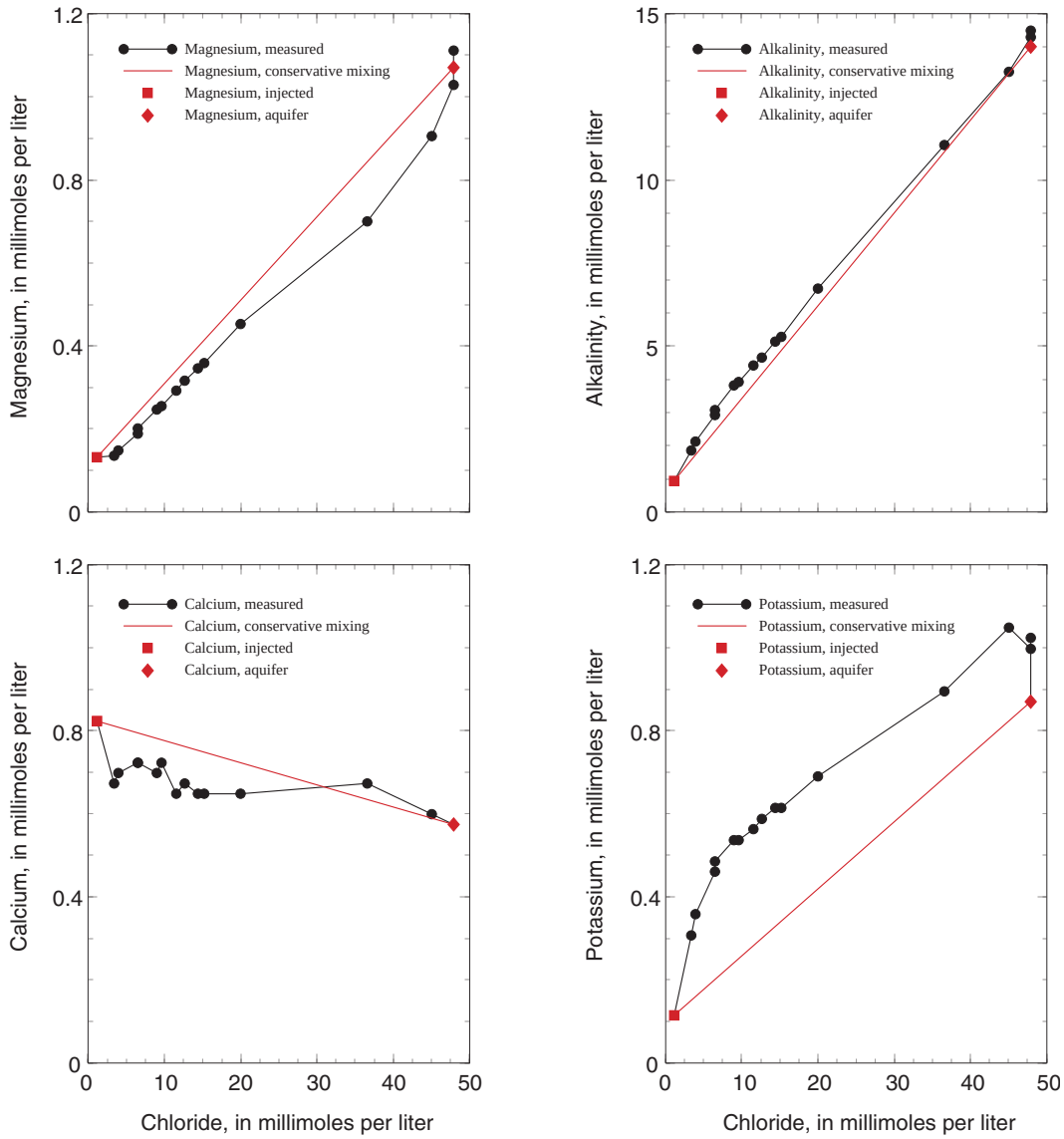


**Figure 16.** Concentrations of selected ions relative to chloride concentrations during the cycle-3 recovery phase of the aquifer storage recovery study in Charleston, South Carolina.

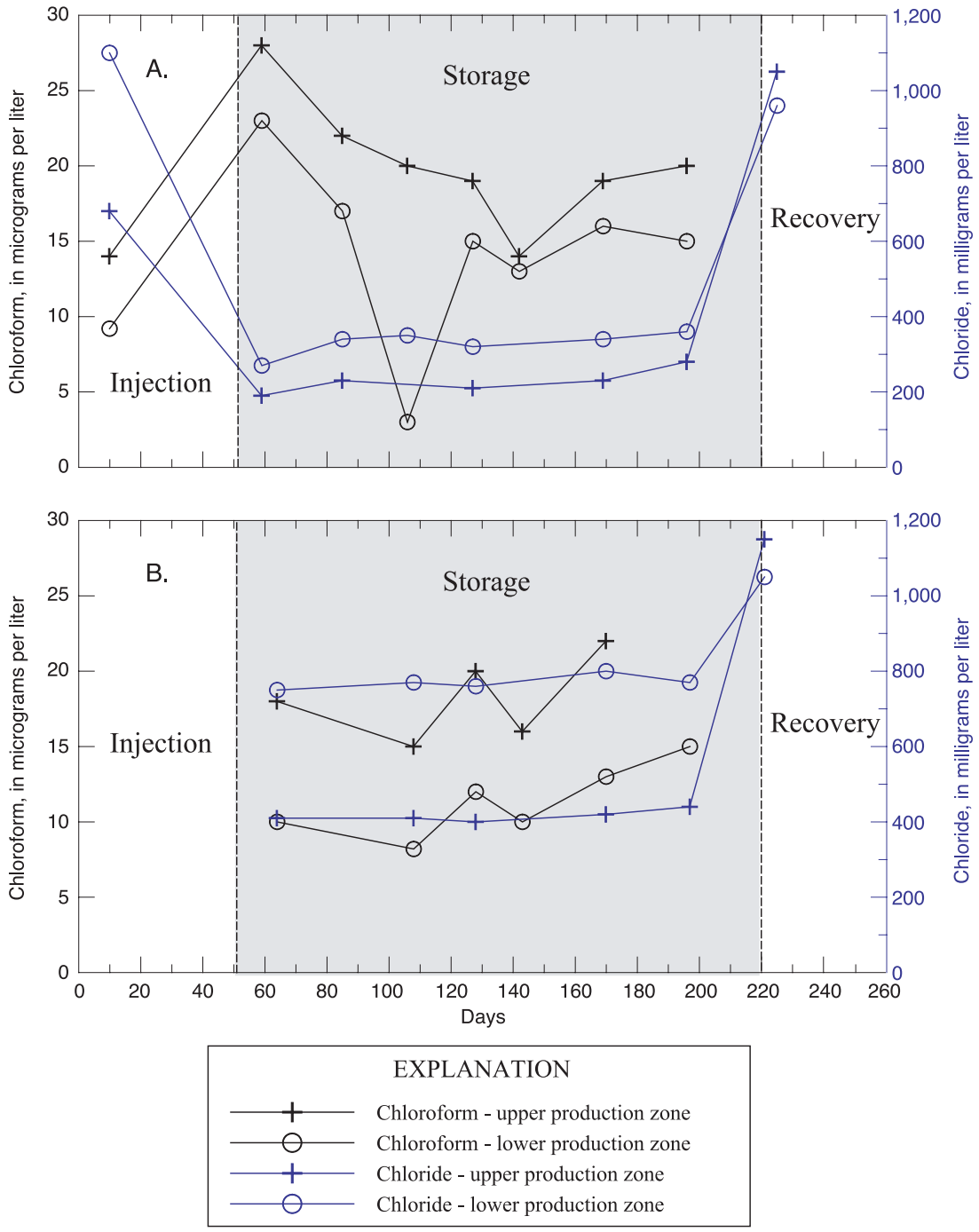


**Figure 17.** Chloroform and chloride concentrations in samples collected from observation wells (A) CHN-809 and (B) CHN-810 during cycle 4 of the aquifer storage recovery study in Charleston, South Carolina.

The fact that this value was higher than the measured concentration in the treated drinking-water sample could be attributed to variability in the THM concentrations in treated drinking water over time. The sample containing 28 µg/L of chloroform was collected early in the storage cycle. The chloroform concentrations generally stabilized in well CHN-809 during mid to late storage with concentrations of 14 to 20 µg/L in the UPZ and 13 to 16 µg/L in the LPZ (fig. 17A, Appendix 2). Chloride concentrations measured in samples collected from well CHN-809 during storage were relatively consistent (fig. 17A; Appendix 2).

The relative percentages of injected water in ground-water samples collected at the observation wells were calculated and plotted (fig. 18) to determine if chloroform behaved conservatively (like chloride) during ASR implementation. Comparison of the relative percentages of injected water based on chloroform and chloride concentrations (fig. 18; Appendix 2) indicated that the majority of the estimates based on chloroform concentrations were lower than the estimates based on chloride. Therefore, mixing of injected and aquifer water cannot solely explain the reduction in chloroform concentrations. At least three chloroform samples collected during this investigation represented a higher concentration than could be explained by mixing. Although the reason for this disparity is unknown, it could be attributed to the variability of chloroform concentrations in the injected water over time.

Relative percentages of injected water in ground-water samples based on chloroform concentrations compared to chloride concentrations (fig. 18) and the absence of bromoform, bromodichloromethane, and chlorodibromomethane in any ground-water samples indicated that the majority of all THM samples collected during ASR cycle 4 were nonconservative. The mechanism of THM reduction was not evaluated during this investigation; however, previous investigations (Pyne, 1995; Thomas and others, 2000) indicate that mixing and microbial biodegradation may decrease THM concentration during storage and recovery.

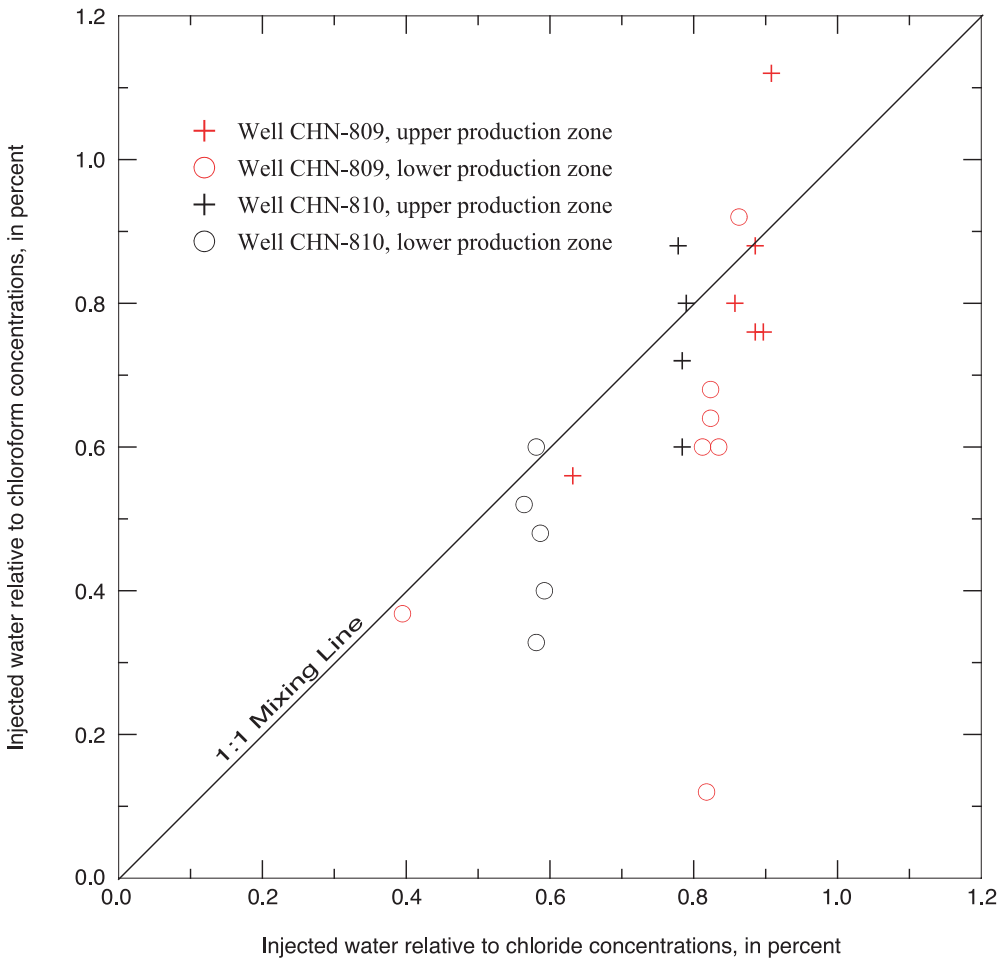


**Figure 18.** Relative percentages of injected water in samples collected from observation wells CHN-809 and CHN-810 during cycle 4 of the aquifer storage recovery study in Charleston, South Carolina.

## Chemical Reactions During Aquifer Storage Recovery

The observed changes in water composition in the stored and recovered waters can be explained by chemical reactions between the injected surface water and the ground water and aquifer sediments. The predominant reactions in Coastal Plain aquifers are carbonate-mineral and cation-exchange reactions (Chapelle and Knobel, 1985). Both dolomite and calcite are carbonate minerals that occur in Coastal Plain sediments, but of the two, only calcite was identified by quantitative X-ray diffraction (table 1) in sediment samples. Cation exchange is indicated by the variations in calcium and sodium concentrations measured in the stored and recovered waters (fig. 15). Thus, calcite dissolution and precipitation and cation exchange are likely reactions in the SL/BM aquifer.

In addition, it is possible to have redox reactions involving dissolved oxygen, organic matter, iron, and sulfur. If organic matter is present in the aquifer, sulfate reduction is a possible reaction. However, the concentration of sulfate was nearly conservative with respect to chloride throughout the investigation, and sulfide concentrations were low in the 59 samples that were analyzed. It is concluded that during the investigation, sulfate reduction was minimal.

The injection of oxygenated water could lead to oxidation of pyrite, organic carbon, and other reduced substances. The maximum amount of dissolved oxygen in the injected water was approximately 0.25 mmol/L (8 mg/L), which can be expected from equilibration of surface water with the atmosphere. This amount of dissolved oxygen is sufficient to generate only about 0.125 mmol/L of sulfate (12.5 mg/L), which is negligible. By the same reasoning, oxidation of reduced iron (ferrous iron) is unlikely to have a large effect on water composition; however, iron oxidation may still be an important reaction in ASR. Oxidation of ferrous iron is mediated by bacteria, and bacterial colonies and the formation of ferric oxyhydroxide precipitates can lead to well clogging. Because of the small effect on water composition, no chemical data are available that are adequate to evaluate whether iron oxidation contributes to the observed decrease in specific capacity over the course of the ASR experiments.

Other reactions with gypsum and aluminosilicates are thought to be unlikely or relatively unimportant. Gypsum is undersaturated in the brackish water (saturation index (SI) < -2) and should not precipitate under injection conditions. Gypsum was not identified by X-ray diffraction analysis and was assumed not to be present within the aquifer. Although aluminosilicate minerals are present in the aquifer, it was assumed that reactions with these minerals were too slow to substantially affect the chemistry or aquifer properties during the four ASR cycles.

The partial pressure of carbon dioxide ($CO_2$, table 5), as calculated from pH and alkalinity by geochemical speciation modeling, is approximately atmospheric (log partial pressure = -3.5) for the injected water and the initial recovered water; however, the partial pressure of carbon dioxide was considerably less than atmospheric in the water at well CHN-809. The low partial pressure at CHN-809 is consistent with dissolution of calcite, which consumes dissolved $CO_2$. Saturation indices for calcite (table 5) indicate that the brackish water in the aquifer is at equilibrium with calcite (SI = 0.02). The injected water is undersaturated with respect to calcite (SI = -0.74), which indicates that calcite may dissolve as the water is injected into the aquifer. The water at CHN-809 at the start of recovery was supersaturated with respect to calcite (SI = 0.76). This supersaturation may have been caused by mixing of waters within the well bore during sampling. At high-pH and low-calcium concentrations, large supersaturations can be caused by mixing waters with differing concentrations of $CO_2$ or calcium.

Inverse geochemical modeling with PHREEQC (Parkhurst and Appelo, 1999) was used to account for the difference in

**Table 5.** Chemical composition of four water samples collected during cycle 3 of the aquifer storage recovery study in Charleston, South Carolina.

[°C, degrees Celsius; su, standard units; mg/L, milligrams per liter; $CaCO_3$, calcium carbonate; $CO_2$, carbon dioxide; <, less than]

| Constituent | Brackish background aquifer water (CHN-812) | Injected surface water | Initial recovered water (CHN-812) | Start of recovery (CHN-809) |
|---|---|---|---|---|
| Sample date | 3/1/01 | 11/2/00 | 1/31/01 | 12/18/00 |
| Temperature, °C | 25.0 | [a]25.4 | [b]21.9 | 17.5 |
| pH[c], su | 7.6 | 7.8 | 7.9 | 10.3 |
| Calcium, mg/L | 23 | 20 | 33 | 2.6 |
| Magnesium, mg/L | 25 | 2.3 | 3.2 | 3.5 |
| Potassium, mg/L | 39 | 2.4 | 4.5 | 12 |
| Sodium, mg/L | 1,500 | 18 | 35 | 330 |
| Alkalinity[d], mg/L as $CaCO_3$ | 715 | 32 | 47 | 291 |
| Chloride, mg/L | 1,700 | 23 | 40 | 280 |
| Silica, mg/L | 41 | 7.6 | 10 | 19 |
| Sulfate, mg/L | 260 | 33 | 40 | 75 |
| Calcite saturation index | .02 | -.74 | -.67 | 0.76 |
| Log partial pressure $CO_2$ | -1.70 | -3.23 | -3.26 | -5.42 |
| Dissolved oxygen, mg/L | <2.0 | [a]7.0 | [e]<2.5 | <1.5 |

[a]Average value for two samples collected on 4/11/01 and 5/23/01.
[b]Temperature value measured on 2/1/01.
[c]Samples collected on 3/1/01 and 12/18/00 are field values; samples collected on 11/2/00 and 1/31/01 are laboratory values.
[d]Alkalinity is assumed to be the acid-neutralizing capacity of the unfiltered water sample.
[e]Dissolved-oxygen concentration measured in sample collected on 2/1/01.

composition between injected and stored waters as represented by the initial recovered water from well CHN-812 and the stored water at observation well CHN-809 just prior to recovery (table 5). The initial recovered water from well CHN-812 had a charge imbalance of about 10 percent between analyzed cations and anions. The charge imbalance, a measure of analytical errors, would be zero if all cations and anions were analyzed exactly. If all of the error in the analysis of recovered water from CHN-812 could be attributed to the calcium concentration, the water composition could be produced, with minimal amounts of cation exchange, by mixing 99 percent injected water and 1 percent brackish water. If the charge-balance error cannot be attributed to calcium, additional minor amounts of calcite and $CO_2$ reaction would be needed to account for the water composition of the recovered water.

Inverse geochemical modeling indicated that mixing, cation exchange, and calcite dissolution were sufficient to account for the evolution in chemical composition from injected water to stored water at well CHN-809 (table 6). The stored water at well CHN-809 can be explained as a mixture of 87 percent injected water and 13 percent brackish aquifer water. In addition, approximately 1.5 mmol/L of calcite was dissolved, and calcium and magnesium were exchanged onto the clays in the aquifer as potassium and sodium were released.

**Table 6.** Mixing fraction and mole transfers that account for the water composition at observation well CHN-809 at the start of the cycle-3 recovery phase during the aquifer storage recovery study in Charleston, South Carolina.

[mmol/L, millimole per liter]

| Solution or reactant | Mixing percentage or mole transfer |
|---|---|
| Injected surface water, percent | 87 |
| Brackish aquifer water, percent | 13 |
| Calcite, mmol/L | 1.5 |
| Calcium exchange, mmol/L[a] | -1.9 |
| Potassium, exchange, mmol/L[a] | .12 |
| Magnesium exchange, mmol/L[a] | -.07 |
| Sodium exchange, mmol/L[a] | 3.9 |

[a]For cation-exchange reactions, positive-mole transfer increases and negative-mole transfer decreases the cation concentration in solution.

# Reactive-Transport Simulations

A small number of chemical reactions appear to account for the major changes in water composition during injection and storage—mixing with brackish aquifer water, calcite dissolution and precipitation, and cation exchange. By assuming that all of the chemical reactions are equilibrium processes, a simple reactive-transport model can be developed to simulate the ASR cycles. The simulator PHAST (Parkhurst and others, 1995; Parkhurst and Kipp, 2002) was used to simulate flow, transport, and chemical reactions in the aquifer. The development of the reactive-transport model requires calibration of flow-model parameters that simulate the water-level altitudes measured during aquifer testing. In addition, other model parameters must be adjusted to adequately simulate the measured breakthrough of conservative chemical constituents at the observation and recovery wells during the ASR cycles. Finally, the chemical reactions and initial conditions in the aquifer must be specified, including the amounts of mineral and exchange sites in the aquifer sediments.

The PHAST simulator does not account for the density effects that can occur by injecting freshwater into an aquifer containing brackish water. Density effects were ignored because the aquifer layers are thin, only 4 m thick, and because the salinity of the brackish water (4,000 mg/L) was less than the 5,000-mg/L suggested threshold for buoyancy stratification (Pyne, 1995; Reese, 2002).

## Flow-Model Parameters

PHAST implements a finite-difference flow model that uses a point distributed grid, such that nodes define the edges of the simulation region. Cell faces are located halfway between nodes in each direction; for cells that border the exterior of the simulation region, cell faces coincide with the nodes. Values of head are calculated for each node in the simulation region for each time step. Aquifer properties (porosity, hydraulic conductivity, and dispersivity among others) are defined by element, which is the volume defined by eight nodes.

Electromagnetic flow logs collected during injection at well CHN-809 delineated two dominant productive zones in the screened sections of the well that were each less than 4 m thick (fig. 7). Formation resistivity logs collected at wells CHN-809 and CHN-810 during ASR cycle 2 injection also showed two productive zones that were about 4 m thick or less containing water with low specific-conductance (fig. 6). Initial flow simulations for parameter estimation included two 2-m-thick productive zones, the centers of which were located at 113.5 and 131.5 m bls. Confining units were assigned above the UPZ, between the UPZ and LPZ (16 m thick), and below the LPZ. The confining zones were discretized with fine node spacing (0.25 m) near the productive zones and coarsening node spacing with distance vertically from the productive zones (fig. 19). The productive zones were represented vertically with three nodes.

Hydraulic conductivity was initially assigned to produce a transmissivity of 37 $m^2$/d (table 3), with the transmissivity of the UPZ about 50 percent greater than that of the LPZ, which is consistent with the transmissivity estimate using the analytical methods discussed earlier and the relative flows observed in the electromagnetic flow logs. In addition, the UPZ and LPZ were divided into two contrasting hydraulic conductivity layers to provide a better simulation of solute transport. The initial hydraulic conductivity estimates were as follows: low hydraulic conductivity layer of LPZ, 1 meter per day (m/d); high hydraulic

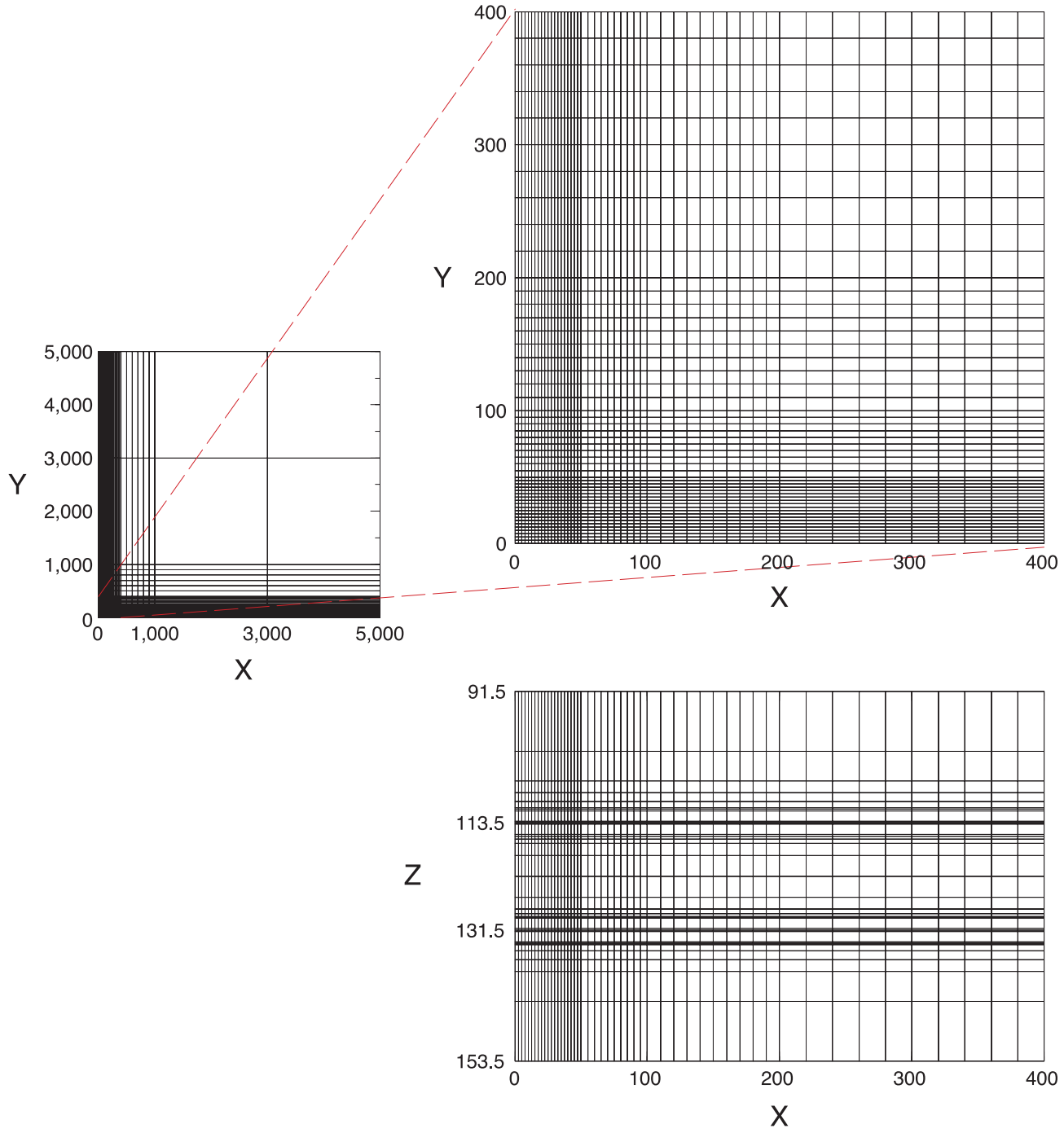


**Figure 19.** Grid used for simulations by which flow-model parameters were estimated from aquifer-test data at the aquifer storage recovery site in Charleston, South Carolina.

conductivity layer of LPZ, 5 m/d; low hydraulic conductivity layer of UPZ, 2.5 m/d; and high hydraulic conductivity layer of UPZ, 10 m/d. Each layer is 2 m thick, which results in transmissivity of 37 $m^2$/d. Because the transmissivity is well constrained by the previous analytical estimation methods, only the hydraulic conductivity of the upper layer of the UPZ (the high hydraulic conductivity layer) was adjusted by parameter estimation to allow less variation in the transmissivity of the combined layers.

The flow system was assumed to be radially symmetric. Simulations were restricted to a single quadrant, the other quadrants being equivalent by symmetry. Pumping and injection rates were decreased to one quarter of the measured or estimated values for the ASR cycles. The restricted simulation region allowed finer grid discretization and faster simulations. The simulation region (fig. 19) was discretized in the X and Y directions with variable but increasing spacing of nodes, ranging from 2.5 m near the pumping well to a maximum of 2,000 m at the margins of the simulation region. The simulation region extended from 0 to 5,000 m in the X and Y directions. The bounding planes at X = 0 m and Y = 0 m were no-flow boundaries because of symmetry; the bounding planes at X = 5,000 m and Y = 5,000 m were assigned leaky boundary conditions. Initial heads for all nodes and the specified heads of the leaky boundary conditions were assigned an arbitrary, uniform value.

Drawdown data for wells CHN-809, CHN-810, CHN-811A, and CHN-800 from the June 1999 and November 2001 aquifer tests were used to fit three flow-model parameters—the hydraulic conductivity of the upper half of the UPZ, the vertical hydraulic conductivity of the confining units, and the specific storage of all layers. Drawdown data were selected for each well at 200; 1,000; 3,000; and 13,360 minutes as representative of the time sequence of drawdown during the 1999 aquifer test. For the 2001 aquifer test, drawdown data were selected for each well at 215; 1,000; 3,000; and 10,500 minutes. The parameter estimation code UCODE (Poeter and Hill, 1999) was used to adjust the flow-model parameters to produce a least-squares fit to the selected drawdown data. A standard deviation of 0.2 m or 0.1 m (last point in the time sequence for each well in each aquifer test) was used to weight the observations for fitting. Because the fits were least sensitive to the vertical hydraulic conductivity of the confining units, this parameter was held constant at a series of values and the other two parameters were fit. Parameters were fit for each aquifer test separately (table 7).

The variation in the fit parameters was small over the entire range of vertical hydraulic conductivity of the confining units. The specific storage ranged from 0.9 x $10^{-5}$ to 1.5 x $10^{-5}$ per meter ($m^{-1}$) and the hydraulic conductivity of the upper half of the UPZ ranged from 6.9 to 9.1 m/d in both aquifer tests (table 7). A vertical hydraulic conductivity for the confining units of 1 x $10^{-6}$ m/d produced the minimum weighted sum of residuals squared ($R^2$) for the 2001 aquifer test, whereas a value of 0.0 m/d marginally produced the smallest $R^2$ for the 1999 aquifer test. The low values of vertical hydraulic conductivity for the confining units indicate that leakage has a relatively small effect on the head distribution during the aquifer tests, which is consistent with the analytical results. A value of 0.0 m/d, which implies no leakage, was selected for use in the reactive-transport simulations. Although leakage certainly occurs in most aquifer systems, ignoring leakage makes very little difference in the goodness of fit to the aquifer test observations or in the reactive-transport simulations. By ignoring leakage, however, it is not necessary to discretize the confining units, which decreases the number of nodes and allows simulations to run faster.

By assuming zero leakage ($K_{z,\ confining}$ = 0.0 m/d), the other parameters for the reactive-transport simulations were selected from the results shown in table 7. The specific storage was 1.5 x $10^{-5}$ $m^{-1}$, which is consistent with the parameter estimation of both aquifer tests, and the hydraulic conductivity of the upper half of the UPZ was selected to be 8.4 m/d, which is the mean of the estimates from the two aquifer tests (table 7). The vertical hydraulic conductivities of the layers of the productive zones were assumed to be one-tenth of the horizontal hydraulic conductivities.

The measured drawdown data and the drawdown simulated by using the selected parameters are shown in figure 20 for the 1999 and 2001 aquifer tests. Oscillations in the measured water levels are due to tidal-loading effects (fig. 20).

**Table 7.** Flow-model parameters fit from the June 1999 and November 2001 aquifer-test data collected at the phase-II aquifer storage recovery site in Charleston, South Carolina.

[$K_{z,\ confining}$, vertical hydraulic conductivity of the confining units; m/d, meter per day; Ss, specific storage; $m^{-1}$, maximum leakage factor; $K_{high,\ UPZ}$, horizontal hydraulic conductivity in the upper half of the upper productive zone; $R^2$, weighted sum of squares of residuals]

| | 2001 aquifer test | | | 1999 aquifer test | | |
|---|---|---|---|---|---|---|
| $K_{z,\ confining}$ (m/d) | Ss ($m^{-1}$) | $K_{high,\ UPZ}$ (m/d) | $R^2$ | Ss ($m^{-1}$) | $K_{high,\ UPZ}$ (m/d) | $R^2$ |
| 1.0x$10^{-5}$ | 0.9x$10^{-5}$ | 6.9 | 109 | 0.9x$10^{-5}$ | 8.2 | 680 |
| 1.0x$10^{-6}$ | 1.2x$10^{-5}$ | 7.4 | 64 | 1.3x$10^{-5}$ | 8.9 | 490 |
| 1.0x$10^{-7}$ | 1.4x$10^{-5}$ | 7.6 | 74 | 1.4x$10^{-5}$ | 9.1 | 459 |
| 0 | 1.5x$10^{-5}$ | 7.6 | 80 | 1.5x$10^{-5}$ | 9.1 | 455 |

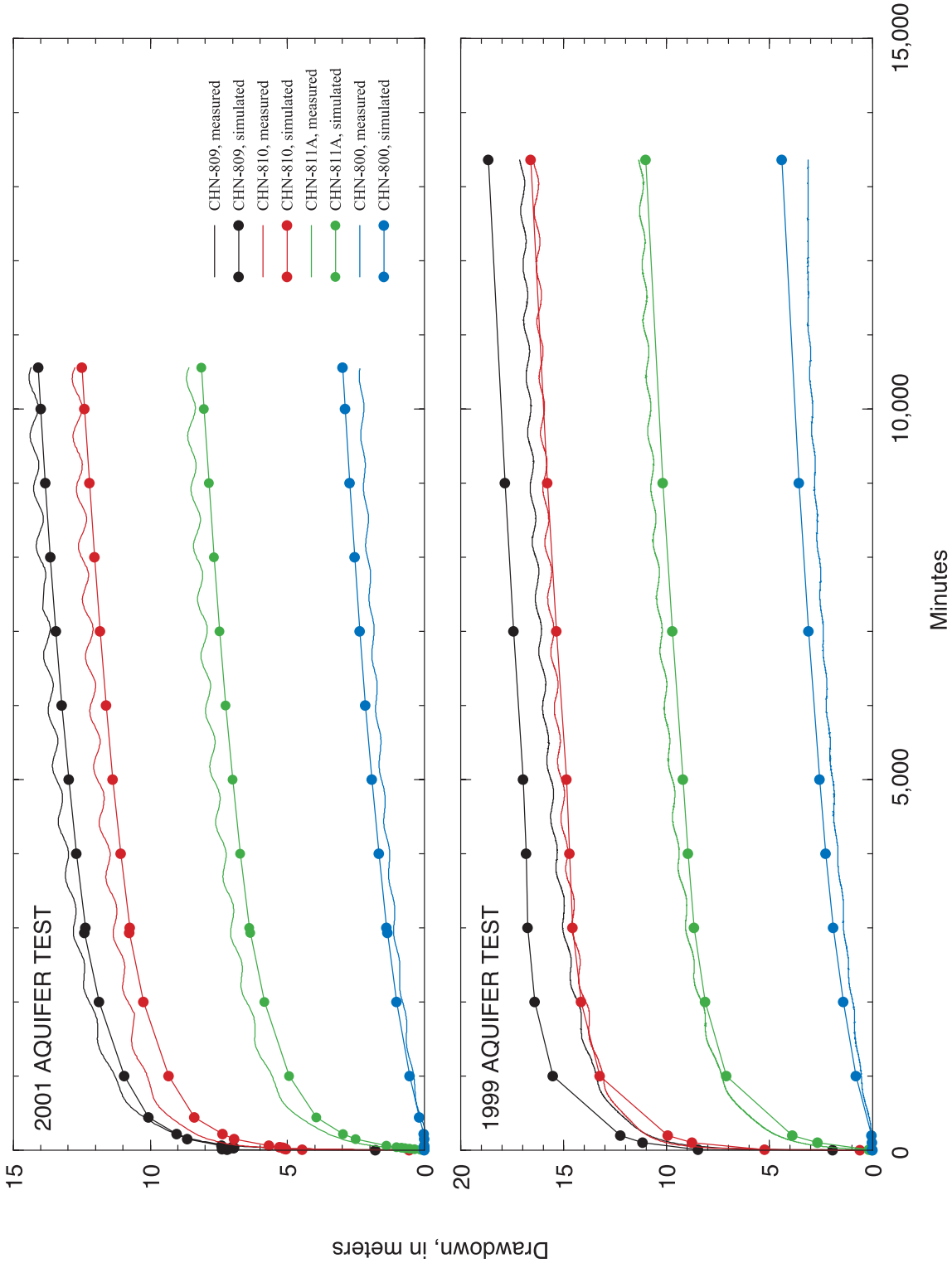


**Figure 20.** Measured drawdown data for November 2001 and June 1999 aquifer tests and simulated drawdowns using the best-fit parameters from the November 2001 aquifer test at the aquifer storage recovery site in Charleston, South Carolina.

The simulated drawdown for the 2001 test differs by less than 1 m for all observations. Results of the 1999 aquifer test are similar except for well CHN-809, which has a difference of more than 1 m between measured and simulated drawdown. The same parameters could not fit simultaneously the drawdown of wells CHN-809 and CHN-810 in the 2001 aquifer test, which were very similar at early times in the test even though the distances to the wells from the pumping well were different (23 m and 37 m, respectively).

Halford and Campbell (2004) reported the potential of large regional anisotropy for the aquifer. However, the fit to the two more distant wells (CHN-811A and CHN-800) and the good correspondence of simulated and measured drawdown at wells CHN-733 and CHN-612 (data not shown), which were not included in the fitting process, are not consistent with a large regional anisotropy. These four wells are located in different directions and as much as 3,000 m distant from the pumping well, conditions that should demonstrate regional anisotropy. In addition, anisotropy cannot explain the inconsistent pattern of drawdown at wells CHN-809 and CHN-810 during the two aquifer tests. The goodness of fit between simulated and measured drawdown for all of the wells justifies the use of horizontal isotropy in all subsequent simulations described in this report.

The water level at well CHN-809 was simulated for the entire period of investigation by using the selected flow-model parameters and the measured and estimated injection and pumping rates. The good match with observations over the entire period of investigation (fig. 21) gives credence to the adequacy of the flow simulations and to the injection rates that had to be estimated. Similar correspondence between measured and simulated water-level altitudes was observed for well CHN-810 (not shown).

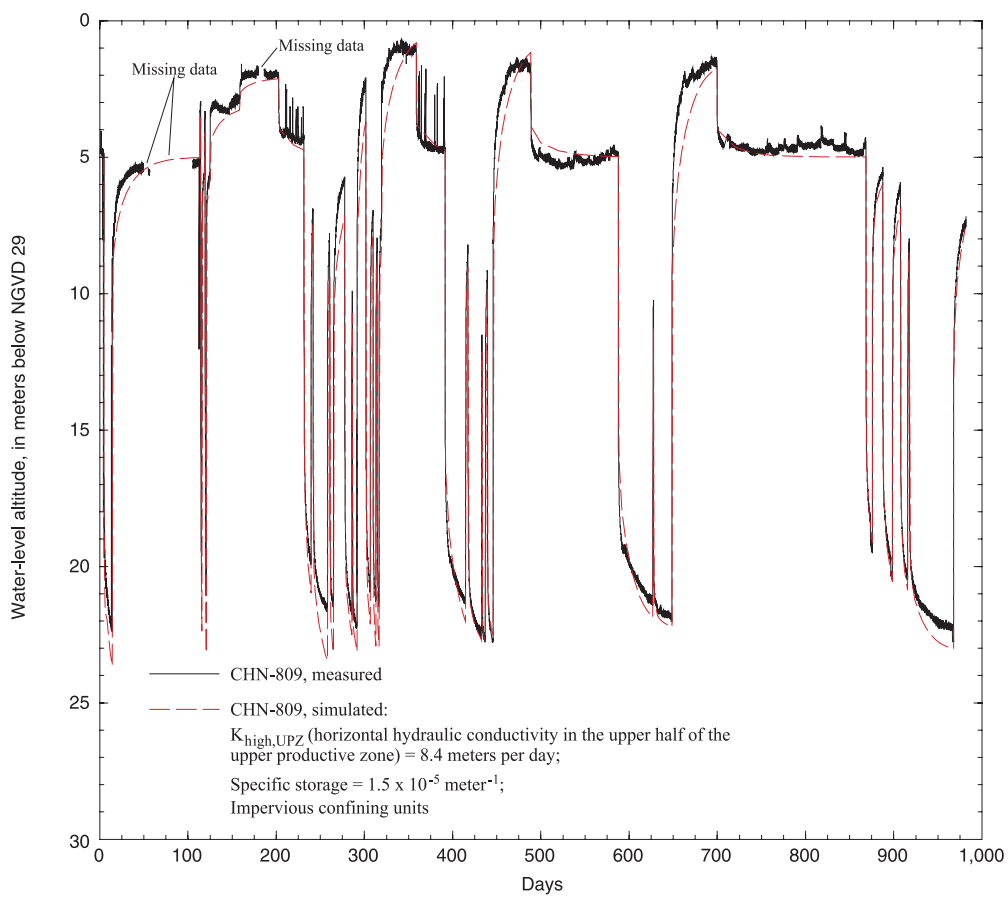


**Figure 21.** Measured water-level altitude in observation well CHN-809 and simulated water-level altitude calculated by using the selected flow-model parameters for the period of investigation at the aquifer storage recovery site in Charleston, South Carolina.

## Transport Simulations

The grid for reactive-transport simulations was modified from the grid used in estimating the flow-model parameters by limiting the extent to 3,000 m and removing some of the nodes in the range of 200 to 400 m in the X and Y directions. The smaller number of nodes was chosen to speed up the overall simulation time, while preserving the fine grid in the vicinity of the production and observation wells. Discretization in the Z direction, boundary conditions, and flow-related parameters were unchanged.

Although the drawdown at multiple wells is adequately described by simulations using the selected values of hydraulic conductivity and storage, breakthrough of chemical constituents at observation wells places even greater constraints on flow and transport model parameters, particularly on the thickness and porosity of the productive zones and the dispersivity of the system. Chloride is a conservative constituent, and the large contrast between the brackish aquifer water and the injected freshwater provides a good tracer of the movement of injected water into the aquifer.

The chloride concentrations at observation well CHN-809 during ASR cycle 3 are shown in figure 22. Given the relatively low injection rate, 0.11 cubic meter per minute ($m^3$/min), which is equivalent to 30 gallons per minute (gal/min), the rapid breakthrough of water with low chloride concentrations at the observation well (23 m from the injection well) indicates that the injected water apparently flows through a small volume of aquifer. The effective volume through which water flows between the injection well and the observation well can be estimated from the cumulative volume of water injected up to the time when chloride concentration at the observation well was approximately half of the concentration in the brackish aquifer water (850 mg/L). For ASR cycle 3, the breakthrough of this concentration occurred approximately 8 days after the start of the injection. The cumulative volume of water injected after 8 days was approximately 1,300 cubic meters ($m^3$). The volume of the aquifer filled by this volume of injected water was calculated by using the following equation:

$$V = \theta h(\pi r^2), \tag{1}$$

where

- $V$ is the volume of injected water, in cubic meters;
- $\theta$ is the porosity of the aquifer, unitless;
- $h$ is the thickness of the aquifer, in meters;
- $\pi$ is approximately 3.14; and
- $r$ is the distance from the injection well to the observation well, 23 m.

Solving for the product of depth times porosity gives $\theta h$ = 0.8 m. Thus, for reasonable values of porosity, 0.4 to 0.1, the thickness of the aquifer must be approximately 2 to 8 m, which is consistent with the relatively thin zones for which significant flow and low specific conductance were indicated in the well logs.

During storage, the minimum concentration of chloride observed at well CHN-809 was approximately 200 mg/L (fig. 22), which differed from the injection concentration of approximately 40 mg/L. Thus, although a large proportion of the water at the observation well during storage in ASR cycle 3 was injected water, a substantial proportion (approximately 10 percent) was brackish aquifer water.

The parameter in the reactive-transport simulations that controls the amount of mixing between injected water and brackish aquifer water is the dispersivity. Initial transport calculations included two 2-m productive zones with flow-model parameters from the analytical methods for aquifer tests. The porosity and dispersivity were adjusted to match the chloride concentrations at observation wells CHN-809 and CHN-810. A dispersivity of about 15 m and porosity of 0.2 resulted in an adequate fit for ASR cycle 3 for the observation wells. However, the simulated chloride concentration for the recovery well (CHN-812) during storage was about 120 mg/L, which is three times greater than the measured chloride concentration in the initial recovered water. Thus, simulations with a 15-m dispersivity failed to reproduce the relatively low chloride concentrations near the pumping well, which is a critical failure with respect to the simulation of recoverable potable water.

A dispersivity of 15 m is quite large relative to the distances to the observation wells (23 m and 37 m). Frequently, the dispersivity is approximately 10 percent of the length scale (Appelo and Postma, 1993, p. 362–363), which would be about 3 m in this case. A large dispersivity can be considered as a lack of knowledge about individual flow paths within an aquifer (Appelo and Postma, 1993). Additional flow paths would result if each productive zone were divided into more layers. To allow for a wider range of

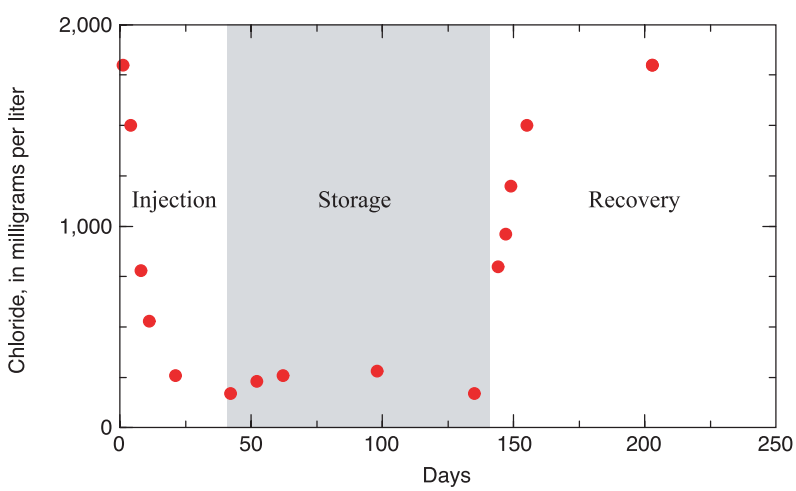


**Figure 22.** Measured chloride concentrations at observation well CHN-809 during injection, storage, and recovery in cycle 3 of the aquifer storage recovery study in Charleston, South Carolina.

possibilities for simulating transport in the aquifer, two productive zones containing two 2-m layers each, were used for estimating the flow-model parameters and in all simulations of reactive transport. As discussed earlier, leakage was assumed to be negligible. A schematic for the aquifer geometry used in reactive-transport simulations is shown in figure 23. The water composition in a well that penetrated a productive zone was calculated as a mixture of waters from the model layers that represented the productive zone. The mixture of waters was determined by the well flow rates for the layers of the productive zone.

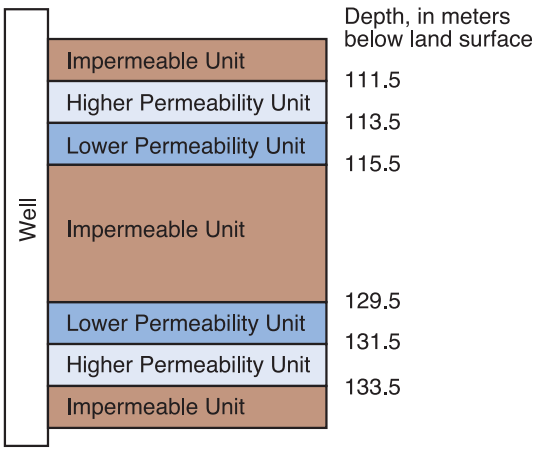


**Figure 23.** Schematic diagram showing aquifer geometry used to simulate chloride transport for cycle 3 of the aquifer storage recovery study in Charleston, South Carolina.

Porosity and dispersivity for the four model layers were adjusted by trial and error to account for the time sequence during ASR cycle 3 of chloride concentrations. Observations included data for observation wells CHN-809 and CHN-810 throughout ASR cycle 3 and for pumping well CHN-812 during recovery. The set of hydraulic and transport parameters for the four aquifer layers are given in table 8. The transmissivity calculated from the horizontal hydraulic conductivities and layer thicknesses is 33.8 $m^2$/d, which is similar to the transmissivity estimated by fitting the drawdown data with analytical methods. The longitudinal dispersivity is 0.5 m for all layers; transverse dispersivity is 0.05 m.

The transport simulations for chloride include dispersion that results from the specified dispersivity (0.5 m) and also numerical dispersion that results from time and space discretization for the finite difference approximations. When using upstream in space and backward in time weighting for the finite difference equations, the numerical dispersivity can be approximated by the formula:

$$D = \Delta X/2 + V\Delta t/2, \qquad (2)$$

where

$D$ is the numerical dispersivity, in meters;
$\Delta X$ is the node spacing, in meters;
$V$ is the velocity in the X direction, in meters per day; and
$\Delta t$ is the time step for the simulation, in days.

In the reactive-transport simulations $\Delta X$ was 2.5 m and $\Delta t$ was 0.1 day during the injection period. The simulated velocity was large near the injection well, approximately 30 m/d during injection, but decreased with distance from the injection well. Thus, numerical dispersivity was approximately 3 m at the pumping well during injection, but approached 1.5 m with distance from the injection well. During recovery, the simulated velocity was approximately 125 m/d at the pumping well, but a time step of 0.01 day was used for the initial 10 days of recovery, which limited the range of the numerical dispersion from 1.5 to 2.5 m. The specified dispersivity for the simulations was 0.5 m. Except near the injection and pumping well, the effective dispersivity was about 2 m (sum of specified and numerical dispersivities).

The simulated chloride concentrations for the three wells, computed by using the adjusted porosities and dispersivity, match the measured chloride concentrations adequately (fig. 24). Simulated chloride concentration during storage at well CHN-812 was 23 mg/L, which is lower than the first measured concentration during recovery (40 mg/L). However, a lower simulated value for storage is not necessarily inconsistent

**Table 8.** Flow and transport model parameters selected for reactive-transport simulations during the aquifer storage recovery study in Charleston, South Carolina.

[UPZ, upper productive zone; LPZ, lower productive zone]

| Layer | Thickness (meter) | Specific storage (1/meter) | Horizontal hydraulic conductivity (meter per day) | Vertical hydraulic conductivity (meter per day) | Porosity (unitless) | Longitudinal dispersivity (meter) |
|---|---|---|---|---|---|---|
| UPZ, upper layer | 2 | $1.5\times10^{-5}$ | 8.4 | 0.84 | 0.1 | 0.5 |
| UPZ, lower layer | 2 | $1.5\times10^{-5}$ | 2.5 | .25 | .3 | .5 |
| LPZ, upper layer | 2 | $1.5\times10^{-5}$ | 1 | .1 | .3 | .5 |
| LPZ, lower layer | 2 | $1.5\times10^{-5}$ | 5 | .5 | .1 | .5 |

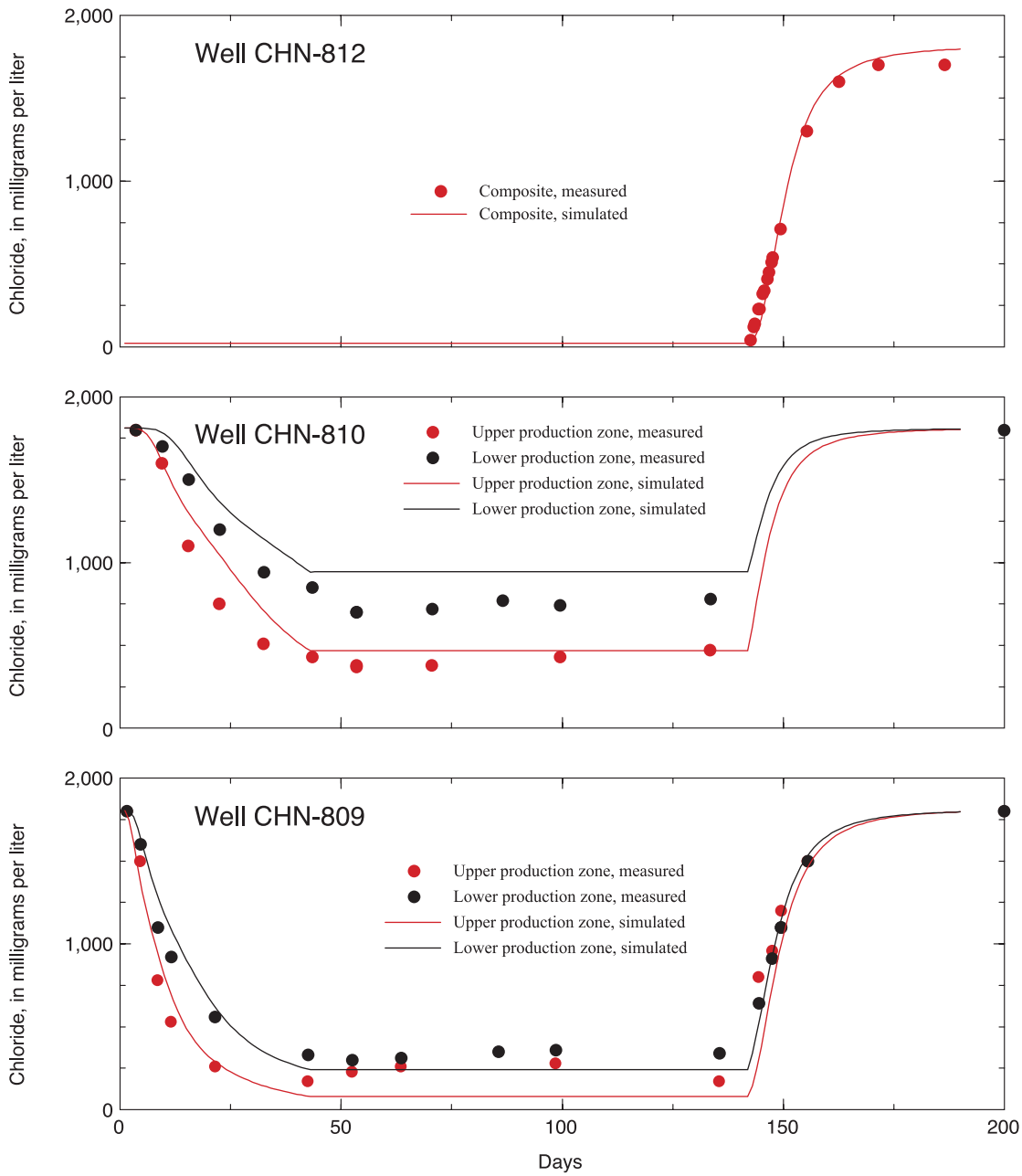


**Figure 24.** Measured chloride concentrations and chloride concentrations simulated by using two 2-meter layers for each production zone and adjusted porosity and hydraulic conductivity during the aquifer storage recovery study in Charleston, South Carolina.

with the measured concentration because the first sample was not taken immediately upon initiation of pumping.

There is no assurance that the selected flow and transport model parameters uniquely produce the goodness of fit for chloride concentrations that are shown in figure 24. Other parameter options that include different layer thicknesses, porosities, hydraulic conductivities, and dispersivities may produce equally good or better simulations. However, the adjusted parameters are plausible—the thicknesses of the productive zones are consistent with well logs; the UPZ has a higher transmissivity, as indicated by well flow logs; and horizontal hydraulic conductivity, porosity, and specific storage are reasonable for a Coastal Plain aquifer, although porosity in the higher hydraulic conductivity layers (0.1) is somewhat less than expected. The porosity could be increased if the thickness of the higher hydraulic conductivity layers were decreased. Simulations using the selected flow and transport model parameters adequately match the drawdown data of the aquifer tests, the chloride concentration data for the observation wells, and the chloride concentration data for the recovered water.

These flow and transport model parameters are assumed to adequately describe flow and transport in the SL/BM aquifer and are used to investigate geochemical reactions and the response of the aquifer to long-term injection and storage of water.

## Chemical Reactions

Results of inverse geochemical modeling indicate that mixing (dispersion), cation exchange, and calcite dissolution and precipitation account for the evolution of water chemistry during the ASR investigation. For reactive-transport simulations, the simplest assumption is that the cation-exchange and calcite reactions are equilibrium controlled; that is, the kinetics of these reactions are sufficiently rapid to attain equilibrium. For the reactive-transport simulations, the equilibrium constants for calcite and cation exchange were taken from the *phreeqc.dat* database (Parkhurst and Appelo, 1999); all aqueous species and equilibrium constants also were taken from *phreeqc.dat*.

For the reactive-transport simulations, data are needed for the number of moles of calcite and exchange sites per liter of water. For calcite, the minimum abundance from table 1 is 19 percent, or 0.19 kilogram (kg) of calcite per kilogram of sediment. Assuming a porosity of 0.3, a density of the solid sediment of 2.7 kg/L, and a formula weight of 100 grams per mole (g/mol) for calcite, the number of moles of calcite per liter of water is (0.19 x 1,000 /100) x 2.7 x 0.7/0.3 = 12 moles per liter (mol/L) water. This amount is sufficient to maintain calcite equilibrium for hundreds of pore volumes, and the simulations are insensitive to the exact abundance of calcite. An estimate of the cation exchange capacity (CEC) is 22 meq/100 g sediment (table 1). By using the same porosity and density, the number of exchange sites is estimated to be (22/1,000/100 x 1,000) x 2.7 x 0.7/0.3 = 1.4 equivalents per liter (eq/L) of water. However, the CEC varied from 0 to 36 meq/100 g in samples from the aquifer, and simulations are sensitive to the number of exchange sites. The number of exchange sites per liter of water was used as an adjustable parameter to produce a match between simulated and measured cation concentrations.

For all simulations, the aquifer was initially filled with brackish aquifer water (table 5). Water entering the simulation region from the boundaries also had the composition of the brackish aquifer water. Injected water had the composition defined in table 5. ASR cycle 3 was simulated with 43 days of injection at a rate of 0.11 $m^3$/min, 99 days of storage, and 48 days (actual was 61 days) of recovery at a pumping rate of 0.456 $m^3$/min (7.57 L/s). Because one quadrant of the aquifer was simulated, injection and pumping rates were reduced by a factor of four for the simulations.

The concentration of exchange sites that produced the best fit for all constituents was approximately 0.1 meq/L water. Simulation results for wells CHN-812 and CHN-809 are shown in figures 25 and 26, respectively.

If the concentration of exchange sites was an order of magnitude higher (1.0 eq/L water), the simulated calcium concentration at the start of recovery at well CHN-812 (approximately 10 mg/L) was lower than the measured concentrations (approximately 25 mg/L). Some of the difference in calcium concentration at the pumping well (CHN-812) also may have been caused by disequilibrium between solution and exchange sites in the rapid-flow environment near the pumping well. For well CHN-809, if the concentration of exchange sites was an order of magnitude lower (0.01 eq/L water), the simulated calcium concentrations during storage were too high (approximately 10 mg/L compared with less than 5 mg/L measured), the simulated alkalinity was too low (100 to 200 mg/L compared with 300 mg/L measured), and the simulated pH was too low (approximately 8.0 compared with 9 to 10 measured). In simulation results with the lower concentration of exchange sites (0.01 eq/L water), calcium concentrations had begun to increase at observation well CHN-809. However, in ASR cycle 3, no breakthrough of calcium at observation well CHN-809 was observed, which implies that the number of exchange sites must be greater than 0.01 eq/L water. During the course of a longer injection, observation of a calcium breakthrough of approximately 20 mg/L (fig. 27), would allow a more accurate estimation of the number of exchange sites in the productive zones and would help to discriminate between concentration of exchange sites and kinetic effects near the pumping well.

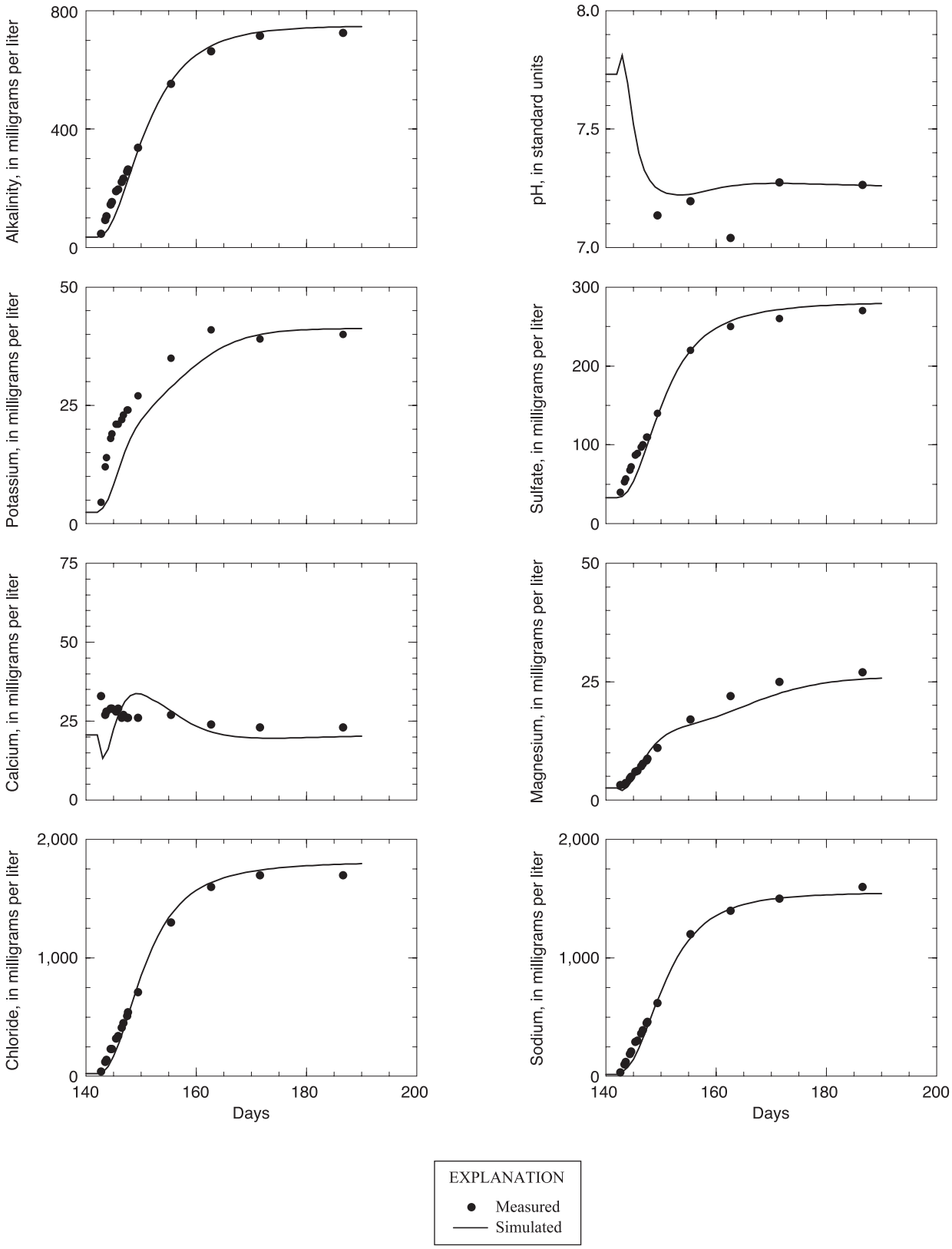


**Figure 25.** Measured concentrations of major ions and pH for production well CHN-812 during cycle 3, and concentrations simulated by using the adjusted flow and transport parameters and a value of 0.1 equivalent per liter of water for the concentration of exchange sites during the aquifer storage recovery study in Charleston, South Carolina.

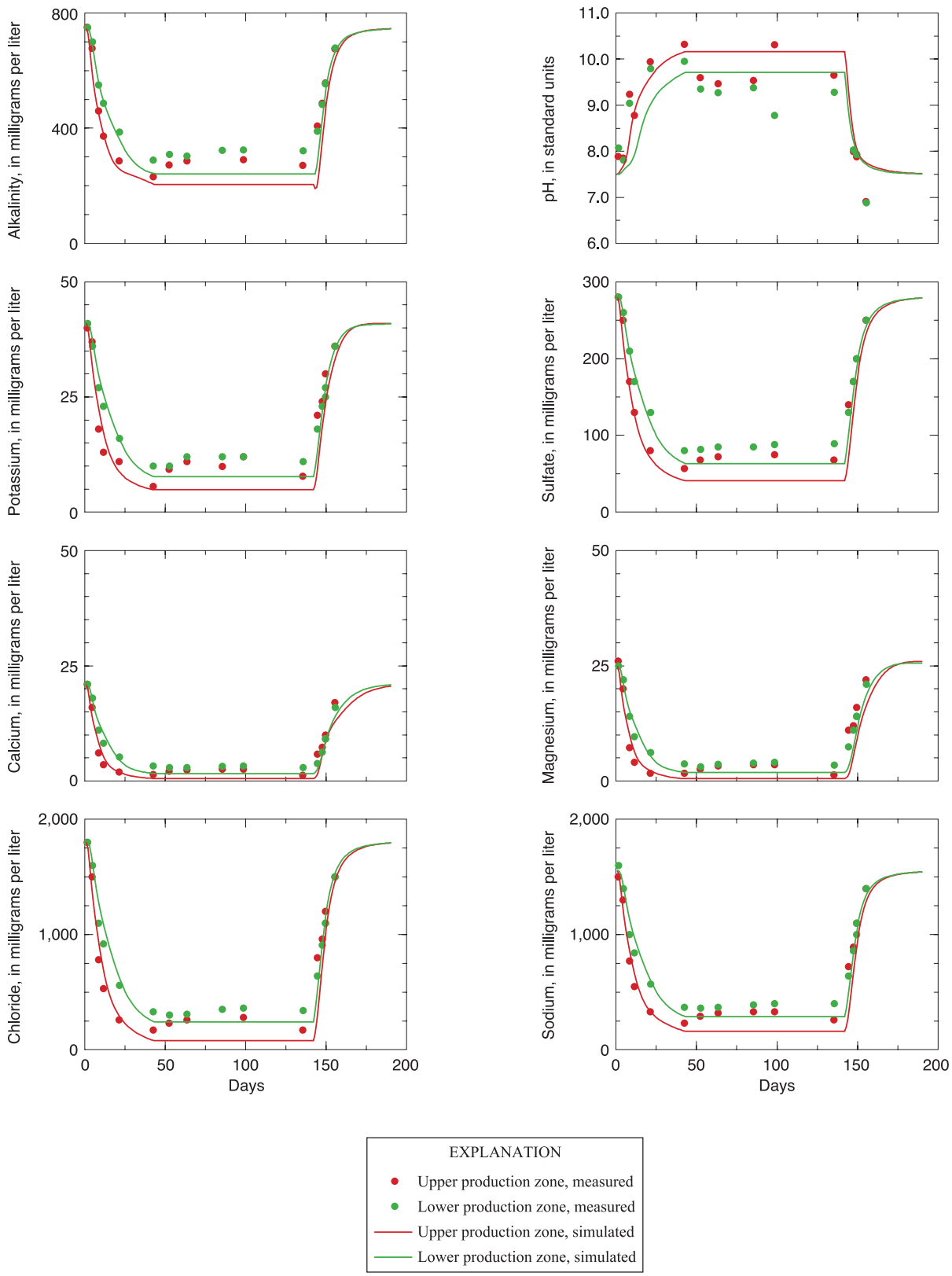


**Figure 26.** Measured concentrations of major ions and pH for observation well CHN-809 during cycle 3 and concentrations simulated by using the adjusted flow and transport parameters and a value of 0.1 equivalent per liter of water for the concentration of exchange sites during the aquifer storage recovery study in Charleston, South Carolina.

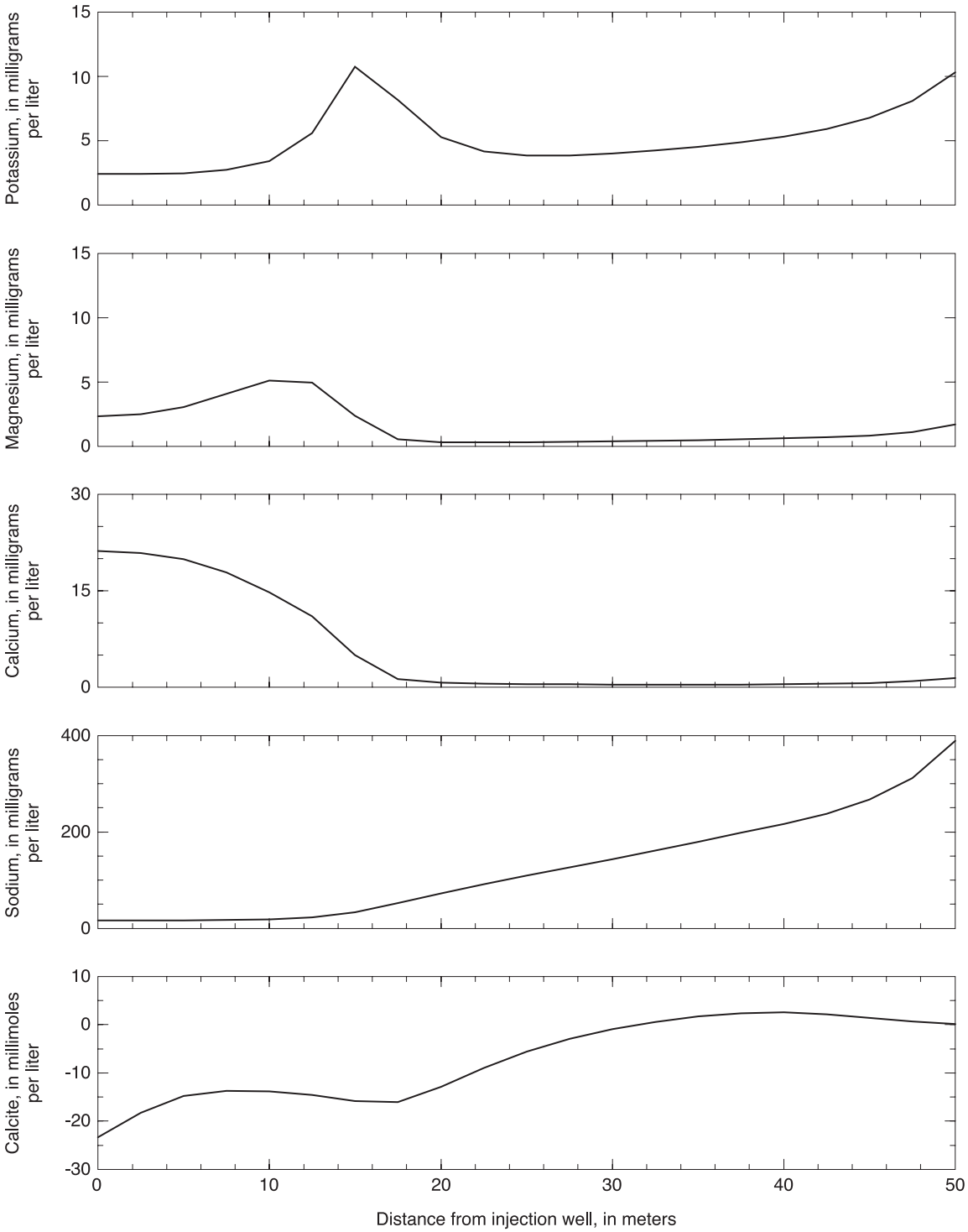


**Figure 27.** Simulated concentrations of cations and amount of calcite reacted with distance from production well CHN-812 at the end of the cycle-3 injection period during the aquifer storage recovery study in Charleston, South Carolina.

## Aquifer Storage Recovery

The calibrated reactive-transport model was used to assess the effectiveness of long-term storage and recovery by simulating the amount of potable water that could be recovered for varying lengths of injection. The model also was used to estimate the effect on porosity caused by the dissolution and precipitation of calcite during ASR. Finally, an additional simulation was performed to determine the effect of a regional hydraulic gradient on the movement of the injected water.

### Recoverable Potable Water

To estimate the amount of potable water that could be recovered with extended injection periods, four simulations were performed using durations of 1, 2, 4, and 8 times the duration of injection for ASR cycle 3. In the longest simulation, water was injected for approximately 1 year (344 days). In all simulations, water was injected at 0.11 $m^3$/min (1.89 L/s) and recovered at a pumping rate of 0.45 $m^3$/min (7.57 L/s). Potable water was defined as water with a chloride concentration less than 250 mg/L. Simulated storage times were variable, but the length of the storage period had no effect on the simulated recoveries. The chloride concentrations during recovery in the four simulations are shown in figure 28. For successively longer injection periods, the duration of recovery of potable water increases from 3.5 to 28 days. The length of time and the volume of potable water that can be recovered are essentially linear with the length of injection. In all cases, the recovery efficiency is approximately 0.3; that is, 30 percent of the volume of injected water can be recovered before chloride concentrations exceed 250 mg/L. In the simulation using the 344-day injection period, approximately 56,000 $m^3$ (15 million gallons [Mgal]) of water were injected and 18,000 $m^3$ (4.8 Mgal) of potable water were recovered. Simulated results

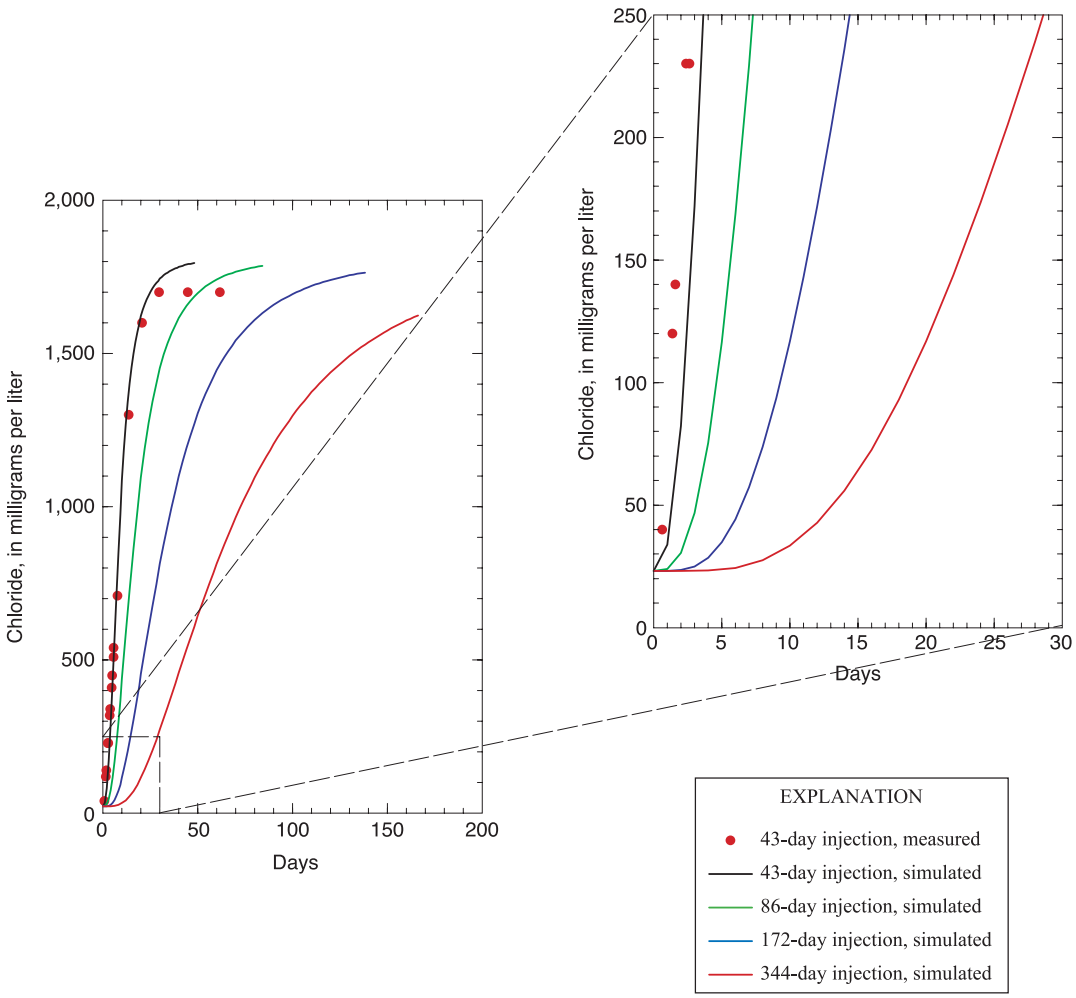


**Figure 28.** Measured chloride concentrations for cycle 3 and simulated chloride concentrations during recovery following injection periods 1, 2, 4, and 8 times the 43-day injection for cycle 3 during the aquifer storage recovery study in Charleston, South Carolina.

compare very well with average recovery efficiencies determined throughout this investigation (21 to 34 percent). The 43-day simulation appears to overestimate the volume of potable water compared to the measured chloride concentrations for ASR cycle 3 (fig. 28). A more conservative recovery efficiency is 25 percent. The observations and simulations indicate that at least four times the target volume of water needs to be injected to be able to successfully recover the target volume of potable water.

## Effects of Injection on Porosity

Simulations assuming equilibrium indicate that calcite dissolves and precipitates in response to a complex series of cation-exchange reactions. The amount of calcite that reacted (negative indicates dissolution and positive indicates precipitation) in the high hydraulic conductivity layer of the UPZ during the simulation of the ASR cycle-3 injection is shown in figure 27. In the simulation, calcite dissolved to the greatest extent in the vicinity of the injection well and precipitated between 30 and 50 m from the well. Calcite dissolution (negative value, fig. 27) is greatest at the injection well (X = 0 m). Calcite dissolution is also greater at 18 m than at any other distance between 6 m and 50 m from the injection well. This pattern of calcite reaction is caused by an injection water that is undersaturated with respect to calcite and cation exchange reactions in the aquifer. Thus, the injected water dissolves calcite near the injection well. However, the calcium, derived from the injected water and dissolution of calcite, interacts with cations exchanged on the clays (and zeolites) of the aquifer. Cation exchange causes an uptake of calcium, and a release of sodium, magnesium, and potassium. The simulations indicate a potassium peak at about 15 m from the injection well after the 43-day injection (fig. 27). A magnesium peak at about 12 m follows the potassium peak, which in turn is followed by high calcium concentrations. Dissolution and precipitation of calcite respond to these cation interactions with the sediments; where calcium is exchanged for magnesium and potassium (approximately 12 to 18 m), calcite tends to dissolve. Calcite precipitated between 30 and 50 m from the injection well, which is the beginning of the mixing zone between the injected water and the brackish aquifer water. Thus, there are cation concentration fronts and calcite dissolution and precipitation fronts that are predicted to proceed through the aquifer as injection of water proceeds.

The volume of calcite, in millimoles per liter, that dissolved in water is minor (fig. 27). Given the porosity of 10 percent for the high hydraulic conductivity layer that was used in the simulations, the change in porosity caused by dissolution of 25 mmol/L calcite is approximately 0.01 percent. Similarly, the amount of calcite precipitated in the calcite precipitation front causes a negligible decrease in porosity. Simulated results indicate that the maximum porosity increase after 100 years of injection is 0.1 percent in cells in the vicinity of the injection well.

## Effects of a Regional Hydraulic Gradient

One additional simulation was performed to investigate the effects of a regional hydraulic gradient on storage and recovery. From the potentiometric surface contours given in Campbell and others (1997), a regional hydraulic gradient toward pumping centers in the northern part of the peninsula is approximately 0.0005 meter per meter (m/m).

A second model was developed that applied the regional hydraulic gradient as boundary conditions such that the potentiometric surface of the initial condition sloped downward from southeast to northwest. Leaky boundary conditions imposed this regional hydraulic gradient. The simulation region was centered on production well CHN-812 (fig. 29) and was discretized in the X and Y directions with variable but increasing spacing of nodes. Grid discretization ranged from 5 m near well CHN-812 to a maximum of 500 m at the margins of the simulation region and represented an area of 10,000 m by 10,000 m. Vertical layering was discretized into two productive zones containing two 2-m layers each (fig. 23). Flow and transport model parameters for this model were equal to those of the reactive-transport model previously discussed (table 8).

A 100-year simulation of constant injection at 0.11 $m^3$/min (30 gal/min) was performed to investigate the short- and long-term effects of the regional hydraulic gradient. The following discussion considers the most hydraulically conductive layer in the simulation; other layers with lower hydraulic conductivity will have smaller extents of freshwater and lower flow velocities.

After 1 year of simulation, the freshwater (based on the location of the 250-mg/L chloride contour) migrated approximately 100 m from the injection well and was essentially circular (fig. 29). The steep gradients from the injection well to the aquifer caused radial flow and completely overwhelmed any effect from the relatively shallow regional hydraulic gradient. After 10 years the freshwater migrated approximately 300 m from the injection well, and the shape of the 250-mg/L chloride contour was only slightly asymmetric. By 100 years, the simulated 250-mg/L chloride contour was substantially asymmetrical around the injection well, which was caused by the regional hydraulic gradient; the freshwater contour extended 800 m from the injection well upgradient and 1,200 m downgradient.

The simulated steady-state velocity configuration for the specified injection rate, flow-model parameters, and imposed regional hydraulic gradient, which was achieved in less than 1 year, had a stagnation point (location where flow velocities are zero) about 1,500 m upgradient from the injection well. Freshwater will never advance past the stagnation point, regardless of the length of injection, and the maximum extent of the 250 mg/L contour will be less than 1,500 m (800 m is the maximum extent in 100 years). The extent of freshwater flow in the downgradient direction was limited only by the length of the injection period. For practical time periods (decades) and reasonable injection rates (0.11 $m^3$/min), the maximum extent

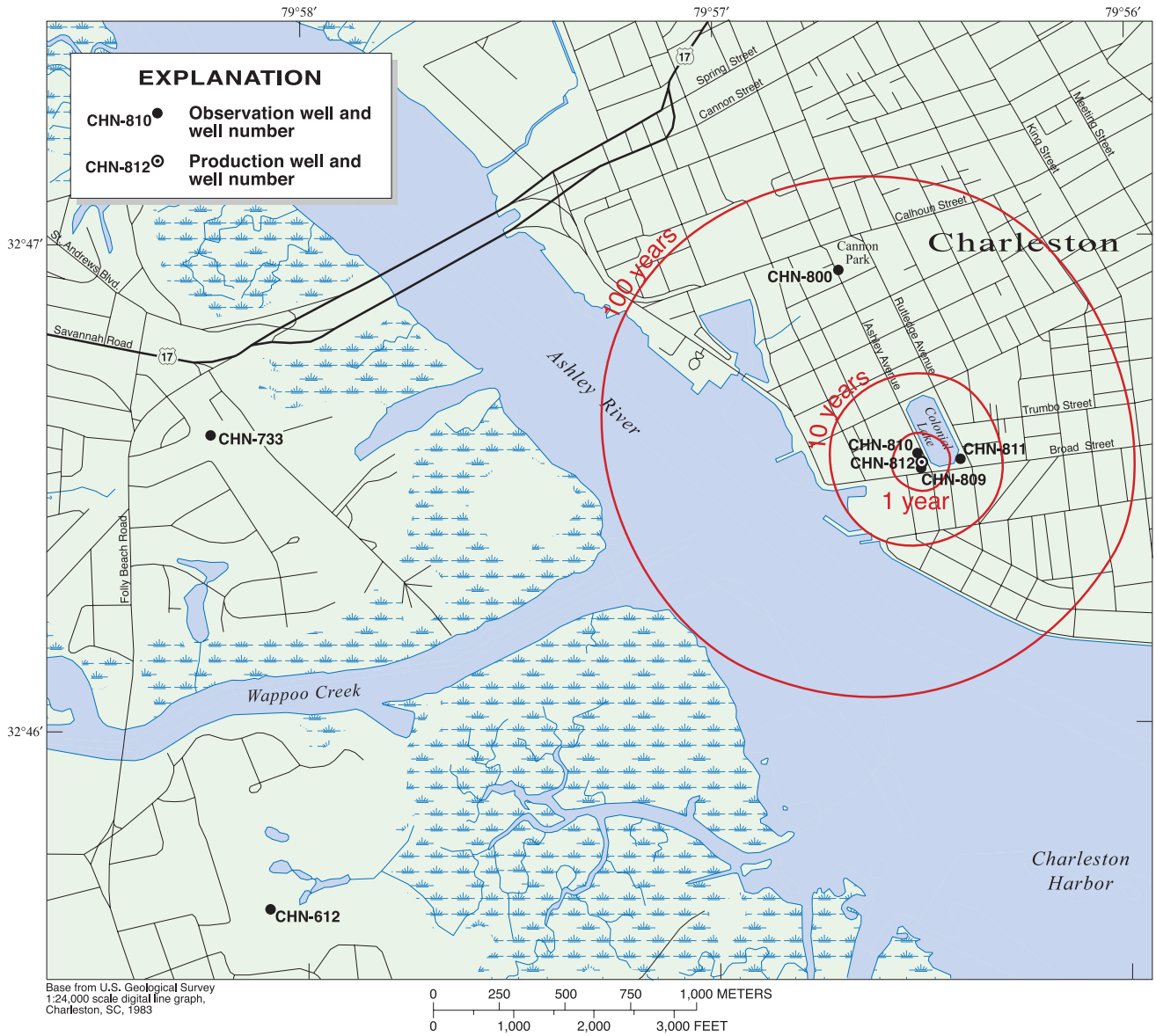


**Figure 29.** Extent of freshwater migration after 1, 10, and 100 years of injection at the aquifer storage recovery site, Charleston, South Carolina.

of the freshwater bubble from a single injection was less than 1 km and nearly symmetrical.

If injection ceases, the regional hydraulic gradient will be reestablished rapidly and the movement of the freshwater bubble can be estimated from Darcy's law:

$$v = K\,(dh/dl)/\theta\ , \qquad (3)$$

where

$v$ is the interstitial velocity, in meters per day;
$K$ is the hydraulic conductivity, in meters per day;
$dh/dl$ is the hydraulic gradient, in meters per meter; and
$\theta$ is the porosity of the aquifer, unitless.

Using $K$ = 8.4 m/d, $dh/dl$ = 0.0005 m/m, and $\theta$ = 0.1, groundwater velocity resulting from the regional hydraulic gradient is about 0.04 m/d or 15 meters per year (m/yr).

After development of the regional hydraulic gradient model, input flow and transport parameters were varied to evaluate the sensitivity of the location of the simulated 250-mg/L chloride contour to four model inputs. The simulated distance from the injection well to the 250-mg/L chloride contour increases as the porosity used in the model is decreased because a larger volume of aquifer is needed to contain the same volume of injected water. The simulated distance to the 250 mg/L contour is maximized when the dispersivity used in the model is zero, and is decreased as dispersivity is increased. The simulated distance to the 250 mg/L contour is independent

of the hydraulic conductivity provided the ratios of hydraulic conductivity are preserved among all the layers of the model. If the ratio of hydraulic conductivity between two layers is changed, the location of the contour will depend on the relative proportion of water transmitted by the layers and the porosities of the layers. The 250 mg/L contour is insensitive to the specific storage used in the model.

# Summary

The hydrologic and geochemical effects of aquifer storage recovery were evaluated to determine the potential for supplying the city of Charleston, South Carolina, with large quantities of potable water during emergencies. Estimates presented in a 2001 South Carolina Emergency Preparedness Division report are that 80 percent of the homes in the Charleston area would be without water for weeks to months because of damaged water pipes following an earthquake of similar magnitude (7.3) to the 1886 Charleston earthquake. To evaluate the feasibility of aquifer storage recovery as a source of drinking water after an earthquake, hurricane, or hard freeze, an aquifer storage recovery system, including a production well and three observation wells, was installed at a site located on the Charleston peninsula. The focus of this study was the 23.2-meter thick Tertiary limestone and sand aquifer of the Santee Limestone and the Black Mingo Group.

The investigation included geophysical logging of wells, aquifer testing, monitoring long-term water levels and water-quality characteristics, and evaluating water-quality data by using geochemical models. Geophysical logging was used to investigate the hydrogeologic features of the Santee Limestone/Black Mingo aquifer and to evaluate the transport of treated surface water within the aquifer. Two aquifer tests were completed to determine the Santee Limestone/Black Mingo aquifer properties surrounding the production well and to determine if the properties changed with successive aquifer storage recovery cycles. Long-term water-level monitoring and analyses of water-quality samples during four injection, storage, and recovery cycles were used to evaluate the trends in injection and withdrawal rates, recovery efficiency, and long-term storage effects on the injectant water quality. Finally, geochemical modeling was used to determine the dominant chemical reactions and hydraulic processes that affect the injectant water quality during the aquifer storage recovery tests.

Analysis of six sediment samples collected from a cored well located near the aquifer storage recovery site showed that quartz and calcite are the dominant minerals in the Santee Limestone/Black Mingo aquifer. Opal, clinoptilolite, ferruginous illite, and smectite were present in quantities of more than 7 percent by weight in the lower production zone and lower permeable zone located between the upper and lower production zones. Ankerite, a calcium carbonate and member of the dolomite group, was present only in the lower permeable section and composed 9 percent of the sample by weight. Estimated cation exchange capacity ranged from 12 to 36 milliequivalents per 100 grams in the lower section of the aquifer.

Formation resistivity logs, borehole electromagnetic flow-meter logs, and specific-conductance profiles were collected in the observation wells to delineate the dominant production zones of the Santee Limestone/Black Mingo aquifer. All three methods delineated a 2- to 3- meter thick upper production zone at depths from about 111 and 115 meters below land surface. Formation resistivity and electromagnetic flow-meter logs delineated at least a 3- to 4-meter thick lower production zone at depths from about 128 and 134 meters below land surface. Borehole electromagnetic flow-meter results while injecting into well CHN-809 indicated that the upper, middle, and lower screen sections received total flow of 58, 1, and 41 percent, respectively.

Two aquifer tests were conducted during this investigation to determine the hydraulic characteristics of the Santee Limestone/Black Mingo aquifer and if the characteristics changed as a result of aquifer storage recovery testing. Analytical results of the two aquifer tests indicated that aquifer properties did not change as a result of aquifer storage recovery cycling. Transmissivity values of the two tests were equal at 37 meters squared per day. The average storage coefficient of the Santee Limestone/Black Mingo aquifer was $3 \times 10^{-5}$. The leakage coefficient of the confining unit was estimated to be less than $7 \times 10^{-4}$ meter$^{-1}$; however, analytical results of tests that did not allow leakage produced a sum of squares error similar to the original results, indicating that leakage into the Santee Limestone/Black Mingo aquifer is minimal. Differences in the storage coefficient and the maximum leakage values for the two aquifer tests are minimal and could be due to the insensitivity of the analytical solution to the leakage factor. The consistent results between the two aquifer tests indicate that physical changes to the Santee Limestone/Black Mingo aquifer because of calcite dissolution or other geochemical reactions have little effect on the hydraulic properties of the Santee Limestone/Black Mingo aquifer.

Specific-capacity data from the June 1999 (0.12 liter per second per meter of drawdown) and November 2001 (0.092 liter per second per meter of drawdown) aquifer tests indicate that the overall specific capacity decreased over time because of residual plugging that occurred within the aquifer or well screen. Additional evidence of plugging is the decrease in injection rates for aquifer storage recovery cycles 2 through 4, even though the same average injection pressure was maintained at the production well. Specific-capacity values improved following inadvertent pump failure during the cycle-4 recovery phase, which indicates that periodic redevelopment during the injection cycles could limit the reduction in specific capacity during aquifer storage recovery implementation at this site.

Four cycles of injection, storage, and recovery were conducted between October 1999 and February 2002. Each

cycle consisted of injecting between 6.90 and 7.19 million liters of water for storage periods of 1, 3, or 6 months. Average injection rates varied from 0.82 to 2.0 liters per second; recovery rates varied from 7.57 to 8.83 liters per second. The volume of recovered water that did not exceed the U.S. Environmental Protection Agency secondary standard for chloride of 250 milligrams per liter varied from 1.48 to 2.46 million liters for the four tests.

Recovery efficiencies for this investigation were comparable to those during a pilot investigation conducted from 1993 to 1995 and varied between 21 and 34 percent for the four aquifer storage recovery tests. Recovery efficiencies were essentially equal for the 3-month (cycle 3) and 6-month (cycle 4) storage cycles. Solute transport modeling indicated a consistent recovery efficiency of approximately 25 percent, regardless of length of injection.

During this investigation, a total of 193 water-quality samples were collected and analyzed for physical properties, major and minor ions, and nutrients. Forty-five samples were collected and analyzed for total trihalomethane. Water-quality samples were collected directly from water spigots inline with the city water mains and the production well discharge pipe and by using a portable piston-driven submersible pump. Samples collected during this investigation documented baseline aquifer and injectant water quality and changes in the mixed water quality during injection, storage, and recovery.

Total trihalomethane data collected during aquifer storage recovery cycle 4 indicated that this constituent would not restrict the use of recovered water for drinking-water purposes. During recovery, total trihalomethane concentrations decreased from 8.5 micrograms per liter to less than detection levels at the production well. Comparison of chloroform to chloride concentrations in ground-water samples and the absence of bromoform, bromodichloromethane, and chlorodibromomethane in all ground-water samples indicate that total trihalomethane was nonconservative during aquifer storage recovery cycle 4. Total trihalomethane concentrations in samples collected from observation wells during storage typically were lower than trihalomethane concentrations that would occur as a result of mixing of injected and aquifer water. The mechanism of trihalomethane reduction was not evaluated during this investigation; however, other investigations have indicated that mixing and microbial degradation may reduce trihalomethane concentrations during storage and recovery.

During this investigation, treated surface water with total-dissolved-solids concentration of approximately 100 milligrams per liter was injected into the Santee Limestone/Black Mingo aquifer with a total-dissolved-solids concentration of 4,000 milligrams per liter. During the injection period, breakthrough of low chloride water at observation well CHN-809 constrained the product of porosity times thickness to 0.8 meter, which is consistent with thin production zones of no more than 4 meters each.

A reactive transport model was developed that included two 2-meter thick layers to describe each of the production zones. The four layers composing the production zones were assigned porosity ranging from 0.1 to 0.3 and hydraulic conductivity ranging from 1 to 8.4 meters per day. Specific storage of the aquifer and confining units was estimated to be $1.5 \times 10^{-5}$ meter$^{-1}$. Dispersivity of all layers was 0.5 meter. Leakage through the confining unit was estimated to be minimal—vertical hydraulic conductivity of the confining units was $1 \times 10^{-6}$ meter per day; therefore, leakage was eliminated in the reactive transport modeling. The assigned properties produced good agreement between measured chloride concentrations and simulated chloride concentrations for two observation wells during an entire aquifer storage recovery cycle and for the recovery well after pumping was initiated.

The aquifer water and treated surface water were sodium-chloride and calcium/sodium-bicarbonate water types, respectively. Inverse geochemical modeling indicated that mixing, cation exchange, and calcite dissolution were the dominant reactions occurring during aquifer storage recovery testing in the Santee Limestone/Black Mingo aquifer. Potable water injected into the Santee Limestone/Black Mingo aquifer evolved chemically by mixing with brackish background water and reaction with calcite and cation exchangers in the sediment. Reactive-transport model simulations indicate that the calcite and exchange reactions can be treated as equilibrium processes. By adjustment of the number of cation exchange sites per liter of water, good agreement was obtained between measured concentrations of major ions and pH and simulated values for observation and recovery wells.

Simulations with the calibrated reactive transport model indicated that approximately one-fourth of the water injected into the aquifer can be recovered as potable water, regardless of the length of the injection period. Simulations indicate that calcite dissolves near the injection well and precipitates where freshwater and brackish water mix during injection. However, the amounts of calcite dissolved and precipitated have a negligible effect on the porosity of the aquifer and should not affect the flow of water during aquifer storage recovery. Finally, simulations indicated that the regional configuration of the potentiometric surface in the aquifer had a negligible effect on the storage of water. Potable water was distributed symmetrically around an injection well even in the presence of a regional hydraulic gradient of about 0.0005 meter per meter in the aquifer. When injection ceased, the regional hydraulic gradient was reestablished and the freshwater bubble moved at a ground-water velocity of approximately 15 meters per year.

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# Appendixes

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 1** | | | | | | | | | | | | | | |
| CHN-809 | 129 | 11/8/99 14:18 | 8.4 | 6,520 | 19 | 25 | 42 | 1,360 | 694 | 1,540 | 2.7 | 39 | 250 | --- |
| CHN-809 | 129 | 12/3/99 13:49 | 8.6 | 3,880 | 8.9 | 12 | 25 | 770 | 487 | 830 | 4.8 | 29 | 160 | --- |
| CHN-809 | 116 | 12/3/99 16:33 | 8.5 | 3,150 | 6.0 | 8.2 | 21 | 630 | 417 | 630 | 5.4 | 26 | 140 | --- |
| CHN-810 | 129 | 12/7/99 13:20 | 8.5 | 6,050 | 16 | 23 | 39 | 1,250 | 680 | 1,400 | 3.0 | 38 | 250 | --- |
| CHN-810 | 112 | 12/7/99 15:47 | 8.6 | 4,710 | 10 | 14 | 29 | 990 | 582 | 1,030 | 3.8 | 33 | 220 | --- |
| CHN-810 | 129 | 1/6/00 14:01 | 8.6 | 2,410 | 11 | 15 | 29 | 920 | 543 | 1,010 | 4.7 | 32 | 190 | --- |
| CHN-810 | 112 | 1/6/00 16:20 | 8.6 | 2,580 | 3.9 | 5.2 | 18 | 530 | 365 | 490 | 6.3 | 23 | 120 | --- |
| CHN-809 | 129 | 1/12/00 14:32 | 8.6 | 2,750 | 6.2 | 8.1 | 18 | 580 | 393 | 570 | 3.7 | 25 | 120 | --- |
| CHN-809 | 116 | 1/12/00 16:34 | 8.5 | 1,870 | 3.8 | 5.1 | 12 | 410 | E309 | 360 | 3.8 | 21 | 92 | --- |
| TDW | NA | 1/12/00 16:34 | 7.4 | 189 | 20 | 1.7 | 1.6 | 7 | 30 | 14 | 0.9 | 6.4 | 29 | --- |
| CHN-809 | 116 | 2/7/00 14:15 | 8.7 | 2,100 | 3.8 | 5.3 | 14 | 440 | 332 | 390 | 4.1 | 21 | 95 | --- |
| CHN-809 | 129 | 2/7/00 16:35 | 8.6 | 2,410 | 4.8 | 6.4 | 16 | 500 | 356 | 480 | 3.9 | 23 | 110 | --- |
| CHN-812 | NA | 3/7/00 12:50 | 8.5 | 6,880 | 22 | 27 | 42 | 1,460 | 728 | 1,650 | 2.2 | 42 | 260 | --- |
| CHN-809 | 129 | 4/6/00 12:36 | 8.6 | 7,130 | 21 | 28 | 42 | 1,470 | 773 | 1,720 | 2.4 | 42 | 270 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 2** | | | | | | | | | | | | | | |
| CHN-809 | 116 | 5/15/00 10:53 | 8.6 | 5,080 | 12 | 15 | 28 | 1,040 | 597 | 1,140 | 3.4 | 35 | 210 | --- |
| CHN-809 | 129 | 5/15/00 12:36 | 8.6 | 6,880 | 13 | 17 | 32 | 1,100 | 623 | 1,210 | 3.4 | 36 | 220 | --- |
| TDW | NA | 5/19/00 12:49 | 7.6 | 202 | 18 | 1.5 | 1.7 | 12 | 27 | 19 | 1.0 | 5.9 | 25 | --- |
| CHN-809 | 116 | 5/23/00 11:04 | 8.7 | 2,930 | 5.2 | 6.8 | 19 | 610 | 404 | 580 | 5.1 | 25 | 130 | --- |
| CHN-809 | 129 | 5/23/00 13:09 | 8.7 | 3,480 | 7.2 | 8.9 | 22 | 710 | 455 | 720 | 4.8 | 28 | 150 | --- |
| CHN-810 | 112 | 5/26/00 10:57 | 8.7 | 4,350 | 8.6 | 11 | 27 | 900 | 559 | 930 | 4.1 | 32 | 200 | --- |
| CHN-810 | 129 | 5/26/00 13:02 | 8.8 | 5,670 | 14 | 19 | 37 | 1,190 | 674 | 1,290 | 3.2 | 37 | 240 | --- |
| CHN-810 | 112 | 6/15/00 11:32 | 8.7 | 2,320 | 3.5 | 4.4 | 17 | 490 | 371 | 440 | 6.1 | 23 | 110 | --- |
| CHN-810 | 129 | 6/15/00 14:31 | 8.6 | 4,120 | 9.3 | 12 | 25 | 840 | 521 | 890 | 4.9 | 30 | 170 | --- |
| CHN-809 | 112 | 6/16/00 11:39 | --- | --- | --- | --- | --- | --- | E271 | --- | --- | --- | --- | --- |
| CHN-809 | 129 | 6/16/00 13:35 | 8.9 | 2,370 | 6.8 | 5.8 | 15 | 480 | 388 | 450 | 3.6 | 25 | 97 | --- |
| CHN-809 | 112 | 7/17/00 11:04 | 9.0 | 1,510 | 2.1 | 3.2 | 10 | 320 | 306 | 630 | 5.0 | 18 | 130 | --- |
| CHN-809 | 129 | 7/17/00 12:52 | 8.9 | 1,830 | 2.7 | 3.6 | 11 | 370 | 335 | 320 | 3.8 | 22 | 79 | --- |
| CHN-810 | 112 | 7/18/00 10:34 | 8.7 | 2,260 | 3.4 | 4.3 | 15 | 470 | 354 | 420 | 2.0 | 22 | 110 | --- |
| CHN-810 | 129 | 7/18/00 12:34 | 8.6 | 3,630 | 7.6 | 10 | 23 | 750 | 473 | 750 | 5.2 | 28 | 160 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 2 (Continued)** | | | | | | | | | | | | | | |
| CHN-812 | NA | 8/2/00 13:30 | 8.8 | 5,950 | 25 | 19 | 39 | 1,270 | 644 | 1,400 | 1.8 | 42 | 220 | --- |
| CHN-809 | 112 | 8/31/00 11:48 | 8.5 | 7,320 | 21 | 30 | 45 | 1,480 | 757 | 1,790 | 2.3 | 41 | 270 | --- |
| CHN-809 | 129 | 8/31/00 13:42 | 8.5 | 7,090 | 20 | 28 | 43 | 1,480 | 753 | 1,710 | 2.4 | 42 | 270 | --- |
| CHN-810 | 112 | 9/6/00 10:22 | 8.6 | 7,640 | 23 | 32 | 58 | 1,580 | 785 | 1,890 | 2.4 | 43 | 280 | --- |
| CHN-810 | 129 | 9/6/00 12:04 | 8.6 | 6,870 | 21 | 30 | 43 | 1,540 | 721 | 1,680 | 2.2 | 44 | 250 | --- |
| **Cycle 3** | | | | | | | | | | | | | | |
| CHN-812 | NA | 9/11/00 13:52 | 8.6 | 7,140 | 22 | 30 | 44 | 1,480 | 760 | 1,720 | 2.4 | 42 | 270 | --- |
| CHN-809 | 112 | 9/12/00 10:40 | 8.7 | 7,220 | 21 | 26 | 40 | 1,500 | 751 | 1,800 | 1.2 | 42 | 280 | --- |
| CHN-809 | 129 | 9/12/00 12:34 | 8.6 | 7,220 | 21 | 25 | 41 | 1,600 | 749 | 1,800 | 2.4 | 41 | 280 | --- |
| CHN-810 | 112 | 9/14/00 11:05 | 8.5 | 7,220 | 21 | 26 | 41 | 1,500 | 745 | 1,800 | 2.4 | 42 | 280 | --- |
| CHN-810 | 129 | 9/14/00 12:57 | 8.5 | 7,210 | 21 | 25 | 40 | 1,500 | 751 | 1,800 | 2.4 | 41 | 280 | --- |
| CHN-809 | 112 | 9/15/00 12:11 | 8.7 | 6,200 | 16 | 20 | 37 | 1,300 | 677 | 1,500 | 1.4 | 39 | 250 | --- |
| CHN-809 | 129 | 9/15/00 14:18 | 8.6 | 6,570 | 18 | 22 | 36 | 1,400 | 700 | 1,600 | 1.3 | 40 | 260 | --- |
| CHN-809 | 112 | 9/19/00 10:44 | 8.6 | 3,690 | 6.1 | 7.2 | 18 | 770 | 460 | 780 | 2.3 | 29 | 170 | --- |
| CHN-809 | 129 | 9/19/00 12:40 | 8.8 | 4,880 | 11 | 14 | 27 | 1,000 | 550 | 1,100 | 1.8 | 34 | 210 | --- |
| CHN-810 | 112 | 9/20/00 10:54 | 8.7 | 6,760 | 19 | 23 | 38 | 1,400 | 721 | 1,600 | 2.6 | 40 | 270 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | |
| CHN-810 | 129 | 9/20/00 12:45 | 8.7 | 7,110 | 20 | 24 | 39 | 1,500 | 736 | 1,700 | 2.5 | 41 | 270 | --- |
| CHN-809 | 112 | 9/22/00 11:45 | 8.7 | 2,680 | 3.5 | 4.1 | 13 | 550 | 373 | 530 | 2.7 | 25 | 130 | --- |
| CHN-809 | 129 | 9/22/00 13:33 | 8.7 | 4,040 | 8.2 | 9.6 | 23 | 840 | 487 | 920 | 2.2 | 31 | 170 | --- |
| CHN-810 | 112 | 9/26/00 9:55 | 8.7 | 5,080 | 11 | 14 | 30 | 1,100 | 597 | 1,100 | 1.7 | 35 | 220 | --- |
| CHN-810 | 129 | 9/26/00 11:39 | 8.6 | 6,130 | 16 | 20 | 34 | 1,300 | 676 | 1,500 | 1.5 | 38 | 260 | --- |
| CHN-809 | 112 | 10/2/00 10:53 | 9.0 | 1,570 | 1.9 | 1.7 | 11 | 330 | 286 | 260 | 2.2 | 20 | 80 | --- |
| CHN-809 | 129 | 10/2/00 12:55 | 8.8 | 2,770 | 5.2 | 6.2 | 16 | 570 | 387 | 560 | 2.2 | 26 | 130 | --- |
| CHN-810 | 112 | 10/3/00 10:35 | 8.7 | 3,570 | 6.2 | 7.2 | 22 | 730 | 469 | 750 | 2.5 | 28 | 170 | --- |
| CHN-810 | 129 | 10/3/00 12:30 | 8.5 | 5,180 | 13 | 15 | 29 | 1,100 | 595 | 1,200 | 1.9 | 35 | 220 | --- |
| CHN-810 | 112 | 10/13/00 10:40 | 8.7 | 2,590 | 3.8 | 4.5 | 16 | 550 | 375 | 510 | 3.0 | 24 | 130 | --- |
| CHN-810 | 129 | 10/13/00 12:51 | 8.8 | 4,300 | 9.5 | 12 | 25 | 900 | 518 | 940 | 2.4 | 30 | 190 | --- |
| CHN-809 | 112 | 10/23/00 11:24 | 9.3 | 1,130 | 1.4 | 1.7 | 5.6 | 230 | 232 | 170 | 2.5 | 17 | 57 | --- |
| CHN-809 | 129 | 10/23/00 13:34 | 9.0 | 1,780 | 3.3 | 3.7 | 10 | 370 | 289 | 330 | 1.6 | 23 | 80 | --- |
| CHN-810 | 112 | 10/24/00 10:54 | 8.7 | 2,250 | 3.5 | 4.0 | 13 | 480 | 339 | 430 | 1.9 | 22 | 110 | --- |
| CHN-810 | 129 | 10/24/00 13:12 | 8.7 | 3,920 | 8.7 | 11 | 22 | 810 | 482 | 850 | 2.4 | 29 | 170 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | |
| CHN-809 | 112 | 11/2/00 11:56 | 9.3 | 1,400 | 2.1 | 2.6 | 9.3 | 290 | 273 | 230 | 2.9 | 18 | 68 | --- |
| CHN-809 | 129 | 11/2/00 14:30 | 9.3 | 1,740 | 2.9 | 3.1 | 10 | 360 | 309 | 300 | 1.7 | 21 | 82 | --- |
| TDW | NA | 11/2/00 14:40 | 7.8 | 227 | 20 | 2.3 | 2.4 | 18 | 32 | 23 | 1.1 | 7.6 | 33 | --- |
| CHN-810 | 112 | 11/3/00 11:25 | 8.8 | 2,030 | 2.6 | 2.8 | 11 | 420 | 325 | 370 | 2.0 | 21 | 100 | --- |
| CHN-810 | 112 | 11/3/00 11:26 | 8.8 | 2,040 | 2.8 | 3.0 | 14 | 430 | 323 | 380 | 2.0 | 22 | 100 | --- |
| CHN-810 | 129 | 11/3/00 13:27 | 8.6 | 3,380 | 6.9 | 8.2 | 19 | 690 | 440 | 700 | 2.6 | 27 | 160 | --- |
| CHN-810 | 129 | 11/3/00 13:28 | 8.8 | 3,380 | 7.0 | 8.4 | 19 | 720 | 438 | 700 | 2.6 | 27 | 160 | --- |
| CHN-809 | 112 | 11/13/00 12:50 | 9.2 | 1,560 | 2.4 | 3.3 | 11 | 320 | 286 | 260 | 2.9 | 18 | 72 | --- |
| CHN-809 | 129 | 11/13/00 14:56 | 9.1 | 1,800 | 2.9 | 3.6 | 12 | 370 | 304 | 310 | 1.7 | 22 | 85 | --- |
| CHN-810 | 112 | 11/20/00 12:35 | 8.8 | 2,060 | 2.8 | 3.0 | 13 | 430 | 327 | 380 | 2.0 | 22 | 100 | --- |
| CHN-810 | 129 | 11/20/00 14:28 | 8.7 | 3,340 | 6.7 | 7.9 | 19 | 680 | 437 | 720 | 2.6 | 27 | 150 | --- |
| CHN-809 | 112 | 12/5/00 12:07 | --- | --- | 2.6 | 3.5 | 9.9 | 330 | --- | --- | --- | 19 | --- | --- |
| CHN-809 | 129 | 12/5/00 14:20 | 9.0 | 1,920 | 3.2 | 3.9 | 11 | 400 | 322 | 360 | 1.7 | 22 | 86 | --- |
| CHN-809 | 129 | 12/5/00 14:21 | 9.0 | 1,920 | 3.2 | 3.9 | 12 | 390 | 323 | 350 | 1.7 | 22 | 85 | --- |
| CHN-810 | 129 | 12/6/00 14:43 | 8.7 | 3,500 | 7.2 | 8.6 | 20 | 720 | 450 | 770 | 2.6 | 28 | 160 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | |
| CHN-809 | 112 | 12/18/00 13:28 | 9.2 | 1,600 | 2.6 | 3.5 | 12 | 330 | 291 | 280 | 1.5 | 19 | 75 | --- |
| CHN-809 | 129 | 12/18/00 15:27 | 9.0 | 1,960 | 3.3 | 4.1 | 12 | 400 | 324 | 360 | 1.7 | 22 | 88 | --- |
| CHN-810 | 112 | 12/19/00 12:45 | 8.8 | 2,260 | 3.6 | 3.9 | 14 | 460 | 341 | 430 | 2.9 | 22 | 110 | --- |
| CHN-810 | 129 | 12/19/00 14:55 | 8.6 | 3,530 | 7.2 | 8.8 | 20 | 750 | 453 | 740 | 5.1 | 27 | 160 | --- |
| CHN-810 | 112 | 1/22/01 12:38 | 8.7 | 2,400 | 4.2 | 4.7 | 16 | 500 | 352 | 470 | 5.7 | 23 | 120 | --- |
| CHN-810 | 129 | 1/22/01 15:27 | 8.7 | 3,620 | 7.9 | 9.5 | 21 | 760 | 461 | 780 | 5.0 | 28 | 170 | --- |
| CHN-809 | 112 | 1/24/01 12:49 | 9.2 | 1,220 | 1.2 | 1.3 | 7.8 | 260 | 271 | 170 | 3.3 | 18 | 68 | --- |
| CHN-809 | 129 | 1/24/01 15:00 | 8.9 | 1,900 | 2.9 | 3.4 | 11 | 400 | 322 | 340 | 3.6 | 22 | 89 | --- |
| CHN-812 | NA | 1/31/01 15:42 | 7.9 | 332 | 33 | 3.2 | 4.5 | 35 | 47 | 40 | 1.0 | 10 | 40 | --- |
| CHN-812 | NA | 2/1/01 10:15 | 8.3 | 685 | 27 | 3.3 | 12 | 100 | 93 | 120 | 1.0 | 17 | 53 | --- |
| CHN-812 | NA | 2/1/01 15:20 | 8.3 | 801 | 28 | 3.6 | 14 | 120 | 106 | 140 | 1.0 | 18 | 57 | --- |
| CHN-812 | NA | 2/2/01 10:06 | 8.4 | 1,160 | 29 | 4.6 | 18 | 190 | 146 | 230 | 1.1 | 21 | 68 | --- |
| CHN-809 | 112 | 2/2/01 12:20 | 8.6 | 3,470 | 5.8 | 11 | 21 | 720 | 408 | 800 | 3.5 | 28 | 140 | --- |
| CHN-809 | 129 | 2/2/01 14:50 | 8.7 | 3,090 | 3.8 | 7.4 | 18 | 640 | 389 | 640 | 1.8 | 27 | 130 | --- |
| CHN-812 | NA | 2/2/01 15:33 | 8.4 | 1,240 | 29 | 4.9 | 19 | 210 | 154 | 230 | 1.1 | 22 | 72 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | |
| CHN-812 | NA | 2/3/01 10:33 | 7.8 | 1,660 | 28 | 6.0 | 21 | 290 | 191 | 320 | 1.4 | 24 | 87 | --- |
| CHN-812 | NA | 2/3/01 18:48 | 8.5 | 1,700 | 29 | 6.2 | 21 | 300 | 196 | 340 | 1.2 | 24 | 89 | --- |
| CHN-812 | NA | 2/4/01 12:00 | 8.3 | 1,960 | 26 | 7.1 | 22 | 360 | 221 | 410 | 1.2 | 26 | 97 | --- |
| CHN-812 | NA | 2/4/01 19:24 | 8.5 | 2,090 | 27 | 7.7 | 23 | 390 | 233 | 450 | 1.3 | 27 | 100 | --- |
| CHN-812 | NA | 2/5/01 10:37 | 8.2 | 2,340 | 26 | 8.4 | 24 | 450 | 257 | 510 | 1.4 | 28 | 110 | --- |
| CHN-809 | 112 | 2/5/01 13:00 | 8.6 | 4,240 | 7.3 | 12 | 24 | 890 | 487 | 960 | 3.3 | 32 | 170 | --- |
| CHN-809 | 129 | 2/5/01 14:49 | 8.5 | 4,080 | 6.3 | 11 | 23 | 860 | 483 | 910 | 3.4 | 32 | 170 | --- |
| CHN-812 | NA | 2/5/01 15:13 | 8.4 | 2,420 | 26 | 8.7 | 24 | 460 | 264 | 540 | 1.4 | 29 | 110 | --- |
| CHN-812 | NA | 2/7/01 11:00 | 8.6 | 3,250 | 26 | 11 | 27 | 620 | 337 | 710 | 1.5 | 33 | 140 | --- |
| CHN-809 | 112 | 2/7/01 12:58 | 8.4 | 5,030 | 10 | 16 | 30 | 1,100 | 558 | 1,200 | 1.5 | 35 | 200 | --- |
| CHN-809 | 129 | 2/7/01 15:06 | 8.4 | 4,890 | 9.2 | 14 | 27 | 1,000 | 554 | 1,100 | 1.5 | 35 | 200 | --- |
| CHN-809 | 129 | 2/7/01 15:07 | 8.4 | 4,890 | 9.1 | 14 | 25 | 1,100 | 554 | 1,100 | 1.5 | 35 | 200 | --- |
| CHN-812 | NA | 2/13/01 10:21 | 8.5 | 5,530 | 27 | 17 | 35 | 1,200 | 553 | 1,300 | 1.9 | 41 | 220 | --- |
| CHN-809 | 112 | 2/13/01 12:20 | 8.4 | 6,410 | 17 | 22 | 36 | 1,400 | 676 | 1,500 | 2.7 | 38 | 250 | --- |
| CHN-809 | 129 | 2/13/01 14:29 | 8.4 | 6,390 | 16 | 21 | 36 | 1,400 | 678 | 1,500 | 2.7 | 39 | 250 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | |
| CHN-812 | NA | 2/20/01 15:41 | 8.4 | 6,600 | 24 | 22 | 41 | 1,400 | 663 | 1,600 | 2.1 | 41 | 250 | --- |
| CHN-812 | NA | 3/1/01 14:07 | 8.4 | 6,970 | 23 | 25 | 39 | 1,500 | 715 | 1,700 | 2.2 | 41 | 260 | --- |
| CHN-812 | NA | 3/5/01 14:05 | 8.3 | 7,050 | 23 | 26 | 39 | 1,500 | 709 | 1,700 | 2.3 | 41 | 270 | --- |
| CHN-812 | NA | 3/16/01 15:12 | 8.2 | 7,180 | 23 | 27 | 40 | 1,600 | 725 | 1,700 | 2.3 | 41 | 270 | --- |
| CHN-810 | 112 | 3/30/01 12:16 | 8.2 | 7,410 | 23 | 27 | 36 | 1,600 | 738 | 1,800 | 2.3 | 41 | 280 | --- |
| CHN-810 | 129 | 3/30/01 14:38 | 8.2 | 7,210 | 21 | 25 | 41 | 1,600 | 739 | 1,800 | 2.4 | 41 | 280 | --- |
| CHN-809 | 112 | 4/2/01 12:37 | 8.7 | 7,320 | 21 | 26 | 41 | 1,600 | 747 | 1,800 | 2.3 | 40 | 280 | --- |
| CHN-809 | 112 | 4/2/01 12:38 | 8.6 | 7,310 | 21 | 25 | 42 | 1,600 | 735 | 1,800 | 2.3 | 39 | 280 | --- |
| CHN-809 | 129 | 4/2/01 13:56 | 8.6 | 7,240 | 21 | 25 | 40 | 1,600 | 748 | 1,800 | 2.3 | 40 | 280 | --- |
| CHN-809 | 129 | 4/2/01 13:57 | 8.4 | 7,250 | 21 | 25 | 41 | 1,600 | 747 | 1,800 | 2.3 | 40 | 280 | --- |
| CHN-812 | NA | 4/2/01 14:27 | 8.3 | 6,990 | 23 | 26 | 34 | 1,500 | 701 | 1,700 | 2.3 | 39 | 260 | --- |
| **Cycle 4** | | | | | | | | | | | | | | |
| CHN-809 | 112 | 4/4/01 11:16 | 8.2 | 7,190 | 21 | 26 | 40 | 1,600 | --- | 1,800 | 1.2 | 42 | 260 | --- |
| CHN-809 | 129 | 4/4/01 13:36 | 8.4 | 7,200 | 22 | 26 | 41 | 1,600 | --- | 1,800 | 1.2 | 42 | 260 | --- |
| CHN-809 | 112 | 4/6/01 11:50 | 8.2 | 6,470 | 17 | 21 | 36 | 1,400 | --- | 1,500 | 1.3 | 40 | 250 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | |
| CHN-809 | 129 | 4/6/01 13:10 | 8.5 | 6,750 | 19 | 23 | 38 | 1,500 | --- | 1,600 | 1.3 | 40 | 250 | --- |
| CHN-809 | 112 | 4/9/01 11:42 | 8.5 | 4,510 | 8.8 | 10 | 25 | 980 | --- | 1,000 | 1.9 | 32 | 200 | --- |
| CHN-809 | 129 | 4/9/01 13:57 | 8.2 | 5,530 | 14 | 17 | 31 | 1,200 | --- | 1,300 | 1.6 | 37 | 220 | --- |
| CHN-810 | 112 | 4/10/01 11:36 | 8.1 | 7,070 | 21 | 25 | 39 | 1,600 | --- | 1,700 | 1.2 | 41 | 260 | --- |
| CHN-810 | 129 | 4/10/01 14:07 | 8.1 | 7,100 | 21 | 26 | 40 | 1,600 | --- | 1,700 | 1.2 | 42 | 260 | --- |
| CHN-811A | Composite[a] | 4/11/01 13:10 | 8.1 | 7,530 | 23 | 28 | 43 | 1,700 | --- | 1,900 | 1.2 | 42 | 280 | --- |
| TDW | NA | 4/11/01 14:27 | 7.8 | 436 | 24 | 3.0 | 3.5 | 55 | --- | 60 | 1.2 | 3.2 | 56 | --- |
| CHN-809 | 112 | 4/12/01 11:14 | 8.5 | 3,330 | 5.3 | 5.9 | 19 | 720 | --- | 680 | 2.5 | 28 | 150 | --- |
| CHN-809 | 129 | 4/12/01 13:36 | 8.3 | 4,680 | 11 | 12 | 26 | 1,000 | --- | 1,100 | 2.0 | 33 | 200 | --- |
| CHN-811A | Composite[b] | 4/13/01 8:39 | 8.1 | 7,420 | 22 | 27 | 41 | 1,700 | --- | 1,800 | 1.2 | 42 | 270 | --- |
| CHN-810 | 112 | 4/17/01 11:29 | 8.3 | 5,520 | 14 | 16 | 32 | 1,200 | --- | 1,300 | 1.5 | 37 | 230 | --- |
| CHN-810 | 129 | 4/17/01 13:51 | 8.2 | 6,440 | 18 | 21 | 36 | 1,400 | --- | 1,500 | 1.4 | 40 | 250 | --- |
| CHN-809 | 129 | 4/20/01 13:50 | 8.5 | 3,310 | 6.4 | 7.4 | 19 | 700 | --- | 710 | 2.4 | 28 | 140 | --- |
| CHN-810 | 112 | 4/23/01 11:29 | 8.4 | 4,300 | 8.3 | 9.8 | 25 | 920 | --- | 930 | 2.1 | 31 | 190 | --- |
| CHN-810 | 129 | 4/23/01 14:28 | 8.5 | 5,750 | 15 | 18 | 32 | 1,300 | --- | 1,300 | 1.6 | 37 | 230 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | |
| CHN-810 | 112 | 5/2/01 11:45 | 8.6 | 3,140 | 5.1 | 5.8 | 20 | 670 | 423 | 630 | 2.8 | 27 | 150 | <1.0 |
| CHN-810 | 112 | 5/2/01 11:46 | 8.4 | 3,130 | 5.4 | 5.6 | 18 | 650 | 423 | 640 | 5.6 | 26 | 150 | <1.0 |
| CHN-810 | 129 | 5/2/01 14:17 | 8.2 | 4,930 | 12 | 15 | 28 | 1,100 | 569 | 1,100 | 2.0 | 34 | 200 | <1.0 |
| CHN-810 | 129 | 5/2/01 14:18 | 8.0 | 4,940 | 12 | 13 | 27 | 1,100 | 569 | 1,100 | 4.1 | 33 | 200 | <1.0 |
| CHN-800 | 109 | 5/9/01 11:22 | 7.5 | 10,000 | 44 | 62 | 65 | 2,100 | 732 | 2,600 | 2.0 | 41 | 370 | 3.2 |
| CHN-810 | 112 | 5/18/01 11:04 | 8.6 | 2,520 | 4.3 | 4.7 | 15 | 520 | 355 | 500 | 2.9 | 24 | 120 | E1.0 |
| CHN-810 | 129 | 5/18/01 13:16 | 8.2 | 4,200 | 10 | 12 | 24 | 860 | 497 | 940 | 2.3 | 30 | 180 | E1.0 |
| CHN-811A | Composite[b] | 5/22/01 12:26 | 7.9 | 7,520 | 23 | 29 | 42 | 1,600 | 746 | 1,800 | 1.2 | 41 | 290 | <1.0 |
| CHN-809 | 112 | 5/23/01 11:40 | 9.6 | 1,350 | 2.1 | 2.4 | 7.7 | 280 | 264 | 210 | 2.6 | 17 | 63 | E1.0 |
| CHN-809 | 112 | 5/23/01 11:41 | 9.6 | 1,340 | 2.1 | 2.4 | 7.8 | 280 | 265 | 210 | 2.6 | 17 | 59 | E1.0 |
| TDW | NA | 5/23/01 13:25 | 8.3 | 220 | 19 | 2.1 | 2.0 | 18 | 31 | 20 | 1.0 | 3.9 | 31 | E1.0 |
| CHN-809 | 129 | 5/23/01 13:50 | 9.2 | 2,040 | 3.8 | 4.4 | 12 | 420 | 326 | 370 | 1.7 | 22 | 91 | E1.0 |
| CHN-809 | 112 | 5/31/01 12:43 | 9.6 | 1,230 | 1.8 | 2.2 | 7.6 | 270 | 257 | 190 | 2.6 | 17 | 60 | E1.0 |
| CHN-809 | 129 | 5/31/01 15:08 | 9.4 | 1,610 | 2.5 | 2.7 | 9.6 | 350 | 295 | 270 | 3.3 | 20 | 76 | E1.0 |
| CHN-810 | 112 | 6/5/01 11:20 | 8.8 | 2,190 | 3.0 | 3.4 | 13 | 480 | 330 | 410 | 5.7 | 22 | 110 | E1.0 |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | |
| CHN-810 | 129 | 6/5/01 13:55 | 8.2 | 3,600 | 7.1 | 8.5 | 21 | 760 | 451 | 750 | 4.9 | 28 | 160 | E1.0 |
| CHN-809 | 112 | 6/26/01 11:56 | 9.4 | 1,480 | 2.4 | 2.9 | 8.8 | 300 | 274 | 230 | 3.0 | 18 | 66 | <1.0 |
| CHN-809 | 129 | 6/26/01 14:15 | 9.2 | 1,860 | 3.2 | 4.0 | 11 | 390 | 307 | 340 | 3.3 | 22 | 79 | <1.0 |
| CHN-810 | 112 | 6/27/01 12:29 | 8.8 | 2,130 | 2.9 | 3.1 | 13 | 440 | 326 | 380 | 5.9 | 22 | 100 | <1.0 |
| CHN-810 | 129 | 6/27/01 14:30 | 8.2 | 3,630 | 7.3 | 8.8 | 20 | 775 | 453 | 750 | 5.0 | 28 | 160 | <1.0 |
| CHN-809 | 112 | 7/17/01 10:55 | --- | --- | --- | --- | --- | --- | --- | --- | 3.0 | --- | --- | --- |
| CHN-809 | 129 | 7/17/01 13:09 | 9.1 | 1,920 | 3.2 | 4.1 | 12 | 414 | 313 | 350 | 3.3 | 21 | 82 | <1.0 |
| CHN-810 | 112 | 7/19/01 11:40 | 8.7 | 2,230 | 3.2 | 3.6 | 14 | 477 | 333 | 410 | 5.7 | 22 | 110 | <1.0 |
| CHN-810 | 129 | 7/19/01 13:46 | 8.2 | 3,650 | 7.4 | 9.1 | 21 | 795 | 458 | 770 | 4.9 | 28 | 160 | <1.0 |
| CHN-809 | 112 | 8/7/01 12:24 | 9.3 | 1,460 | 2.1 | 2.5 | 8.6 | 296 | 276 | 210 | 3.0 | 17 | 66 | <1.0 |
| CHN-809 | 129 | 8/7/01 14:56 | 9.2 | 1,790 | 2.8 | 3.4 | 11 | 388 | 309 | 320 | 3.5 | 21 | 82 | <1.0 |
| CHN-810 | 112 | 8/8/01 12:22 | 8.7 | 2,210 | 3.2 | 3.4 | 13 | 465 | 332 | 400 | 5.9 | 22 | 110 | <1.0 |
| CHN-810 | 129 | 8/8/01 14:40 | 8.2 | 3,610 | 7.7 | 9.3 | 20 | 782 | 454 | 760 | 5.1 | 28 | 160 | <1.0 |
| CHN-809 | 112 | 8/22/01 12:10 | 9.3 | 1,430 | 2.2 | 2.9 | 10 | 310 | 275 | 230 | 3.0 | 17 | 66 | <1.0 |
| CHN-809 | 129 | 8/22/01 14:40 | 9.1 | 1,830 | 2.8 | 3.6 | 12 | 380 | 311 | 330 | 3.5 | 21 | 81 | <1.0 |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | |
| CHN-810 | 129 | 8/23/01 14:40 | 8.3 | 3,600 | 7.5 | 9.8 | 24 | 750 | 456 | 800 | 5.0 | 28 | 160 | <1.0 |
| CHN-809 | 112 | 9/18/01 12:02 | 9.3 | 1,440 | 2.0 | 2.5 | 8.6 | 310 | 276 | 230 | 3.1 | 17 | 66 | <1.0 |
| CHN-809 | 129 | 9/18/01 14:43 | 9.1 | 1,890 | 2.9 | 3.7 | 11 | 400 | 316 | 340 | 3.4 | 21 | 83 | <1.0 |
| CHN-810 | 112 | 9/19/01 11:46 | 8.8 | 2,280 | 3.4 | 3.9 | 13 | 470 | 336 | 420 | 5.7 | 22 | 110 | <1.0 |
| CHN-810 | 129 | 9/19/01 14:08 | 8.2 | 3,630 | 7.3 | 9.3 | 19 | 780 | 457 | 800 | 5.1 | 27 | 160 | <1.0 |
| CHN-809 | 112 | 10/15/01 11:52 | 9.2 | 1,580 | 2.6 | 3.6 | 9.9 | 336 | E313 | 276 | 3.0 | 18 | 72 | <1.0 |
| CHN-809 | 129 | 10/15/01 14:12 | 9.0 | 1,960 | 3.1 | 4.1 | 11 | 406 | E339 | 360 | 3.4 | 21 | 85 | <1.0 |
| CHN-810 | 112 | 10/16/01 12:18 | 8.6 | 2,360 | 3.8 | 4.5 | 14 | 490 | 344 | 440 | 5.6 | 22 | 110 | <1.0 |
| CHN-810 | 129 | 10/16/01 14:17 | 8.4 | 3,740 | 7.8 | 10 | 20 | 800 | 465 | 770 | 4.8 | 28 | 160 | <1.0 |
| CHN-812 | NA | 11/8/01 14:40 | 7.9 | 401 | 21 | 2.8 | 7.6 | 49 | E70 | 48 | 1.0 | 13 | 42 | <1.0 |
| CHN-812 | NA | 11/9/01 8:40 | 8.1 | 823 | 27 | 4.1 | 15 | 130 | E112 | 150 | 1.0 | 18 | 53 | <1.0 |
| CHN-810 | 112 | 11/9/01 13:00 | 8.2 | 4,960 | 13 | 18 | 33 | 1,030 | E552 | 1,150 | 4.0 | 32 | 200 | <1.0 |
| CHN-810 | 129 | 11/9/01 15:20 | 8.2 | 4,550 | 11 | 15 | 28 | 966 | E534 | 1,050 | 4.2 | 33 | 189 | <1.0 |
| CHN-812 | NA | 11/9/01 16:10 | 8.1 | 986 | 27 | 4.8 | 17 | 160 | E130 | 190 | 1.1 | 19 | 58 | <1.0 |
| CHN-812 | NA | 11/10/01 10:20 | 8.1 | 1,380 | 26 | 6.0 | 21 | 236 | E174 | 289 | 1.2 | 22 | 72 | <1.0 |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; E, estimated; TDW, treated drinking water; NA, not applicable; <, less than; µg/L, microgram per liter; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Laboratory pH (su) | Laboratory specific conductance (µS/cm) | Calcium, dissolved (mg/L) | Magnesium, dissolved (mg/L) | Potassium, dissolved (mg/L) | Sodium, dissolved (mg/L) | ANC (mg/L as $CaCO_3$) | Chloride, dissolved (mg/L) | Fluoride, dissolved (mg/L) | Silica, dissolved (mg/L) | Sulfate, dissolved (mg/L) | Sulfide, total (mg/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | |
| CHN-812 | NA | 11/10/01 16:17 | 8.2 | 1,290 | 26 | 5.7 | 20 | 216 | E165 | 256 | 1.1 | 21 | 70 | <1.0 |
| CHN-812 | NA | 11/11/01 12:20 | 8.2 | 1,700 | 25 | 7.1 | 22 | 310 | E204 | 365 | 1.3 | 23 | 83 | <1.0 |
| CHN-809 | 112 | 11/13/01 13:50 | 8.3 | 4,460 | 8.8 | 15 | 28 | 968 | E504 | 1,050 | 3.1 | 34 | 174 | <1.0 |
| CHN-809 | 129 | 11/13/01 16:20 | 8.5 | 4,260 | 6.5 | 13 | 26 | 924 | E521 | 957 | 3.1 | 33 | 172 | <1.0 |
| CHN-812 | NA | 11/13/01 17:21 | 8.2 | 2,560 | 24 | 10 | 25 | 510 | E286 | 581 | 1.4 | 29 | 115 | <1.0 |
| CHN-812 | NA | 11/27/01 16:30 | 8.0 | 3,540 | 35 | 15 | 23 | 690 | E346 | 790 | 1.3 | 31 | 140 | <1.0 |
| CHN-812 | NA | 12/6/01 12:30 | 8.1 | 6,260 | 24 | 21 | 39 | 1,270 | E641 | 1,570 | 1.9 | 42 | 232 | <1.0 |
| CHN-809 | 112 | 12/18/01 12:14 | 8.1 | 6,890 | 20 | 28 | 40 | 1,430 | 709 | 1,680 | 2.4 | 41 | 250 | <1.0 |
| CHN-812 | NA | 12/18/01 13:27 | 8.2 | 6,470 | 27 | 25 | 36 | 1,330 | 650 | 1,560 | 1.9 | 43 | 240 | <1.0 |
| CHN-809 | 129 | 12/18/01 14:39 | 8.1 | 6,710 | 18 | 26 | 39 | 1,400 | 713 | 1,600 | 2.4 | 41 | 250 | <1.0 |
| CHN-809 | 112 | 2/14/02 13:46 | 7.8 | 7,250 | 21 | 29 | 41 | 1,570 | 756 | 1,800 | 2.2 | 41 | 270 | <1.0 |
| CHN-809 | 129 | 2/14/02 15:48 | 7.7 | 7,110 | 20 | 27 | 39 | 1,530 | 754 | 1,740 | 2.3 | 41 | 260 | <1.0 |
| CHN-810 | 112 | 2/15/02 12:55 | 7.8 | 7,090 | 19 | 27 | 41 | 1,490 | 750 | 1,720 | 2.3 | 41 | 270 | <1.0 |
| CHN-810 | 129 | 2/15/02 15:24 | 7.8 | 7,380 | 23 | 30 | 45 | 1,560 | 755 | 1,840 | 2.2 | 42 | 270 | 2.0 |
| CHN-812 | NA | 2/15/02 16:06 | 7.8 | 7,120 | 21 | 29 | 42 | 1,510 | 747 | 1,740 | 2.3 | 41 | 270 | 1.0 |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 1** | | | | | | | | | | | | | | | | |
| CHN-809 | 129 | 11/8/99 14:18 | --- | --- | --- | 3,570 | --- | --- | 1,160 | 23.9 | 7.5 | 6,840 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 12/3/99 13:49 | --- | --- | --- | 2,230 | --- | --- | 530 | 21.5 | 7.9 | 3,960 | <1.0 | --- | --- | --- |
| CHN-809 | 116 | 12/3/99 16:33 | --- | --- | --- | 1,850 | --- | --- | 350 | 19.2 | 8.3 | 3,120 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 12/7/99 13:20 | --- | --- | --- | 3,420 | --- | --- | 1,030 | 17.5 | 7.6 | 6,180 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 12/7/99 15:47 | --- | --- | --- | 2,710 | --- | --- | 670 | 17.5 | 7.9 | 4,810 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 1/6/00 14:01 | --- | --- | --- | 2,590 | --- | --- | 690 | 19.8 | 8.0 | 4,610 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 1/6/00 16:20 | --- | --- | --- | 1,590 | --- | --- | 270 | --- | --- | --- | --- | --- | --- | --- |
| CHN-809 | 129 | 1/12/00 14:32 | --- | --- | --- | 1,520 | --- | --- | 360 | 22.3 | 9.0 | 2,840 | <1.0 | --- | --- | --- |
| CHN-809 | 116 | 1/12/00 16:34 | --- | --- | --- | 1,080 | --- | --- | 220 | 19.7 | 9.4 | 2,030 | <1.0 | --- | --- | --- |
| TDW | NA | 1/12/00 16:34 | --- | --- | --- | 17 | --- | --- | 39 | --- | --- | --- | --- | --- | --- | --- |
| CHN-809 | 116 | 2/7/00 14:15 | --- | --- | --- | 1,120 | --- | --- | 220 | 21.6 | 9.2 | 2,110 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 2/7/00 16:35 | --- | --- | --- | 1,270 | --- | --- | 270 | 19.5 | 9.1 | 2,380 | <1.0 | --- | --- | --- |
| CHN-812 | NA | 3/7/00 12:50 | --- | --- | --- | 3,970 | --- | --- | 1,240 | 23.6 | 7.8 | 7,040 | --- | --- | --- | --- |
| CHN-809 | 129 | 4/6/00 12:36 | --- | --- | --- | 4,410 | --- | --- | 1,430 | 23.6 | 7.9 | 7,380 | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 2** | | | | | | | | | | | | | | | | |
| CHN-809 | 116 | 5/15/00 10:53 | --- | --- | --- | 3,050 | --- | --- | 770 | 24.7 | 7.7 | 5,280 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 5/15/00 12:36 | --- | --- | --- | 3,180 | --- | --- | 840 | 24.4 | 7.6 | 5,390 | <1.0 | --- | --- | --- |
| TDW | NA | 5/19/00 12:49 | --- | --- | --- | 23 | --- | --- | 42 | --- | --- | --- | --- | --- | --- | --- |
| CHN-809 | 116 | 5/23/00 11:04 | --- | --- | --- | 1,870 | --- | --- | 320 | 26.6 | 8.5 | 3,070 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 5/23/00 13:09 | --- | --- | --- | 2,140 | --- | --- | 440 | 26.7 | 8.3 | 3,730 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 5/26/00 10:57 | --- | --- | --- | 2,710 | --- | --- | 560 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 129 | 5/26/00 13:02 | --- | --- | --- | 3,380 | --- | --- | 940 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 112 | 6/15/00 11:32 | --- | --- | --- | 1,640 | --- | --- | 210 | 30.2 | 8.9 | 2,540 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 6/15/00 14:31 | --- | --- | --- | 2,540 | --- | --- | 600 | 28.5 | 8.2 | 4,420 | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 6/16/00 11:39 | --- | --- | --- | --- | --- | --- | --- | 30.2 | 9.8 | 1,380 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 6/16/00 13:35 | --- | --- | --- | 1,350 | --- | --- | 270 | 28.6 | 9.3 | 2,540 | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 7/17/00 11:04 | --- | --- | --- | 782 | --- | --- | 130 | 27.7 | 9.8 | 1,520 | <1.5 | --- | --- | --- |
| CHN-809 | 129 | 7/17/00 12:52 | --- | --- | --- | 1,020 | --- | --- | 160 | 27.1 | 9.5 | 1,960 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 7/18/00 10:34 | --- | --- | --- | 1,510 | --- | --- | 220 | 25.3 | 8.9 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 7/18/00 12:34 | --- | --- | --- | 2,260 | --- | --- | 480 | 26.8 | 8.3 | --- | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 2 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-812 | NA | 8/2/00 13:30 | --- | --- | --- | 3,410 | --- | --- | 880 | --- | --- | --- | --- | --- | --- | --- |
| CHN-809 | 112 | 8/31/00 11:48 | --- | --- | --- | 4,360 | --- | --- | 1,500 | 24.8 | 8.3 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 8/31/00 13:42 | --- | --- | --- | 4,360 | --- | --- | 1,420 | 25.5 | 8.3 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 9/6/00 10:22 | --- | --- | --- | 4,620 | --- | --- | 1,630 | 23.6 | 8.4 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 9/6/00 12:04 | --- | --- | --- | 4,590 | --- | --- | 1,520 | 22.1 | 8.4 | --- | <1.0 | --- | --- | --- |
| **Cycle 3** | | | | | | | | | | | | | | | | |
| CHN-812 | NA | 9/11/00 13:52 | --- | --- | --- | 4,170 | --- | --- | 1,450 | 26.7 | 7.8 | --- | --- | --- | --- | --- |
| CHN-809 | 112 | 9/12/00 10:40 | --- | --- | --- | 4,310 | 21 | <1 | 1,400 | 26.9 | 7.9 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 9/12/00 12:34 | --- | --- | --- | 4,210 | 27 | 1 | 1,400 | 26.3 | 8.1 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 9/14/00 11:05 | --- | --- | --- | 4,410 | 271 | 6 | 1,400 | 25.6 | 8.3 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 9/14/00 12:57 | --- | --- | --- | 4,270 | 17 | <1 | 1,400 | 25.7 | 8.3 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 9/15/00 12:11 | --- | --- | --- | 3,910 | 186 | 4 | 1,100 | 28.1 | 7.9 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 9/15/00 14:18 | --- | --- | --- | 3,830 | 20 | <1 | 1,200 | 26.8 | 7.8 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 9/19/00 10:44 | --- | --- | --- | 2,270 | 194 | 3 | 390 | 25.6 | 9.2 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 9/19/00 12:40 | --- | --- | --- | 2,890 | 22 | <1 | 720 | 25.6 | 9.0 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 9/20/00 10:54 | --- | --- | --- | 3,970 | 311 | 6 | 1,300 | 25.2 | 7.8 | --- | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-810 | 129 | 9/20/00 12:45 | --- | --- | --- | 4,170 | 363 | 13 | 1,300 | 23.8 | 7.8 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 9/22/00 11:45 | --- | --- | --- | 1,790 | 342 | 15 | 220 | 27.2 | 8.8 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 9/22/00 13:33 | --- | --- | --- | 2,480 | 15 | <1 | 500 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 112 | 9/26/00 9:55 | --- | --- | --- | 3,140 | 66 | 2 | 740 | 21.8 | 8.2 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 9/26/00 11:39 | --- | --- | --- | 3,590 | 18 | 1 | 1,100 | 23.3 | 8.0 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 10/2/00 10:53 | --- | --- | --- | 1,120 | 16 | <1 | 87 | 24.4 | 9.9 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 10/2/00 12:55 | --- | --- | --- | 1,760 | 12 | <1 | 310 | 25.0 | 9.8 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 10/3/00 10:35 | --- | --- | --- | 2,330 | 23 | 1 | 380 | 22.9 | 9.5 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 10/3/00 12:30 | --- | --- | --- | 3,090 | 20 | 1 | 810 | 25.7 | 9.1 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 10/13/00 10:40 | --- | --- | --- | 1,780 | 66 | 3 | 230 | 21.3 | 10.5 | --- | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 10/13/00 12:51 | --- | --- | --- | 2,610 | 8 | 1 | 610 | 24.2 | 9.8 | --- | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 10/23/00 11:24 | --- | --- | --- | 670 | 16 | <1 | 81 | 24.2 | 10.3 | 1,130 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 10/23/00 13:34 | --- | --- | --- | 1,050 | 6 | <1 | 190 | 24.8 | 10.0 | 1,800 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 10/24/00 10:54 | --- | --- | --- | 1,530 | 9 | <1 | 210 | 22.1 | 9.0 | 2,150 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 10/24/00 13:12 | --- | --- | --- | 2,400 | 8 | <1 | 560 | 23.7 | 8.3 | 3,580 | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-809 | 112 | 11/2/00 11:56 | --- | --- | --- | 750 | 6 | <1 | 120 | 24.2 | 9.6 | 1,240 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 11/2/00 14:30 | --- | --- | --- | 940 | <2 | <1 | 150 | 24.0 | 9.4 | 1,630 | <1.0 | --- | --- | --- |
| TDW | NA | 11/2/00 14:40 | --- | --- | --- | 37 | 44 | 12 | 44 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 112 | 11/3/00 11:25 | --- | --- | --- | 1,430 | 6 | <1 | 150 | 23.9 | 9.0 | 1,940 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 11/3/00 11:26 | --- | --- | --- | 1,420 | 9 | <1 | 160 | 23.9 | 9.0 | 1,940 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 11/3/00 13:27 | --- | --- | --- | 2,150 | 4 | <1 | 420 | 23.6 | 8.4 | 3,120 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 11/3/00 13:28 | --- | --- | --- | 2,090 | <2 | <1 | 430 | 23.6 | 8.4 | 3,120 | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 11/13/00 12:50 | --- | --- | --- | 940 | 7 | <1 | 140 | 23.2 | 9.5 | 1,810 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 11/13/00 14:56 | --- | --- | --- | 1,040 | 6 | <1 | 170 | 22.5 | 9.3 | 2,090 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 11/20/00 12:35 | --- | --- | --- | 1,480 | 10 | <1 | 170 | 19.3 | 9.4 | 1,910 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 11/20/00 14:28 | --- | --- | --- | 2,120 | 3 | 45 | 410 | 19.6 | 9.0 | 2,980 | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 12/5/00 12:07 | --- | --- | --- | 810 | 6 | <1 | 150 | 17.8 | 9.5 | 1,290 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 12/5/00 14:20 | --- | --- | --- | 1,060 | 5 | <1 | 180 | 20.3 | 9.4 | 1,630 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 12/5/00 14:21 | --- | --- | --- | 1,060 | 3 | <1 | 180 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 129 | 12/6/00 14:43 | --- | --- | --- | 2,200 | 10 | 1 | 450 | 19.7 | 8.1 | 2,820 | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-809 | 112 | 12/18/00 13:28 | --- | --- | --- | 970 | 6 | <1 | 150 | 17.5 | 10.3 | 1,610 | <1.5 | --- | --- | --- |
| CHN-809 | 129 | 12/18/00 15:27 | --- | --- | --- | 1,060 | 7 | <1 | 190 | 23.9 | 8.8 | 1,640 | <1.0 | --- | --- | --- |
| CHN-810 | 112 | 12/19/00 12:45 | --- | --- | --- | 1,500 | 20 | 2 | 220 | 16.1 | 8.8 | 2,290 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 12/19/00 14:55 | --- | --- | --- | 2,180 | 9 | <1 | 450 | 16.2 | 8.3 | 3,590 | <1.5 | --- | --- | --- |
| CHN-810 | 112 | 1/22/01 12:38 | --- | --- | --- | 1,520 | 32 | <1 | 230 | 13.8 | 9.5 | 2,440 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 1/22/01 15:27 | --- | --- | --- | 2,230 | 18 | <1 | 480 | 15.7 | 10.4 | 3,670 | <1.0 | --- | --- | --- |
| CHN-809 | 112 | 1/24/01 12:49 | --- | --- | --- | 670 | 10 | <1 | 61 | 19.0 | 9.7 | 1,220 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 1/24/01 15:00 | --- | --- | --- | 1,020 | 3 | <1 | 150 | 20.4 | 9.3 | 1,870 | <1.0 | --- | --- | --- |
| CHN-812 | NA | 1/31/01 15:42 | --- | --- | --- | 97 | 718 | 31 | 100 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/1/01 10:15 | --- | --- | --- | 210 | 42 | 16 | 130 | 21.9 | 8.1 | 697 | <2.5 | --- | --- | --- |
| CHN-812 | NA | 2/1/01 15:20 | --- | --- | --- | 250 | 16 | 15 | 150 | 22.2 | 8.0 | 808 | <3.0 | --- | --- | --- |
| CHN-812 | NA | 2/2/01 10:06 | --- | --- | --- | 420 | 9 | 14 | 190 | | | | | | | |
| CHN-809 | 112 | 2/2/01 12:20 | --- | --- | --- | 1,890 | 27 | <1 | 440 | 21.9 | 7.7 | 1,180 | <2.5 | --- | --- | --- |
| CHN-809 | 129 | 2/2/01 14:50 | --- | --- | --- | 1,790 | 17 | <1 | 320 | 17.8 | --- | 3,140 | --- | --- | --- | --- |
| CHN-812 | NA | 2/2/01 15:33 | --- | --- | --- | 460 | 9 | 14 | 200 | --- | --- | --- | --- | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-812 | NA | 2/3/01 10:33 | --- | --- | --- | 700 | 8 | 12 | 240 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/3/01 18:48 | --- | --- | --- | 710 | 111 | 12 | 250 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/4/01 12:00 | --- | --- | --- | 860 | 7 | 9 | 260 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/4/01 19:24 | --- | --- | --- | 950 | 46 | 10 | 290 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/5/01 10:37 | --- | --- | --- | 1,100 | 18 | 9 | 310 | 23.6 | --- | 2,430 | --- | --- | --- | --- |
| CHN-809 | 112 | 2/5/01 13:00 | --- | --- | --- | 2,430 | 19 | <1 | 560 | 21.3 | 8.0 | 4,320 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 2/5/01 14:49 | --- | --- | --- | 2,420 | 28 | <1 | 500 | 21.8 | 8.0 | 4,130 | <1.0 | --- | --- | --- |
| CHN-812 | NA | 2/5/01 15:13 | --- | --- | --- | 1,180 | 19 | 9 | 320 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 2/7/01 11:00 | --- | --- | --- | 1,670 | 52 | 8 | 410 | 23.5 | 7.3 | 3,270 | <2.0 | --- | --- | --- |
| CHN-809 | 112 | 2/7/01 12:58 | --- | --- | --- | 3,040 | 35 | <1 | 750 | 23.2 | 7.9 | 5,040 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 2/7/01 15:06 | --- | --- | --- | 2,840 | <2 | <1 | 700 | 22.7 | 7.9 | 4,880 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 2/7/01 15:07 | --- | --- | --- | 2,850 | 16 | <1 | 700 | 22.7 | 7.9 | 4,880 | <1.0 | --- | --- | --- |
| CHN-812 | NA | 2/13/01 10:21 | --- | --- | --- | 3,080 | 9 | 6 | 770 | 19.8 | 7.4 | 5,640 | <2.0 | --- | --- | --- |
| CHN-809 | 112 | 2/13/01 12:20 | --- | --- | --- | 3,770 | 30 | <1 | 1,100 | 16.8 | 6.9 | 6,320 | <1.0 | --- | --- | --- |
| CHN-809 | 129 | 2/13/01 14:29 | --- | --- | --- | 3,780 | <2 | <1 | 1,100 | 20.6 | 6.9 | 6,290 | <1.0 | --- | --- | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 3 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-812 | NA | 2/20/01 15:41 | --- | --- | --- | 3,940 | 5 | 4 | 1,100 | 23.3 | 7.1 | 6,520 | <2.0 | --- | --- | --- |
| CHN-812 | NA | 3/1/01 14:07 | --- | --- | --- | 4,020 | <2 | 2 | 1,300 | 25.0 | 7.6 | 6,990 | <2.0 | --- | --- | --- |
| CHN-812 | NA | 3/5/01 14:05 | --- | --- | --- | 3,980 | <2 | 3 | 1,300 | 23.0 | 7.4 | 7,230 | <2.5 | --- | --- | --- |
| CHN-812 | NA | 3/16/01 15:12 | --- | --- | --- | 4,390 | <2 | 1 | 1,400 | 26.0 | 7.5 | 7,260 | <2.5 | --- | --- | --- |
| CHN-810 | 112 | 3/30/01 12:16 | --- | --- | --- | 4,270 | 15 | 1 | 1,500 | 22.0 | 7.0 | 7,420 | <1.0 | 0.1 | 0.15 | --- |
| CHN-810 | 129 | 3/30/01 14:38 | --- | --- | --- | 4,110 | <2 | <1 | 1,400 | 23.6 | 7.0 | 7,180 | <1.0 | 0.1 | 0.1 | --- |
| CHN-809 | 112 | 4/2/01 12:37 | --- | --- | --- | 4,260 | <2 | <1 | 1,400 | 22.6 | 7.0 | 7,310 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 4/2/01 12:38 | --- | --- | --- | 4,270 | 6 | <1 | 1,400 | --- | --- | --- | --- | --- | --- | --- |
| CHN-809 | 129 | 4/2/01 13:56 | --- | --- | --- | 4,390 | <2 | <1 | 1,400 | 22.4 | 7.0 | 7,220 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 4/2/01 13:57 | --- | --- | --- | 4,270 | <2 | <1 | 1,400 | --- | --- | --- | --- | --- | --- | --- |
| CHN-812 | NA | 4/2/01 14:27 | --- | --- | --- | 3,900 | <2 | 2 | 1,400 | 25.0 | 7.4 | 7,170 | <1.0 | 0.2 | <0.05 | --- |
| **Cycle 4** | | | | | | | | | | | | | | | | |
| CHN-809 | 112 | 4/4/01 11:16 | --- | --- | --- | 4,230 | 12 | <1 | 1,400 | 23.3 | 7.5 | 7,130 | <1.0 | 0.2 | <0.05 | --- |
| CHN-809 | 129 | 4/4/01 13:36 | --- | --- | --- | 4,260 | 9 | <1 | 1,400 | 23.4 | 7.5 | 7,220 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 4/6/01 11:50 | --- | --- | --- | 3,800 | 11 | <1 | 1,200 | 24.7 | 7.8 | 6,470 | <1.0 | 0.15 | <0.05 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-809 | 129 | 4/6/01 13:10 | --- | --- | --- | 3,910 | 9 | 1 | 1,200 | 24.5 | 7.7 | 6,880 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 4/9/01 11:42 | --- | --- | --- | 2,680 | 9 | <1 | 570 | 26.9 | 8.0 | 4,550 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 4/9/01 13:57 | --- | --- | --- | 3,220 | 7 | <1 | 890 | 27.7 | 7.7 | 5,610 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 4/10/01 11:36 | --- | --- | --- | 4,120 | 17 | 2 | 1,400 | 25.1 | 7.6 | 7,140 | <1.0 | 0.15 | 0.05 | --- |
| CHN-810 | 129 | 4/10/01 14:07 | --- | --- | --- | 4,200 | 9 | <1 | 1,400 | 27.1 | 7.5 | 7,160 | <1.0 | 0.05 | 0.05 | --- |
| CHN-811A | Composite[a] | 4/11/01 13:10 | --- | --- | --- | 4,380 | 23 | <1 | 1,500 | 22.6 | 7.5 | 7,580 | <1.0 | <0.05 | 0.2 | --- |
| TDW | NA | 4/11/01 14:27 | --- | --- | --- | 140 | 114 | 8 | 170 | 23.0 | 8.3 | 237 | 7.2 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 4/12/01 11:14 | --- | --- | --- | 2,060 | 25 | <1 | 320 | 24.7 | 8.3 | 3,350 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 4/12/01 13:36 | --- | --- | --- | 2,770 | 4 | <1 | 670 | 26.3 | 7.9 | 4,720 | <1.0 | 0.2 | <0.05 | --- |
| CHN-811A | Composite[b] | 4/13/01 8:39 | --- | --- | --- | 4,310 | 6 | <1 | 1,500 | 24.5 | 7.5 | 7,400 | <2.0 | 0.1 | 0.1 | --- |
| CHN-810 | 112 | 4/17/01 11:29 | --- | --- | --- | 3,300 | 21 | 2 | 870 | 20.5 | 7.8 | 5,470 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 4/17/01 13:51 | --- | --- | --- | 3,790 | 3 | <1 | 1,200 | 22.5 | 7.6 | 6,430 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 4/20/01 13:50 | --- | --- | --- | 2,030 | <2 | <1 | 390 | 25.2 | 8.7 | 3,310 | <1.0 | 0.05 | <0.05 | --- |
| CHN-810 | 112 | 4/23/01 11:29 | --- | --- | --- | 2,550 | 20 | <1 | 530 | 24.9 | 8.0 | 4,280 | <1.0 | 0.05 | <0.05 | --- |
| CHN-810 | 129 | 4/23/01 14:28 | --- | --- | --- | 3,390 | 9 | 1 | 960 | 25.9 | 7.7 | 5,680 | <1.0 | 0.05 | <0.05 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manganese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temperature (° C) | Field pH (su) | Field specific conductance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-810 | 112 | 5/2/01 11:45 | --- | 0.69 | 0.58 | 2,080 | 177 | 4 | 310 | 24.6 | 8.2 | 3,120 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 5/2/01 11:46 | --- | 0.69 | 0.58 | 2,060 | 12 | <1 | 320 | --- | --- | --- | --- | --- | --- | --- |
| CHN-810 | 129 | 5/2/01 14:17 | --- | 0.32 | 0.28 | 3,000 | 6 | <1 | 780 | 23.3 | 7.7 | 4,950 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 5/2/01 14:18 | --- | 0.32 | 0.28 | 3,050 | 7 | <1 | 760 | --- | --- | --- | --- | --- | --- | --- |
| CHN-800 | 109 | 5/9/01 11:22 | --- | <0.02 | <0.01 | 3,970 | 375 | 10 | 2,300 | 27.4 | 7.7 | 10,100 | <1.0 | 1.0 | 0.25 | --- |
| CHN-810 | 112 | 5/18/01 11:04 | --- | 0.69 | 0.55 | 1,690 | 16 | 1 | 260 | 27.1 | 8.6 | 2,510 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 5/18/01 13:16 | --- | 0.42 | 0.36 | 2,610 | 29 | 2 | 640 | 25.6 | 7.8 | 4,150 | <1.0 | 0.1 | <0.05 | --- |
| CHN-811A | Composite[b] | 5/22/01 12:26 | --- | 0.02 | <0.01 | 4,200 | 11 | <1 | 1,500 | 24.1 | 7.6 | 7,550 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 5/23/01 11:40 | --- | 0.88 | 0.16 | 660 | <2 | <1 | 120 | 24.3 | 9.3 | 1,330 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 5/23/01 11:41 | --- | 0.87 | 0.15 | 660 | 3 | <1 | 120 | --- | --- | --- | --- | --- | --- | --- |
| TDW | NA | 5/23/01 13:25 | --- | 0.24 | <0.01 | 33 | <2 | 3 | 50 | 27.8 | 8.5 | 215 | 6.8 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 5/23/01 13:50 | --- | 0.68 | 0.15 | 1,110 | 2 | <1 | 220 | 27.4 | 8.9 | 2,010 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 5/31/01 12:43 | --- | 0.90 | 0.32 | 600 | <2 | <1 | 99 | 29.4 | 9.4 | 1,220 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 5/31/01 15:08 | --- | 0.72 | 0.28 | 880 | <2 | <1 | 130 | 26.5 | 9.1 | 1,600 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 6/5/01 11:20 | --- | 0.56 | 0.58 | 1,470 | <2 | <1 | 190 | 26.0 | 8.5 | 2,210 | <1.0 | 0.1 | <0.05 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-810 | 129 | 6/5/01 13:55 | --- | 0.36 | 0.37 | 2,230 | <2 | <1 | 460 | 29.9 | 8.0 | 3,570 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 6/26/01 11:56 | --- | 0.69 | 0.43 | 740 | 2 | 2 | 130 | 28.6 | 9.2 | 1,480 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 6/26/01 14:15 | --- | 0.57 | 0.35 | 1,000 | <2 | 2 | 180 | 26.7 | 9.0 | 1,850 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 6/27/01 12:29 | --- | 0.44 | 0.46 | 1,470 | <2 | 2 | 170 | 25.8 | 8.5 | 2,150 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 6/27/01 14:30 | --- | 0.27 | 0.27 | 2,320 | <2 | 2 | 463 | 25.3 | 7.9 | 3,580 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 7/17/01 10:55 | --- | --- | --- | --- | --- | --- | --- | 28.3 | 9.2 | 1,610 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 7/17/01 13:09 | --- | 0.45 | 0.30 | 995 | <2 | <1 | 194 | 26.6 | 8.9 | 1,910 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 7/19/01 11:40 | --- | 0.36 | 0.32 | 1,440 | 5 | <1 | 206 | 27.6 | 8.5 | 2,210 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 7/19/01 13:46 | --- | 0.20 | 0.19 | 2,270 | 2 | 1 | 486 | 25.2 | 7.9 | 3,680 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 8/7/01 12:24 | --- | 0.56 | 0.34 | 679 | 4 | <1 | 114 | 28.8 | 9.5 | 1,440 | --- | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 8/7/01 14:56 | --- | 0.41 | 0.26 | 974 | 4 | <1 | 159 | 28.7 | 9.3 | 1,840 | --- | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 8/8/01 12:22 | --- | 0.25 | 0.25 | 1,450 | 3 | <1 | 193 | 29.7 | 9.0 | 2,020 | --- | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 8/8/01 14:40 | --- | 0.14 | 0.14 | 2,250 | 4 | 1 | 492 | 27.8 | 8.5 | 3,270 | --- | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 8/22/01 12:10 | --- | 0.49 | 0.33 | 701 | 4 | <1 | 120 | 27.8 | 9.3 | 1,450 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 8/22/01 14:40 | --- | 0.36 | 0.24 | 962 | 2 | <1 | 160 | 27.2 | 9.2 | 1,860 | <1.0 | 0.1 | <0.05 | --- |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-810 | 129 | 8/23/01 14:40 | --- | 0.10 | 0.10 | 2,300 | 7 | 1 | 490 | 25.0 | 8.3 | 3,680 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 9/18/01 12:02 | --- | 0.40 | 0.28 | 687 | 4 | <1 | 110 | 25.7 | 9.6 | 1,390 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 9/18/01 14:43 | --- | 0.27 | 0.19 | 976 | 2 | <1 | 160 | 27.7 | 9.3 | 1,850 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 112 | 9/19/01 11:46 | --- | 0.11 | 0.10 | 1,440 | 5 | <1 | 210 | 26.1 | 8.7 | 2,170 | <1.0 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 9/19/01 14:08 | --- | 0.06 | 0.05 | 2,220 | 5 | <1 | 460 | 25.2 | 8.2 | 3,570 | <1.0 | 0.1 | <0.05 | --- |
| CHN-809 | 112 | 10/15/01 11:52 | 0.89 | 0.26 | 0.25 | 800 | 4 | M | 152 | 25.6 | 9.1 | 1,590 | <1.5 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 10/15/01 14:12 | 0.86 | 0.19 | 0.17 | 1,020 | 5 | M | 182 | 24.9 | 9.0 | 1,930 | 1.0 | --- | --- | --- |
| CHN-810 | 112 | 10/16/01 12:18 | 0.74 | 0.04 | 0.04 | 1,480 | 7 | <1 | 240 | 25.8 | 9.2 | 2,350 | <1.5 | 0.1 | <0.05 | --- |
| CHN-810 | 129 | 10/16/01 14:17 | 0.84 | <0.02 | <0.01 | 2,300 | 6 | <1 | 510 | 24.7 | 8.7 | 3,680 | 1.0 | 0.1 | <0.05 | --- |
| CHN-812 | NA | 11/8/01 14:40 | 0.49 | <0.02 | <0.01 | 106 | 7 | 20 | 87 | 25.1 | 7.9 | 399 | <1.0 | <0.05 | <0.05 | 56 |
| CHN-812 | NA | 11/9/01 8:40 | 0.31 | 0.12 | 0.03 | 270 | 4 | 17 | 160 | 21.7 | 9.5 | 834 | <1.0 | 0.05 | <0.05 | 160 |
| CHN-810 | 112 | 11/9/01 13:00 | 1.9 | <0.02 | <0.01 | 2,890 | 144 | 6 | 850 | 23.5 | 7.2 | 5,140 | <1.0 | 0.2 | 0.1 | --- |
| CHN-810 | 129 | 11/9/01 15:20 | 1.5 | <0.02 | <0.01 | 2,920 | 56 | M | 693 | 22.0 | 7.1 | 4,670 | <1.0 | 0.1 | <0.05 | --- |
| CHN-812 | NA | 11/9/01 16:10 | 0.30 | 0.14 | 0.04 | 347 | 3 | 15 | 180 | 23.2 | 7.4 | 1,030 | <1.0 | 0.05 | <0.05 | 190 |
| CHN-812 | NA | 11/10/01 10:20 | 0.43 | 0.09 | 0.04 | 549 | 3 | 11 | 215 | 23.0 | 8.4 | 1,300 | <1.0 | <0.05 | <0.05 | 260 |

**Appendix 1.** Dissolved inorganic constituent concentrations and water-quality characteristics measured in samples collected during aquifer storage recovery testing in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; su, standard units; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; ANC, acid neutralizing capacity; $CaCO_3$, calcium carbonate; ---, data not available; NA, not applicable; E, estimated; TDW, treated drinking water; <, less than; M, presence verified, but not quantified]

| Sample location (fig. 1; table 2) | Sample depth, m bls | Date/time (EST) | Nitrogen ammonia (mg/L) | Nitrate + nitrate, dissolved (mg/L) | Nitrogen, nitrite (mg/L) | Boron, dissolved (µg/L) | Iron, dissolved (µg/L) | Manga-nese, dissolved (µg/L) | Strontium, dissolved (µg/L) | Water temper-ature (° C) | Field pH (su) | Field specific conduc-tance (µS/cm) | Field dissolved oxygen, mg/L | Field iron, mg/L | Field sulfide, mg/L | Field chloride, mg/L |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cycle 4 (Continued)** | | | | | | | | | | | | | | | | |
| CHN-812 | NA | 11/10/01 16:17 | 0.38 | 0.11 | 0.04 | 505 | 4 | 12 | 208 | 23.1 | 8.0 | 1,420 | <1.0 | 0.05 | <0.05 | 320 |
| CHN-812 | NA | 11/11/01 12:20 | 0.56 | 0.05 | 0.02 | 736 | 4 | 10 | 243 | 24.4 | 7.9 | 1,720 | <1.0 | <0.05 | --- | 420 |
| CHN-809 | 112 | 11/13/01 13:50 | E1.6 | <0.02 | <0.01 | 2,670 | 27 | M | 673 | 23.0 | 8.3 | 4,570 | <1.0 | 0.05 | <0.05 | --- |
| CHN-809 | 129 | 11/13/01 16:20 | 1.3 | <0.02 | <0.01 | 2,520 | 18 | M | 553 | 20.9 | 8.3 | 4,340 | <1.0 | 0.05 | <0.05 | --- |
| CHN-812 | NA | 11/13/01 17:21 | 0.74 | <0.02 | <0.01 | 1,310 | 5 | 7 | 331 | 22.0 | 8.2 | 2,440 | <1.0 | <0.05 | <0.05 | 500 |
| CHN-812 | NA | 11/27/01 16:30 | 0.59 | <0.02 | 0.02 | 1,880 | 28 | 25 | 600 | 24.4 | 8.1 | 3,570 | <1.5 | --- | --- | --- |
| CHN-812 | NA | 12/6/01 12:30 | 1.3 | <0.02 | <0.01 | 3,490 | 59 | 4 | 1,020 | 25.6 | 7.5 | 6,480 | <1.0 | 0.1 | <0.05 | >2,500 |
| CHN-809 | 112 | 12/18/01 12:14 | 1.9 | <0.02 | <0.01 | 4,050 | 65 | 1 | 1,400 | 20.0 | 7.5 | 7,070 | <1.0 | 0.1 | 0.1 | --- |
| CHN-812 | NA | 12/18/01 13:27 | 1.3 | <0.02 | <0.01 | 3,710 | 72 | 5 | 1,190 | 24.2 | 7.5 | 6,700 | <1.5 | 0.1 | <0.05 | --- |
| CHN-809 | 129 | 12/18/01 14:39 | 1.6 | <0.02 | <0.01 | 4,080 | 43 | <1 | 1,330 | 21.5 | 7.5 | 6,950 | <1.0 | <0.05 | <0.05 | --- |
| CHN-809 | 112 | 2/14/02 13:46 | 1.8 | <0.02 | <0.01 | 4,430 | 54 | <2 | 1,500 | 21.3 | 7.4 | 7,250 | <1.0 | 0.1 | 0.1 | --- |
| CHN-809 | 129 | 2/14/02 15:48 | 1.5 | <0.02 | <0.01 | 4,450 | 47 | <2 | 1,420 | 21.3 | 7.4 | 7,260 | <1.0 | <0.05 | <0.05 | --- |
| CHN-810 | 112 | 2/15/02 12:55 | 1.6 | <0.02 | <0.01 | 4,340 | 62 | <2 | 1,410 | 16.2 | 7.4 | 7,680 | <1.0 | --- | --- | --- |
| CHN-810 | 129 | 2/15/02 15:24 | 2.0 | 0.03 | <0.01 | 4,310 | 97 | <2 | 1,680 | 20.6 | 7.3 | 7,240 | <1.0 | 0.1 | <0.05 | --- |
| CHN-812 | NA | 2/15/02 16:06 | 1.6 | <0.02 | <0.01 | 4,250 | 76 | <2 | 1,470 | 21.2 | 7.4 | 7,280 | <1.0 | 0.1 | 0.1 | --- |

[a]Sample was collected from within the cased portion of the well after purging approximately 60 percent of one casing volume of water.

[b]Sample was collected from within the cased portion of the well after purging three casing volumes of water.

**Appendix 2.** Dissolved trihalomethane and chloride concentrations measured in samples collected during cycle 4 of the aquifer storage recovery study in Charleston, South Carolina.

[m bls, meter below land surface; EST, Eastern Standard Time; µg/L, microgram per liter; mg/L, milligram per liter; TDW, treated drinking water; NA, not applicable; ---, data not available; UPZ, upper production zone; <, less than; LPZ, lower production zone]

| Sample location | Sample depth, m bls | Days (from start of injection) | Date/Time (EST) | Total trihalomethane[a] (µg/L) | Dissolved chloride[b], mg/L | Chloroform, relative percent[c] | Dissolved chloride, relative percent[c] |
|---|---|---|---|---|---|---|---|
| TDW | NA | 8.97 | 4/11/01 14:27 | --- | 60 | --- | NA |
| TDW | NA | 9.89 | 4/12/01 12:37 | 57 | --- | NA | --- |
| CHN-809 UPZ | 112 | 9.84 | 4/12/01 11:14 | 14 | 680 | 0.56 | 0.63 |
| CHN-809 UPZ | 112 | 58.90 | 5/31/01 12:43 | 28 | 190 | 1.12 | 0.91 |
| CHN-809 UPZ | 112 | 84.87 | 6/26/01 11:56 | 22 | 230 | 0.88 | 0.89 |
| CHN-809 UPZ | 112 | 105.82 | 7/17/01 10:55 | 20 | --- | 0.80 | --- |
| CHN-809 UPZ | 112 | 126.89 | 8/7/01 12:24 | 19 | 210 | 0.76 | 0.90 |
| CHN-809 UPZ | 112 | 141.88 | 8/22/01 12:10 | 14 | --- | 0.56 | --- |
| CHN-809 UPZ | 112 | 168.87 | 9/18/01 12:02 | 19 | 230 | 0.76 | 0.89 |
| CHN-809 UPZ | 112 | 195.86 | 10/15/01 11:52 | 20 | 280 | 0.80 | 0.86 |
| CHN-809 UPZ | 112 | 224.95 | 11/13/01 13:50 | --- | 1,050 | --- | 0.42 |
| CHN-809 UPZ | 112 | 259.98 | 12/18/01 14:39 | <4.0 | --- | NA | --- |
| CHN-809 LPZ | 129 | 9.94 | 4/12/01 13:36 | 9.2 | 1,100 | 0.37 | 0.39 |
| CHN-809 LPZ | 129 | 59.00 | 5/31/01 15:08 | 23 | 270 | 0.92 | 0.86 |
| CHN-809 LPZ | 129 | 84.96 | 6/26/01 14:15 | 17 | 340 | 0.68 | 0.82 |
| CHN-809 LPZ | 129 | 105.92 | 7/17/01 13:09 | 3 | 350 | 0.12 | 0.82 |
| CHN-809 LPZ | 129 | 126.99 | 8/7/01 14:56 | 15 | 320 | 0.60 | 0.83 |
| CHN-809 LPZ | 129 | 141.98 | 8/22/01 14:40 | 13 | --- | 0.52 | --- |
| CHN-809 LPZ | 129 | 168.98 | 9/18/01 14:43 | 16 | 340 | 0.64 | 0.82 |
| CHN-809 LPZ | 129 | 195.96 | 10/15/01 14:12 | 15 | 360 | 0.60 | 0.81 |
| CHN-809 LPZ | 129 | 225.05 | 11/13/01 16:20 | --- | 960 | --- | 0.47 |
| CHN-810 UPZ | 112 | 63.84 | 6/5/01 11:20 | 18 | 410 | 0.72 | 0.78 |
| CHN-810 UPZ | 112 | 107.85 | 7/19/01 11:40 | 15 | 410 | 0.60 | 0.78 |
| CHN-810 UPZ | 112 | 127.88 | 8/8/01 12:22 | 20 | 400 | 0.80 | 0.79 |
| CHN-810 UPZ | 112 | 142.90 | 8/23/01 12:42 | 16 | --- | 0.64 | --- |
| CHN-810 UPZ | 112 | 169.86 | 9/19/01 11:46 | 22 | 420 | 0.88 | 0.78 |
| CHN-810 UPZ | 112 | 196.88 | 10/16/01 12:18 | --- | 440 | --- | 0.77 |
| CHN-810 UPZ | 112 | 220.91 | 11/9/01 13:00 | <4.0 | 1,150 | NA | 0.37 |
| CHN-810 LPZ | 129 | 63.95 | 6/5/01 13:55 | 10 | 750 | 0.40 | 0.59 |
| CHN-810 LPZ | 129 | 107.94 | 7/19/01 13:46 | 8.2 | 770 | 0.33 | 0.58 |
| CHN-810 LPZ | 129 | 127.98 | 8/8/01 14:40 | 12 | 760 | 0.48 | 0.59 |
| CHN-810 LPZ | 129 | 142.98 | 8/23/01 14:40 | 10 | --- | 0.40 | --- |
| CHN-810 LPZ | 129 | 169.96 | 9/19/01 14:08 | 13 | 800 | 0.52 | 0.56 |
| CHN-810 LPZ | 129 | 196.96 | 10/16/01 14:17 | 15 | 770 | 0.60 | 0.58 |
| CHN-810 LPZ | 129 | 221.01 | 11/9/01 15:20 | <4.0 | 1,050 | NA | 0.42 |
| CHN-812 | NA | 219.98 | 11/8/01 14:40 | 8.5 | 48 | 0.34 | 0.99 |

**Appendix 2.** Dissolved trihalomethane and chloride concentrations measured in samples collected during cycle 4 of the aquifer storage recovery study in Charleston, South Carolina.—Continued

[m bls, meter below land surface; EST, Eastern Standard Time; µg/L, microgram per liter; mg/L, milligram per liter; TDW, treated drinking water; NA, not applicable; ---, data not available; UPZ, upper production zone; <, less than; LPZ, lower production zone]

| Sample location | Sample depth, m bls | Days (from start of injection) | Date/Time (EST) | Total trihalomethane[a] (µg/L) | Dissolved chloride[b], mg/L | Chloroform, relative percent[c] | Dissolved chloride, relative percent[c] |
|---|---|---|---|---|---|---|---|
| CHN-812 | NA | 220.73 | 11/9/01 8:40 | 4.5 | 150 | 0.18 | 0.93 |
| CHN-812 | NA | 221.04 | 11/9/01 16:10 | <4.0 | 190 | NA | 0.91 |
| CHN-812 | NA | 221.80 | 11/10/01 10:20 | <4.0 | 290 | NA | 0.85 |
| CHN-812 | NA | 222.05 | 11/10/01 16:17 | <4.0 | 260 | NA | 0.87 |
| CHN-812 | NA | 222.88 | 11/11/01 12:20 | 4.1 | 360 | 0.16 | 0.81 |
| CHN-812 | NA | 225.09 | 11/13/01 17:21 | <4.0 | 580 | NA | 0.69 |
| CHN-812 | NA | 239.06 | 11/27/01 16:30 | <4.0 | 790 | NA | 0.57 |
| CHN-812 | NA | 240.87 | 11/29/01 11:55 | <4.0 | --- | NA | --- |
| CHN-812 | NA | 247.89 | 12/6/01 12:30 | <4.0 | 1,570 | NA | 0.13 |
| CHN-812 | NA | 259.93 | 12/18/01 13:27 | <4.0 | 1,560 | NA | 0.14 |

[a]Total trihalomethanes consist of chloroform, bromoform, bromodichloromethane, and chlorodibromomethane. Chloroform was the only constituent that contributed to the total trihalomethane concentration in ground-water samples. The chloroform concentration in the 4/12/2001 treated drinking-water sample was 25 µg/L.

[b]Average chloride concentrations for all treated drinking-water samples collected during this investigation was 27 mg/L. Native aquifer chloride concentration was 1,800 mg/L.

[c]Percent of sample concentration that represents the contribution of chloroform or dissolved chloride from treated drinking water.