AQUIFER STORAGE AND RECOVERY MYRTLE BEACH, SOUTH CAROLINA

PHASE II: A HYDROLOGIC, GEOCHEMICAL, AND ECONOMIC INVESTIGATION

by

Joffre E. Castro

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ABSTRACT

Stringent national water quality regulations have forced Horry County, S.C., water utilities and municipalities to upgrade their water treatment facilities. In this county, during the last seven years, surface water has replaced ground water as the major source of drinking water, reversing a 100-year practice. Although quality and quantity of the drinking water have improved, water costs have risen. Large investment in peak treatment capacity that is rarely utilized has also contributed to higher water rates.

To develop new water management alternatives that could curtail rising water costs in the region, an aquifer storage and recovery (ASR) project was undertaken. A prototype ASR system was tested in Myrtle Beach to study its hydrologic, geochemical, and economic applicability. Ten injection tests were made between February 1991 and October 1992. The first nine injection tests were a series of short-term tests designed to evaluate the hydrologic changes in the aquifer resulting from the injection of treated surface water. The other test was a long-term test set up to investigate the geochemical system and the water quality changes of the recovered water. During this last test, approximately 8 million gallons of treated surface water were injected into an aquifer of poor water quality. After several months of storage, the water was successfully recovered through the same well.

The geochemical modeling of the system revealed that mostly it was mixing and, to a lesser degree, chemical reactions that governed the quality of the recovered water. The most important geochemical processes modeled were pyrite oxidation, calcite dissolution, and calcium-sodium exchange. Near the well bore, chemical reactions prevailed and developed under aerobic conditions. Farther away from the well, mixing was the prevalent process and chemical reaction, which developed under anaerobic conditions, was a subordinate process.

This study demonstrated that the ASR concept is applicable in unconsolidated sediments of the South Carolina Coastal Plain. Moreover, it showed that ASR operational systems could inexpensively augment daily flows in a distribution system and thereby provide for long-term and emergency demands. It additionally suggested that the total unit cost of an ASR system is no more than half that for expansion of a treatment plant of similar capacity.

KEY WORDS: Aquifer Storage Recovery; aquifer injection; geochemical modeling; cost analysis.

INTRODUCTION

An Aquifer Storage Recovery (ASR) concept and its economic applicability in unconsolidated sediments of the Atlantic Coastal Plain were investigated at Myrtle Beach, Horry County, South Carolina. The project was a cooperative effort of the South Carolina Department of Natural Resources, Water Resources Division; the Division of Local Governments; the City of Myrtle Beach; and the Grand Strand Water and Sewer Authority.

Continuous growth along the Grand Strand and stringent national water quality regulations have forced local water utilities and municipalities to upgrade their water treatment facilities. Today in Horry County, surface water has replaced ground water as the major source of drinking water, reversing a 100-year practice. The Intracoastal Waterway and Bull Creek presently meet most of the water needs of Horry County.

Although treated surface water meets the Environmental Protection Agency's drinking water standards and the supply is plentiful, the unit cost has increased. Furthermore, large investments in peak treatment capacity that is rarely utilized, because of seasonal water demands, have also contributed to higher water costs. An alternative is now available to systems with seasonal demands. ASR systems could provide Horry County cities and others along the eastern seaboard with a simple and innovative solution to the problem of rising water costs. Plant and transmission line expansions, consequently, could be deferred until such times as growth in population and water demand fully utilizes their capacity the year around.

The ASR concept, although new to South Carolina, is presently being investigated or applied in more than 60 sites throughout the United States (Pyne, 1994). Canada (Pyne, 1995), Colombia (Molano and others, 1994), England, the Netherlands, Switzerland, Greece, France (Connorton and McIntosh, 1994), Holland (Stakelbeek and others, 1994), Israel (Harpz and Bear, 1963), Italy (Drusiani and others, 1994), Japan (Abiko and Katsuragi, 1994), China (Peimin and others, 1994), Kuwait (Mukhopadhyay and others, 1994), India (Athavale and others, 1994), and Australia (Dillon and others, 1994) have injection facilities. Other applications include aquifer recharge (Bear and Jacobs, 1965), storage of thermal energy (Molz and others, 1979), aquifer restoration (Rebhun and Schwarz, 1968; Brown and Silvey, 1977), disposal of treated effluent (Hamlin, 1987), and the development of salinity barriers. The Myrtle Beach ASR project utilized a confined aquifer, without jeopardizing its integrity, as a reservoir where high-quality surface water was periodically stored and removed from the ground water system (Castro, 1987).

In developing the Myrtle Beach ASR project, there was always a keen interest in balancing its scientific aspect with its practicality. Thus, the project objectives were to (a) study the hydrogeologic and geochemical suitability of coastal plain aquifers to store significant quantities of treated surface water; and (b) determine the economic feasibility of ASR by comparing its annual cost to that of a water treatment plant expansion.

PREVIOUS AND PRESENT TASKS

Owing to the complexity of the investigation, the Myrtle Beach ASR project was divided into two phases. During Phase I, hydrogeologic and geochemical studies of local aquifers were carried out. Continuous drill cores were obtained through the Black Creek, Middendorf, and Cape Fear Formations to a depth of 1,327 feet; and a Middendorf well was completed. A geochemical analysis, using a column test, was implemented to assess changes in hydraulic conductivity, pretreatment procedures, and chemical compatibility of injected and aquifer water (CH2MHILL, 1990). Following the recommendations of the Phase I Report (Castro and others, 1995; CH2MHILL, 1990), a Phase II project was developed, which is the subject of this report. During this phase of the study, a pilot ASR site was set up and several injection, storage, and recovery cycles were completed. Extensive hydrological and geochemical analyses of injection effects on the aquifer and retrieved water, which occurred during storage, were carried out. Geochemical processes were identified from the water quality data using NETPATH, a U.S. Geological Survey geochemical code. Ground water flow, solute transport, and energy transport (Petkewich, 1992) were determined by developing a solute transport model using SUTRA, a U.S. Geological Survey flow, solute, and energy transport code. Finally, an annual cost analysis was completed.

TESTING SITE AND FACILITIES

The prototype ASR system tested in Myrtle Beach was made up of three basic components: (a) a source of treated surface water (b) a suitable aquifer, and (c) an injection-recovery well (Fig. 1). Treated surface water, produced at the city's treatment plant, was routed through the distribution system to the injection site. The water was injected into the aquifer through the well's riser, using the line pressure of the distribution system, normally 50 pounds per square inch. The injected water remained stored in the aquifer for several days or months, depending on the test. Finally, at the end of the storage period, the well was continuously pumped and an equivalent volume of injected water, plus some of the native ground water was removed. During all of the field tests, the same well was used for both injection and recovery (dual-purpose well).

Throughout this report, the period of time during which water from the city's distribution system was injected in the aquifer is called the ***injection period***; the period of time during which the injected water remained in the aquifer is the ***storage period***; and the time during which the injected water was pumped and discharged off the site is the ***recovery period***.

SOURCE OF INJECTED WATER

The water treatment plant at Myrtle Beach, which was constructed in 1988, has its raw water intake in the Atlantic Intracoastal Waterway (AICW). The AICW is composed of a series of natural and manmade canals, built by the Army Corps of Engineers, that extends from New York to Florida. Within Horry County, and for only 30 miles, the AICW is freshwater. The city of Myrtle Beach has used this special characteristic of the AICW to develop its drinking water system.

Raw water from the AICW is pumped into a wet well, passed through a pre-ozonization contact chamber, conveyed to the rapid-mix basin for coagulation, and then moved to a flocculation basins and sedimentation tanks. Next, the water is sent to a post-ozonization, filtration, and feeding facilities and then sent into two clearwells where the water is stored prior to distribution. Alum and activated carbon are used for color removal; a polymer is used for coagulation and filtration; lime and caustic soda are used to adjust the alkalinity and pH; copper sulfate is used to control algae growth; chlorine and ammonia are used for disinfection; hydrofluorosilic acid is added to provide fluoride; and pyrophosphate and zinc are used as corrosion inhibitors (Ted Welch, City of Myrtle Beach written communication, 1994).

The finished water is low in dissolved-solids concentration (130 milligrams per liter), sodium (23 mg/L), and chloride (20 mg/L). During the project, pH increased from 7.4 to 8.0. Some of the constituents and properties of the product water fluctuate seasonally, responding to variations in the raw-water quality. The water temperature can be as low as 10°C during winter and exceed 30°C during summer. Dissolved-oxygen, aluminum, silica, and nitrate concentrations also vary seasonally, and minor variations in chloride and sodium concentrations occur. Representative concentrations of the finished water are given in Table 1.

SUITABLE AQUIFER

Following the recommendations from the Phase I study (Castro and others, 1995), the aquifers of the Black Creek Formation were selected to conduct the Phase II investigation. These aquifers have been extensively studied in Swift and Heron, 1969; Spigner and others, 1977; Colqhoun and others, 1983; South Carolina Water Resources Commission, 1983; Zack, 1977; Zack, 1980; Pelletier, 1985; Reid and others, 1986; Castro, 1987; Castro and Hockensmith, 1987; Zack and Roberts, 1988; Gohn, 1988; CH2MHILL, 1990; Hockensmith and Castro, 1993; and Castro and others, 1995.

The Black Creek Formation, which overlies the Middendorf Formation and underlies the Peedee Formations, is between 288 and 961 feet bls (below land surface) at the test site. The sediments consist mostly of

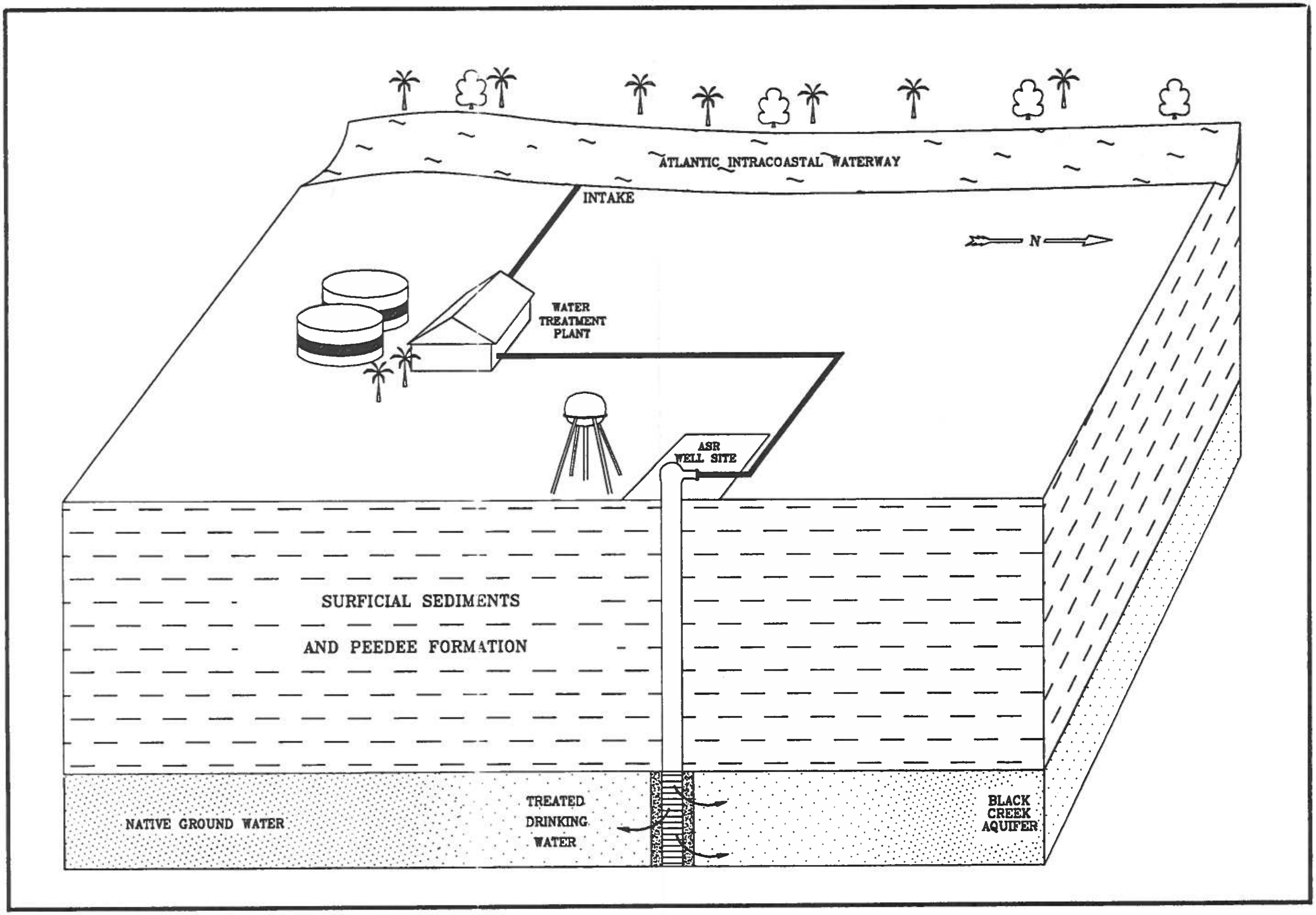


Figure 1. The Aquifer Storage Recovery (ASR) system at Myrtle Beach, S.C.

Table 1. Representative chemical composition of treated surface water and natural ground water

CONSTITUENT OR PROPERTY	TREATED SURFACE WATER (Sample 33T01, 5/27/92)	GROUND WATER (Well 5S-i8)
Temperature (degrees C)	32.2	24
Specific conductance (μsiemens/cm)	242	1,360
pH (standard units)	7.9	8.4
Dissolved oxygen (mg/L)	6.9	<0.01
Alkalinity, as bicarbonate (mg/L)	42.6	672
Calcium (mg/L)	12	2.4
Magnesium (mg/L)	1.4	1.62
Sodium (mg/L)	28	325
Potassium (mg/l)	3	6
Chloride (mg/L)	18	141
Sulfate (mg/L)	41	2.1
Fluoride (mg/L)	0.9	4.2
Silica (mg/L)	5.1	14
Iron (mg/L)	0.04	0.02
Nitrate (mg/L)	0.58	<0.01
Phosphate (mg/L)	0.01	0.18
Total organic carbon (mg/L)	5.7	2
Aluminum (mg/L)	0.64	0.05

dark- to light-gray, fine-grained, micaceous, phosphatic and glauconitic sand and clay. Glauconite is abundant (up to 30 percent) in the sand layers. Phosphate in the form of shark teeth and large pebble-size grains is common. Lignite is present in trace amounts or in distinct laminations. Pyrite occurs, mostly as casts and cement, in amounts up to 2 percent. Shell fragments are present in trace amounts throughout the formation, with higher abundance in the shallower units. Thin layers of calcareous cemented sandstone are numerous, particularly in the upper third of the formation (Castro and others, 1995). The sediments were probably deposited in an estuarine or near-shore marine environment. Figure 2 illustrates the Black Creek sand zones, and the screen settings for the pumping well and an observation well. The quality of the native ground water is only marginal for drinking. In general, the water is of a sodium bicarbonate type, soft, alkaline, and low in iron. It commonly has objectionable concentrations of fluoride, sodium, and total dissolved solids. Moreover, the salinity of the water increases with depth. Table 1 compares the water quality of the Black Creek Formation and the treated water from the AICW.

Aquifers of the Black Creek Formation were, for nearly 100 years, the primary source of drinking water in Horry County, and thus were subjected to increasingly heavier pumping. Hence, rates of drawdown greater than 10 ft (feet) per year were common during the early 1980's (Pelletier, 1985). It is believed that, at the beginning of the 1900's, most of the wells in Horry County were free-flowing. By the 1980's, however, a large cone of depression

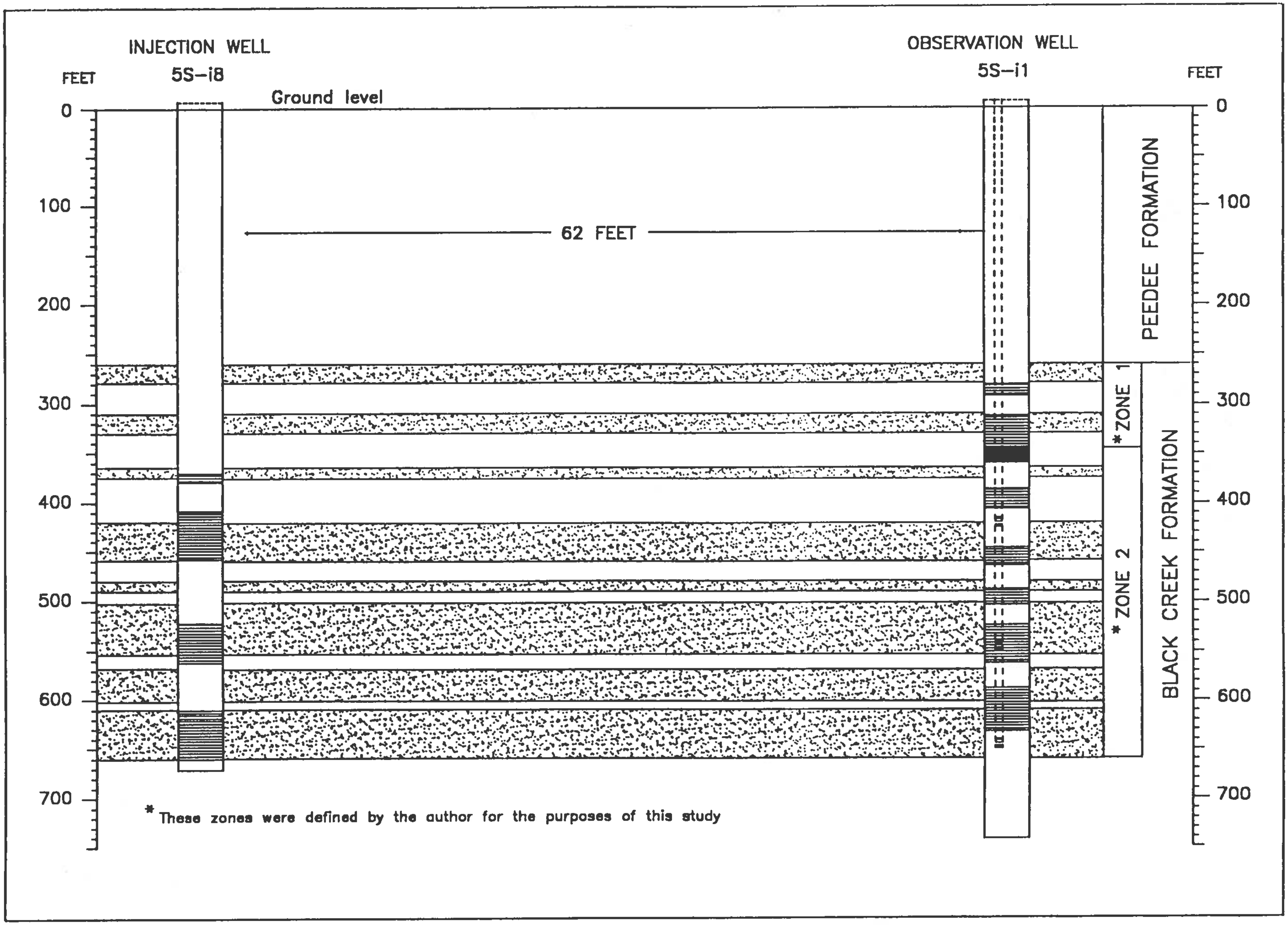


Figure 2. The Black Creek aquifers and screen settings at the ASR injection site. See Figure 5 for well construction details.

had developed that extended throughout the eastern section of Horry County. In 1988, the center of the cone had a potentiometric level of 180 ft bls (Hockensmith, 1990). Since the end of 1988, after the city of Myrtle Beach abandoned the public supply wells, the aquifer has been slowly recovering (Fig. 3) (Curley, 1988).

ASR PROTOTYPE WELL

An ASR site was set up in Myrtle Beach near the intersection of 38th Avenue North and Oak Street. Two Black Creek wells and one water table well are present at the site (Fig. 4). One of the Black Creek wells (5S-i8), a former public supply well, was used as the production well; the other, 5S-i1, is a multilevel observation well used to monitor two zones of the Black Creek Formation. For the purpose of this study, the section of the well that monitors the upper aquifers of the Black Creek was named observation zone 1; the lower section of the well was called observation zone 2. Most of the screens in the production well were located in the depth range of those of observation-well zone 2. The water table well, 5S-i6, initially monitored the phreatic zone; however, it was later discontinued because no changes in water levels were observed during the injection period. Construction details of the injection and observation wells are shown in Figure 5.

During the field tests, well 5S-i8 was used for both injection and recovery (dual-purpose well). Only minor wellhead modifications were necessary for retrofitting the former public supply well for injection. This helps illustrate the point that most water-production wells can be inexpensively upgraded to dual-purpose ASR wells.

OBJECTIVES AND DESIGN CONSIDERATIONS

The number and type of ASR applications have significantly increased as more systems have become operational. Pyne, 1995, listed 22 ASR applications:

- Seasonal storage
- Long-term storage
- Emergency storage
- Disinfection byproducts reduction
- Restoration of ground water levels
- Reduction of subsidence
- Maintenance of distribution system pressure
- Maintenance of distribution system flow
- Improvement of water quality
- Prevention of salt water intrusion
- Reduction of environmental effects of streamflow diversions
- Agricultural water supply
- Nutrient reduction in agricultural runoff
- Enhanced wellfield production
- Deferred expansion of water facilities
- Compensation of surface salinity barrier leakage losses
- Reclaimed water storage for re-use
- Soil aquifer treatment
- Stabilization of aggressive water
- Hydraulic control of contaminant plumes
- Diurnal storage
- Fish hatchery temperature control

Four additional applications can be added to the list:

- Stabilization of treatment plant production
- Optimization of treatment plant production
- Energy recovery during injection
- Provision of effective means of conjunctive ground water/surface water use

ASR facilities are normally designed so that drinking water can be stored in an aquifer during periods of low water demand, such as fall and winter, and recovered during high water demand periods, such as spring and summer. ASR systems could become an essential alternative in the management of water supply systems. In public supply systems driven by seasonal water demands, ASR facilities not only provide additional flow but also extend the service life of treatment facilities and their transmission lines. The city of Myrtle Beach, for example, has a well-defined seasonal demand (Fig. 6). The average demand in 1992 was 11.5 mgd (million gallons per day).

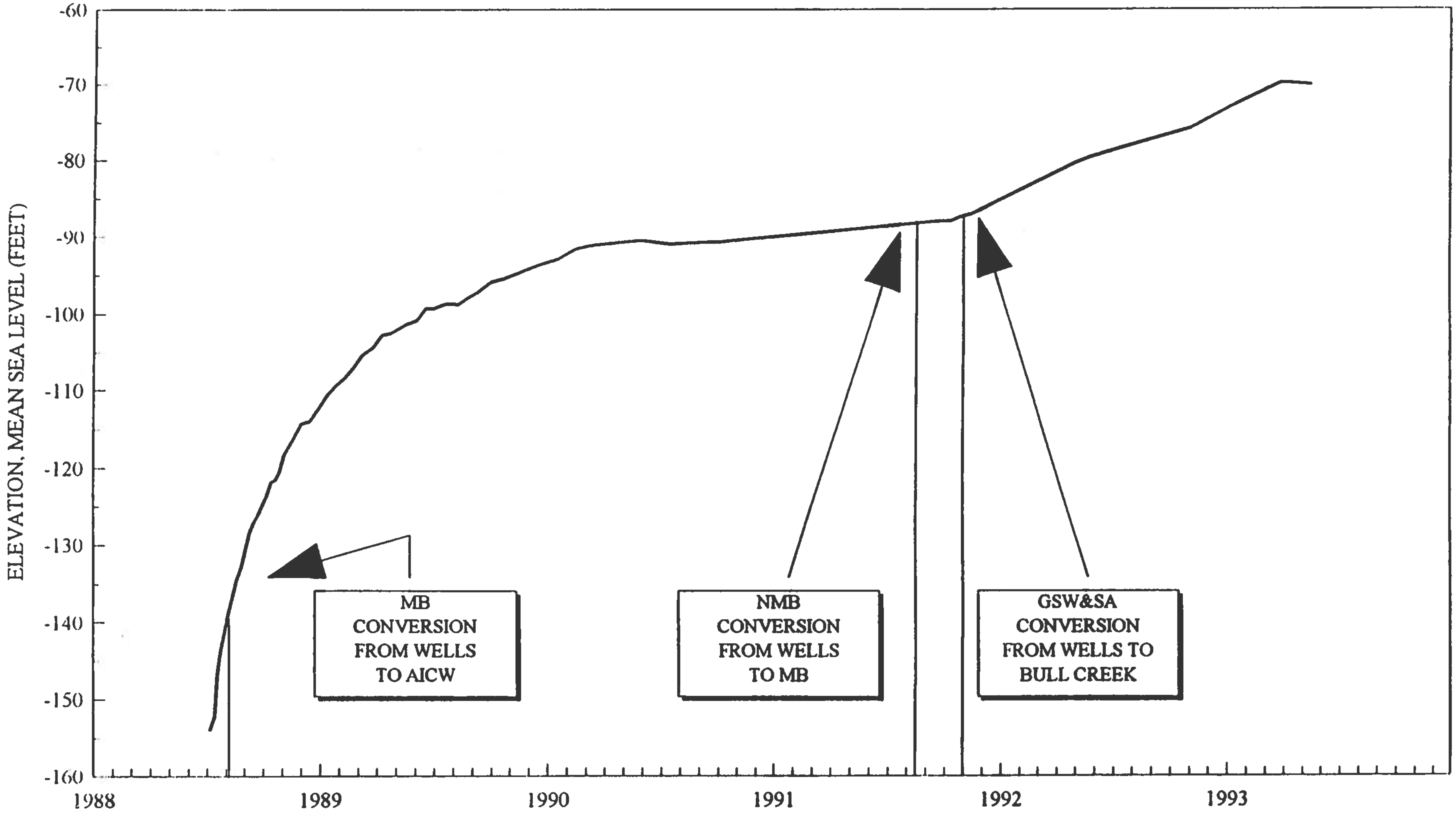


Figure 3. Recovery of the potentiometric level of the Black Creek aquifers near downtown Myrtle Beach, S.C. MB is Myrtle Beach; NMB is North Myrtle Beach; GSWSA is Grand Strand Water and Sewer Authority.

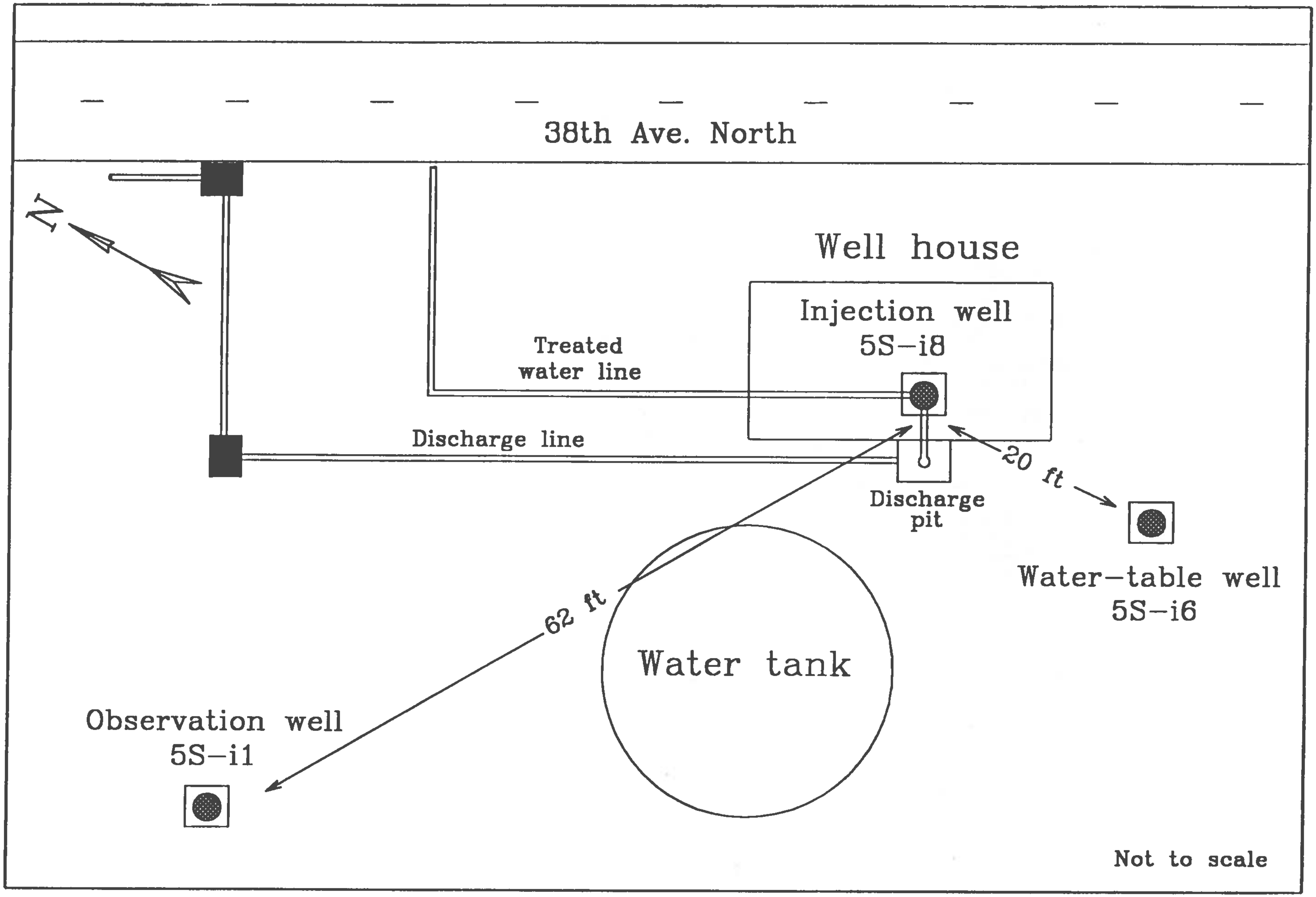


Figure 4. Plan view of the ASR injection site at Myrtle Beach, S.C.

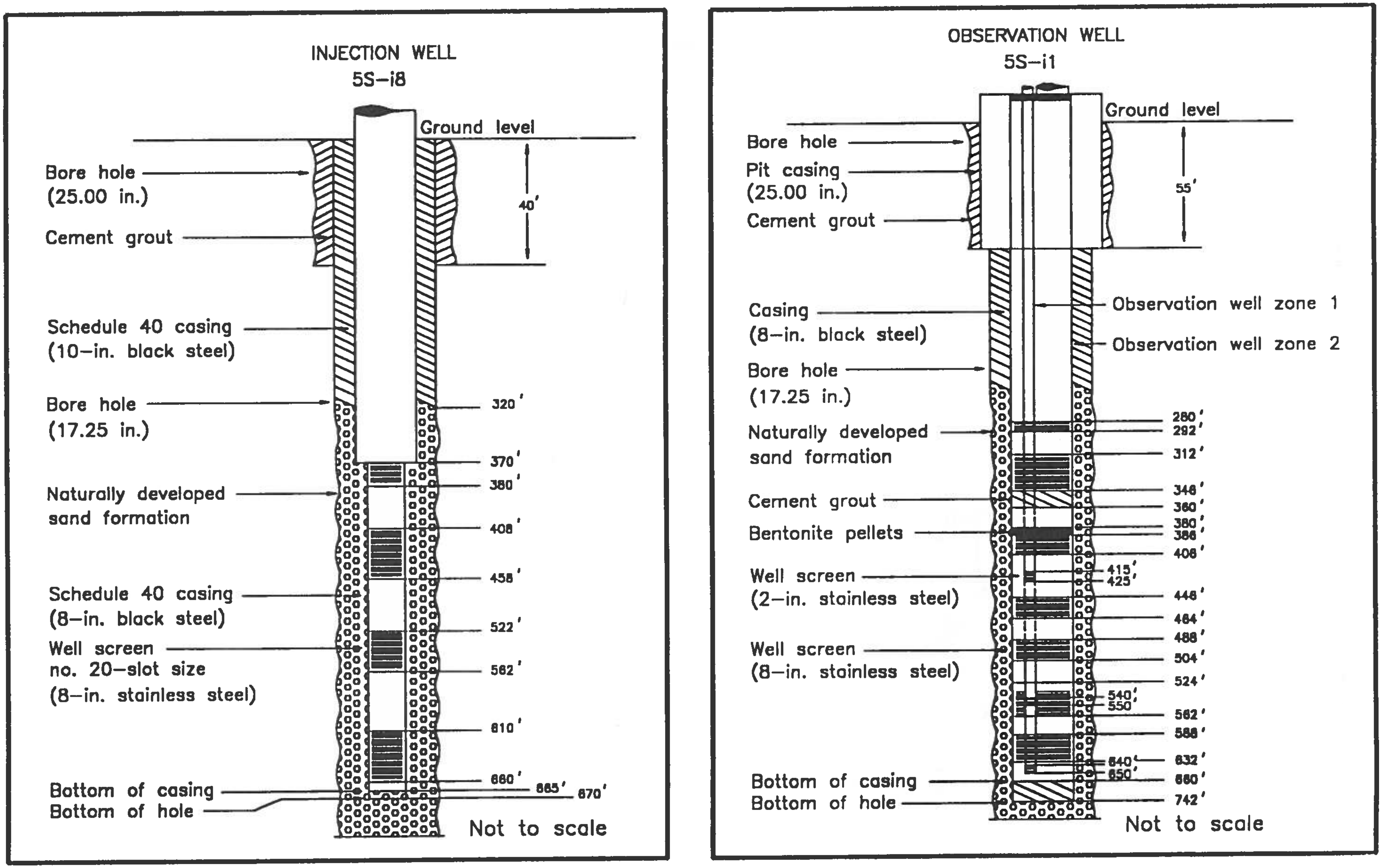


Figure 5. Construction details of the injection well (5S-i8) and observation well (5S-il).

The average demand in July was 19.2 mgd; but in December it was only 6.6 mgd. The July demand was almost 3 times the December demand; this difference becomes more critical when peak demands, rather than average daily demands, are considered. The July peak daily demand was 8 percent larger than the daily average for the month and 81 percent larger than the daily average for the year. Therefore, future peak daily demands and average summer month demands are expected to exceed the plant production capacity (Fig. 7), although the number of such days would be few. Drinking water regulations require that the capacity of the system be considered for expansion when demand exceeds 80 percent of plant capacity. Plant operations after expansion could be difficult and expensive, especially during the first half-life of the plant. Operations would be difficult because the plant would operate at flow rates below the optimum capacity, and treatment processes might not be as efficient or effective. Operations would be more expensive because plant production would be based on a varying demand and not an optimum flow.

If the city of Myrtle Beach decides to implement an ASR well system to avoid an expansion of their present system and to stabilize production of the water treatment plant, the design considerations that must be determined are:

- volume of injection
- number of ASR wells
- schedules of injection and recovery

The city of Myrtle Beach used over 4 billion gallons of water in 1992. Figure 6 shows that the tourist season, months characterized by above-average demand, extends from April to September. An ASR system designed to stabilize the operation of a treatment plant would, consequently, inject water from October through March and recover water from April through September.

The volume available for injection is estimated by computing the water produced by the treatment plant, when operating at the yearly average of 11.5 mgd, minus the volume required to meet the monthly demands. The monthly demands are computed by multiplying the month's average daily demand by the number of days in the month. Table 2 shows that the injection volume would be near 570 mgy (million gallons per year).

The number of wells required for the ASR system was estimated by comparing the total injection volume to an average injection capacity per well. If the average production (pumping) capacity of a Black Creek well was estimated to be 0.69 mgd, then it could be assumed that the corresponding injection capacity is 0.23 mgd, one-third of the production capacity. To stabilize the production of the treatment plant in 1992, 13 wells injecting at a rate of 0.23 mgd for 182 days would have been required.

Other economic considerations must be examined in determining the desired production level of the plant and the corresponding number of ASR wells; they will be explained later.

Table 2. Monthly schedule for an ASR well

MONTH	AVERAGE USAGE (MGD)	INJECTED	RECOVERED
		(MILLIONS OF GALLONS)	
January	7.69	117.80	
February	8.57	84.68	
March	9.95	47.74	
April	11.76		8.10
May	12.05		17.36
June	14.32		84.90
July	19.20		231.30
August	17.48		185.70
September	12.76		38.10
October	9.46	62.93	
November	7.96	105.90	
December	6.62	150.90	
TOTAL		569.95	565.46

FIELD OPERATIONAL PROCEDURES

An ASR yearly cycle is made up of injection and recovery periods. Both periods can be long, short, or a combination of long injections with short recoveries. Each injection period consists of a series of continuous-injection days interrupted by short backflushing periods.

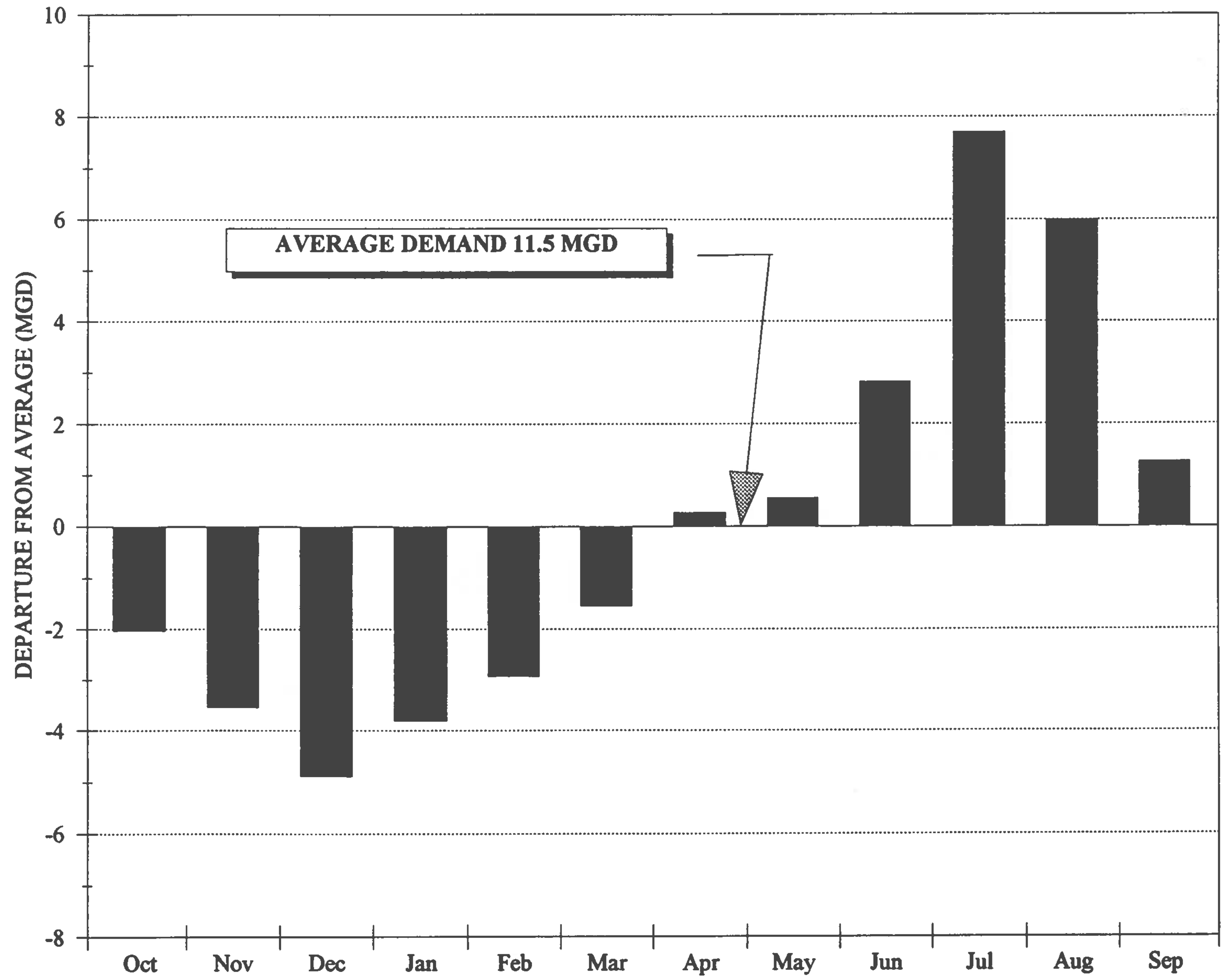


Figure 6. Seasonal water-use pattern at Myrtle Beach in 1992. Estimated yearly average demand was 11.5 million gallons per day.

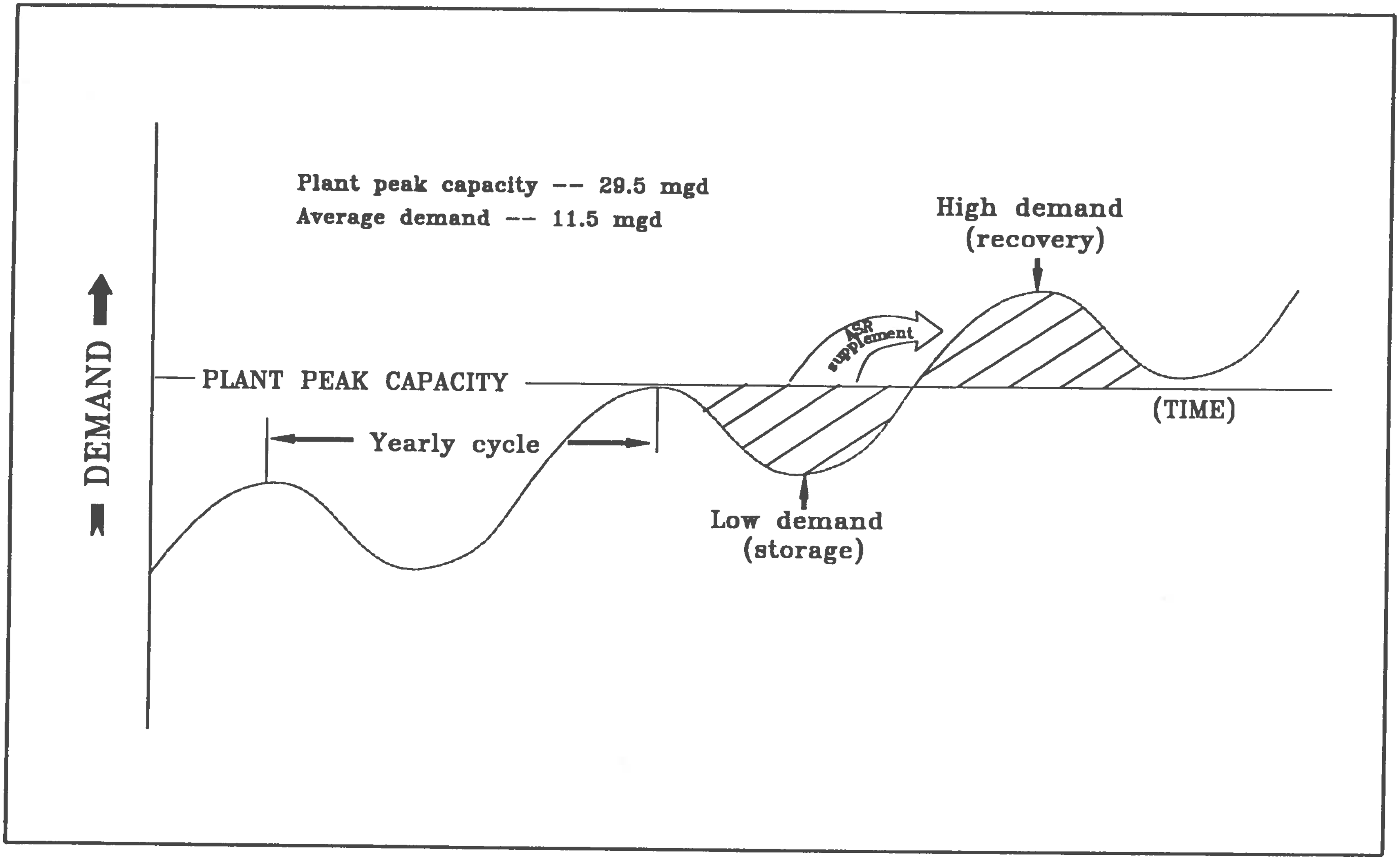


Figure 7. Application of ASR in a public-supply system characterized by a seasonal demand. High-demand months are supplemented by water stored during low-demand months.

INJECTION RATE

Injection rates of ASR wells are best determined in the field. The objective in selecting an appropriate injection rate is to attain a near-constant head buildup in the injection well, with a rate of increase small enough to allow for the injection of a specified volume of water for a given time period. Injection rates may vary, depending on the conditions of the system. For example, if colder, treated water becomes available for injection, the injection rate would have to be decreased accordingly in order to avoid an excessive buildup of pressure or even an overflow of the well. A good starting injection rate is one-third of the production capacity of the well. This rate can be modified, depending on the performance of the well and aquifer. During the long-term test in 1992, the injection rate was 100 gpm (gallons per minute); in subsequent tests it was greater. The maximum injection rate tested was 200 gpm. The best performance of the well and aquifer was obtained when the injection rate was 130 gpm, 29 percent of the production capacity of the well.

BACKFLUSHING PROCEDURES

Injection was halted periodically in order to backflush the well. A backflushing procedure normally consisted of three short intervals. The first interval was a rest period (10-minutes long) that allowed the shaft of the pump to stop spinning after injection was stopped—the pump was turned off during injection. Injected water, while moving down the pump column (riser), spun the shaft of the pump motor in a direction opposite to the rotation during pump operation. If the shaft were not allowed to stop, it could have been broken when the pump was turned on. The second interval was a 20-minute pumping period. The well was pumped long enough to remove water stored in the well casing, gravel pack, and immediate vicinity of the well bore. This was done to clean any suspended particles mobilized during injection and trapped in the screens and gravel pack. Finally, the third interval was another 10-minute rest period. During this period, the shaft was allowed to stop spinning, and air that was trapped in the pump column after the pump was turned off vented out. During the test in 1992, the well was backflushed every day; however, in subsequent tests, it was backflushed only twice a week.

During backflushing, the recovered water was observed to have a reddish or, sometimes, greenish tint. This discoloration of the water, along with a slightly musty odor, normally dissipated within 5 minutes. These changes in quality were attributed to suspended solids. The removal of these solids with each backflushing procedure improved the quality of the water and the performance of the well. Figure 8 shows two events during the long-term test in 1992. The first half of the graph shows four consecutive injection days with daily backflushes. The second half shows three consecutive injection days without backflushes. The large and rapid rise of water levels (buildup) in the second half were attributed solely to the absence of a backflush period. In this particular case, the water level came within 10 ft of land surface. The situation was immediately corrected when a backflush procedure was reinstated.

Throughout the tests, backflushing was always considered an essential element of the field procedures. Although backflushing efficiency was never quantified, available quantitative information supported the importance of a well-balanced backflushing procedure. Nevertheless, the benefits achieved by backflushing the well—increase in injection storage— have to be weighed against the loss of stored water during the backflush.

SITE-MONITORING REQUIREMENTS AND EQUIPMENT

The ASR site was set up to monitor a variety of factors. The most important were:

- injection and discharge rates
- volume injected/recovered
- head (water levels)
- water temperature
- specific conductance
- water quality

A flow-through cell was installed to monitor dissolved oxygen, specific conductance, redox potential, pH, and temperature. The cell was a transparent plexiglass

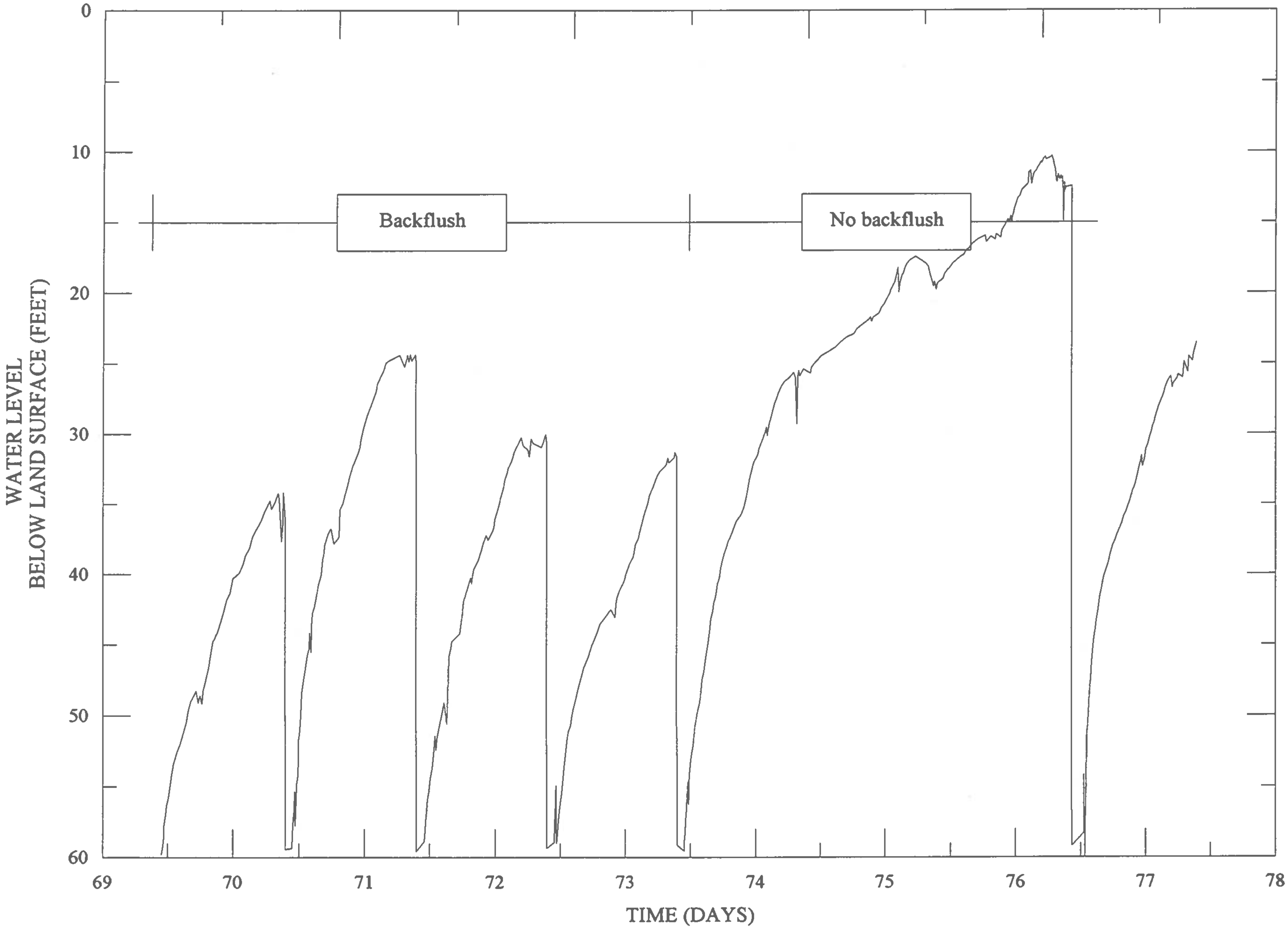


Figure 8. Injection-well water-level response to backflush procedures during the 1992 injection test.

pipe 5 ft long and 4 inches in diameter. Along its length were five orifices that held the monitoring probes. One end of the cell was connected to the well, allowing it to divert either the injected or discharged water, depending on the mode of the system. The other end was connected to a discharge pit. The cell worked under pressure and was regulated to a flow of 10 gpm. The purpose of the cell was to obtain readings, particularly of dissolved oxygen, before the water came in contact with the atmosphere. This also eliminated carbon dioxide degassing, which could have altered pH. Water samples were periodically collected to monitor chemical composition and to determine bacterial abundances.

Rates of injection and recovery were important because they provide an indication of the performance of both the well and aquifer. The rates were used in estimating water-level buildup, specific capacity, and in accurately computing volumes and water budgets. In the field, injection and recovery rates were adjusted by pressure valves. Gate valves, commonly found in wellhead installations, were adequate for ASR sites; however, butterfly valves were better suited because they were more easily operated.

Water volumes, estimated from injection/recovery rates and time or directly measured by flow meters, were necessary in the preparation of water budgets and in management of the injected plume. It was always important to know the total volume injected or recovered, as well as the volume left in storage. Digital flow meters were useful because they provided instantaneous flow rates and cumulative volumes.

Head, or water level, was measured continuously at both wells. It was useful in estimating specific drawdowns or specific capacities. Pressure transducers were installed on the wells to measure water level. Readings were taken regularly at various sampling intervals, depending on the test. For the pressure transducers to be effective, special care must be exercised in selecting operational ranges and accuracy of sensors. The sensor range must accommodate low pressures during pumping and high pressures during injection. For this study, the sensor had a 100 PSI (pounds per square inch) range (231 ft) and an accuracy of 0.1 percent of the range (2.8 inches).

The temperature of the injected water was closely monitored because the hydraulic conductivity and the performance of the well varies with temperature. During winter nights, the water level in the pumping well reached unusually high levels. On more than one occasion, the water level was less than 5 ft below land surface. If the temperature of the injected water dropped from 15°C to 10°C, the aquifer hydraulic conductivity, and therefore the performance of the aquifer, decreased by more than 30 percent.

The specific conductance, which can be easily and inexpensively monitored on site, was measured regularly in the pumping well. This property provides an indication of water quality and can be used instead of more expensive laboratory analyses.

FIELD TESTS

Between February 1991 and October 1992, 10 injection, storage, and recovery tests were completed. The first nine injection tests were a series of short-term tests designed to investigate the hydraulic changes of the aquifer properties resulting from the injection of treated surface water. The last injection test, in contrast, was a long-term test designed to study geochemical processes affecting the quality of the recovered water. Table 3 provides dates, volumes, and rates for all tests.

SHORT-TERM TESTS

Each short-term test, as shown in Table 3, consisted of one or more days of injection, a day of resting (storage), and a day or less of recovery. After the 1991 tests, two aquifer tests were made to assess changes in the hydraulic properties of the aquifer caused by the injection cycles. Current information suggests that no significant change in the transmissivity of the aquifer has occurred. Two pumping tests, made in 1982 and 1990 prior to any injection tests, gave an average transmissivity of 12,000 gpd/ft (gallons per day per foot). The tests made in 1991, after the short-term injection tests, gave the same average transmissivity.

Table 3. Field-test summary

TEST	DATE	INJECTION VOLUME (gallons)	INJECTION RATE (gpm)	RECOVERY VOLUME (gallons)	RECOVERY RATE (gpm)
1	FEB 6-7,1991	74,000	100	183,800	385
2	MAR 20-21,91	35,600	100	116,200	385
3	APR 11-15,91	150,300	100	174,300	385
4	MAY 15-20,91	148,200	103	169,500	374
5	JUN 5-10,1991	294,800	102	363,000	393
6	DEC 3,1991	36,100	100	65,300	430
7	DEC 4,1991	34,400	96	66,900	434
8	DEC 5,1991	38,600	103	67,000	427
9	JAN 5-6, FEB 3-13,1992	397,100	102	552,600	360
10	FEB 24- OCT 7,1992	7,703,800	100	11,711,500	313

LONG-TERM TEST

During the long-term test (February 24 to October 7, 1992), treated surface water was injected for 23 hours a day throughout February, March, April, and May. The injection rate was 100 gpm, and the distribution system's pressure (50 psi) was used to force the water into the aquifer. Injection was suspended for 30 minutes every weekday, in order to backflush the well ,and often during entire weekends and holidays. Injection rates, moreover, were continuously adjusted to compensate for pressure changes in the distribution system. After 71 days, injection was stopped. Throughout June, July, and August, 7.7 million gallons of treated surface water remained stored in the Black Creek aquifers. Beginning on September 8 and continuing for the next 21 days, 11.7 million gallons were recovered and pumped to waste.

Water levels. At the injection site, water levels for the Black Creek aquifers were measured by recording pressure in wells. Pressure transducers were installed in the pumping and observation wells. The accuracy of these readings was ± 0.25 ft; but water levels have been reported to only the nearest half foot. All readings were reported as depth to water in feet below land surface (ft bls).

Prior to January 1991 (before the injection tests), the potentiometric surface of the Black Creek was 110.5 ft bls at the test site. This depth had been steadily decreasing after the public supply wells were shut down following the Myrtle Beach conversion of its drinking water supply from ground water to surface water. The sudden end to the pumping brought about a rapid and extensive recovery of the aquifers. This increase is readily observed in Figure 3, which shows water levels for well 5S-n1, located in downtown Myrtle Beach.

Just before the long-term test, in February of 1992, the water level at the ASR site was 109.5 ft bls. At the end of the storage period it was 103.5 ft, and by the end of the recovery period it was 105 ft. In January 1994, the water level was 94 ft. The aquifer appeared to have recovered 16.5 ft between January 1991 and January 1994, most of which, surprisingly, occurred during 1993.

Water levels for the pumping well and observation-well zone 2 are given in Figures 9 and 10, respectively. Each graph shows three distinctive zones corresponding to the injection, storage, and recovery periods. Blank spaces between data points are unrecorded periods caused either by recorder malfunction or because data were not collected. The injection period, between 0 and 93 days, can be recognized by successive peaks and troughs. The peaks, which reached levels of less than 50 ft bls, were periods of active injection and normally lasted for 23 hours per day. The companion deep troughs, which reached levels greater than 250 ft bls, were backflush intervals that lasted about 30 minutes and immediately preceded injection cycles. The storage period, between 93 and 195 days, was marked by weekly, 2-hour-long pumping cycles, which are not shown in the graph. These pumping intervals were necessary to collect water samples. The recovery period, between 195 and 250 days, was characterized by a single, deep, and wide trough which represented pumping of the well for 20 consecutive days.

Injection period. The 7.7 million gallons of treated water injected in the aquifer raised the water level by an average of 55 ft in the pumping well and 10 ft in the observation-well zone 2. Water levels in the pumping well, in general, rose at unusual and irregular rates. Between late February and early April, the rate of change in the pumping well was positive (levels constantly rose); however, between late April and the end of May (end of injection) the rate of change was negative (levels progressively declined). In the observation-well zone 2, by contrast, the rate of change was always positive throughout the entire injection period. In both wells, unexpected high water levels were observed, higher than calculated. The behavior of these wells, although unexpected, was not unique, as similar trends have been observed at other injection sites (floor discussion at the International Symposium on Artificial Recharge of Ground Water, Orlando, Fla., July 17-22, 1994).

To properly study the hydraulic behavior of the well and aquifer during injection, two new factors were defined:

1. Buildup, expressed in feet, measures the gain in head due to injection. It is similar to drawdown during pumping. It gives an indication of the force required to drive the water into the aquifer.
2. Specific capacity, during injection, given in gallons per minute per foot, is defined as injection rate over buildup. It measures the performance of the well and aquifer during injection. It is similar to specific capacity during pumping.

Figure 11(a) and (b) shows the rate of buildup for the pumping well and observation-well zone 2. An arbitrary value N was added to the ordinates of each curve to plot them distinctively along the y-axis. The original slopes, however, were preserved (the slope of the curve is proportional to the transmissivity of the aquifer). In Figure 11(a) (pumping well), two slopes for each of the curves are apparent as marked by the solid lines. For April 7, the slopes correspond to transmissivity values of 2,000 gpd/ft for the first limb and 770 gpd/ft for the second limb. The transmissivities are smaller than the 12,000 gpd/ft calculated from pumping tests. Changes in slope between the first and second limb were attributed to increasing injection rates. Higher injection rates, in general, were observed during midnight hours. The higher rates were the result of daily fluctuations on water and pressure in the distribution system. Low water demands caused higher pressures and thereby higher injection rates. In the wells, higher injection rates induced accelerated buildup rates and higher water levels. The transmissivity values deduced from these curves are low and are difficult to explain, even after corrections for temperature are considered. The changes in transmissivity, moreover, cannot be attributed to plugging of the aquifer, as the specific capacity did not decline over time, and it even regained its pre-injection value after the injected water was recovered.

In observation-well zone 2 (Fig. 11(b)) the transmissivity was indicated to be 8,300 gpd/ft. A closer inspection of the graph shows that on April 7 there was a slightly larger transmissivity than on March 9 and May 4; although the values were similar, they were smaller than the 12,000 gpd/ft obtained in the pumping test.

Preliminary analysis of the specific-capacity data showed that (1) faster than anticipated rates of buildup were frequent, (2) reduction of transmissivity was more noticeable at the injection well than at the observation

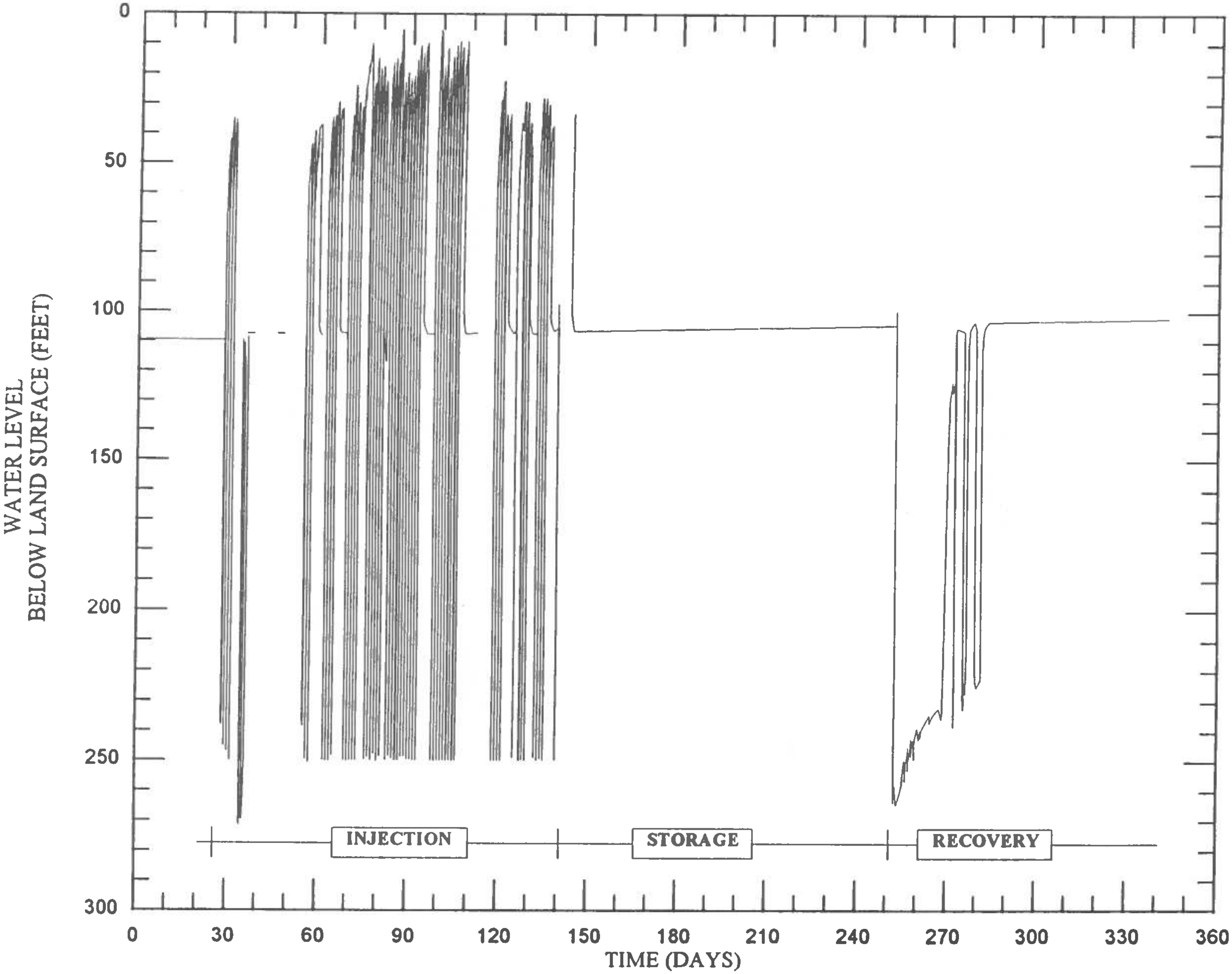


Figure 9. Pumping-well water levels during the 1992 injection test, which began February 24, 1992.

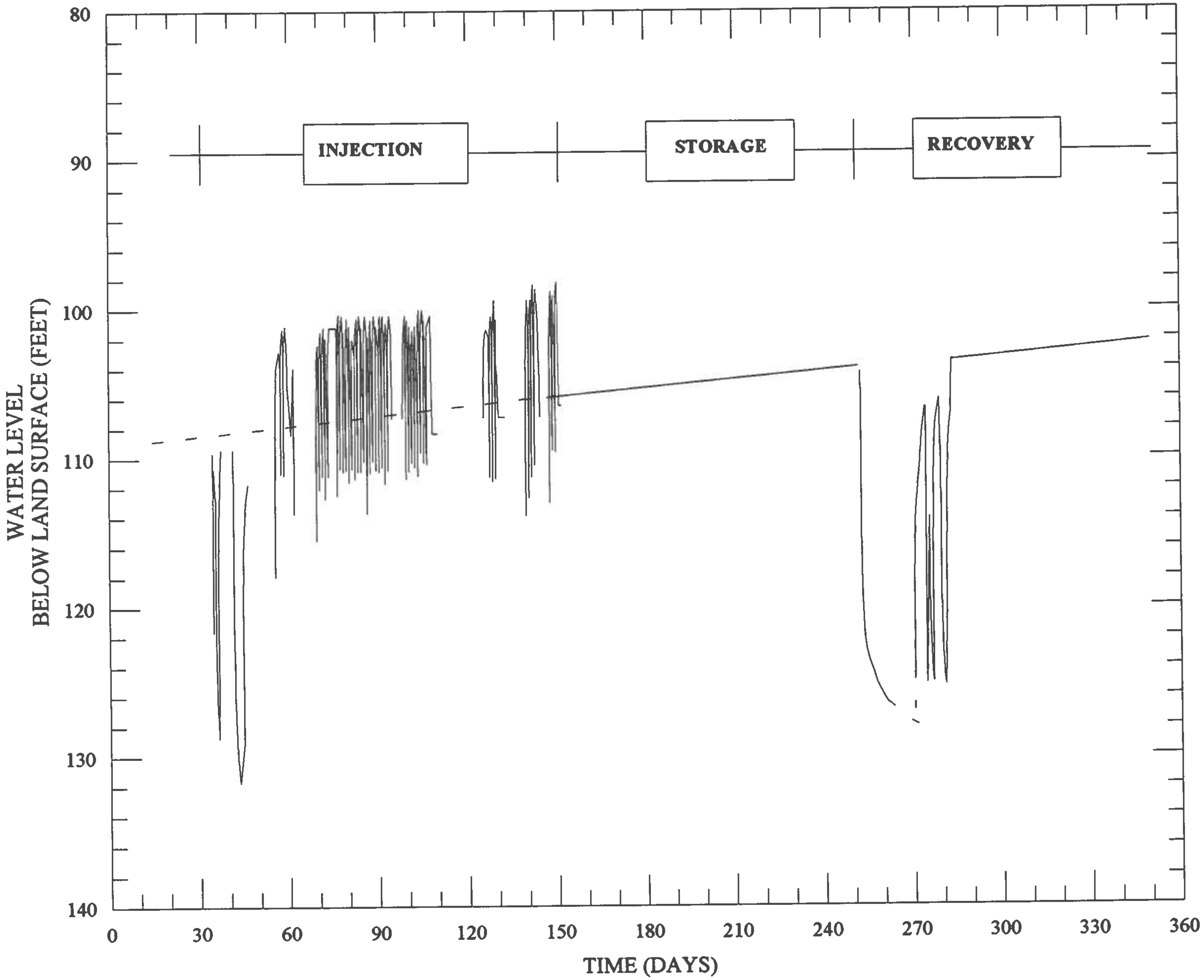


Figure 10. Observation-well zone 2 water levels during the 1992 injection test, which began February 24, 1992.

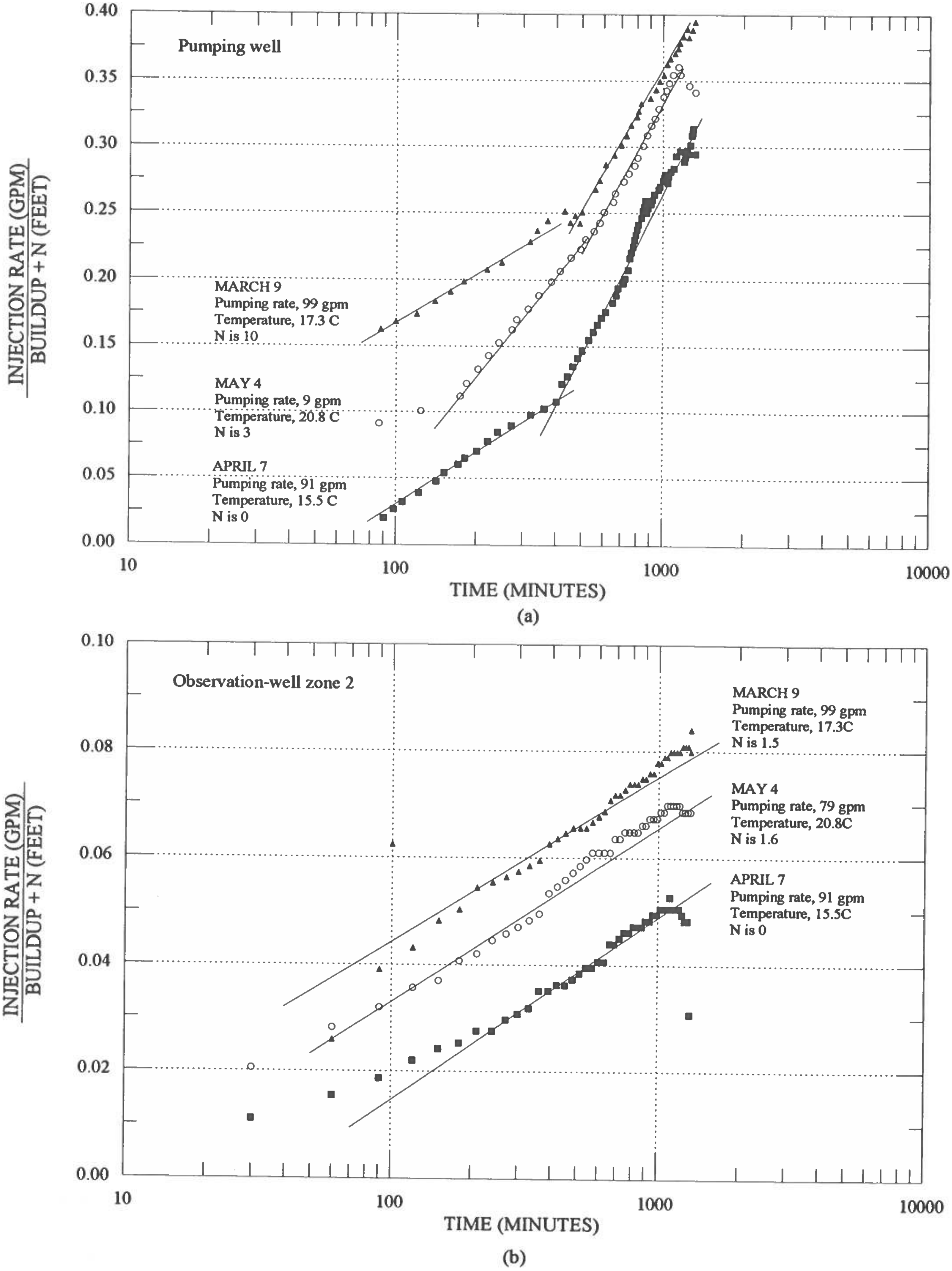


Figure 11. Comparison of buildup rates for the pumping well (a) and observation-well zone 2 (b) during the injection test in 1992.

well, (3) chemical and mechanical plugging (air entrainment) may be a problem, (4) loss of transmissivity was only temporary, and (5) reduction in water temperature decreased transmissivity.

Traditionally, injection has been treated as the reverse of pumping (Freeze and Cherry, 1979). Under this assumption, water levels (buildup) should be predicted by using the Theis equation or the Jacob-Cooper approximation if the aquifer properties and injection rates are known. Analysis of the ASR data, however, suggested otherwise.

During injection, specific capacity did not behave as predicted by theoretical equations. Specific-capacity values, in general, were smaller than corresponding specific-capacity values during pumping.

The following discussion briefly reviews some possible explanations for this anomalous behavior. Future work will address these issues in greater detail:

1. Increased energy losses due to turbulent flow in and around screens and filter pack. Screens of production wells are often designed to be nonclogging by using a V-shaped slot. During injection, exit velocities increase as water flows through the V-shaped slot, developing turbulent conditions that increase energy losses.

2. Plugging of the aquifer by chemical precipitation. This explanation has been, tentatively discarded because the specific capacity increased (improved) after the injected water was removed. Therefore, it can be assumed that the aquifer was not permanently affected by injection and, more importantly, that the changes were reversible.

3. Plugging of the aquifer by air entrainment. Most wellhead fittings are normally watertight but are seldom airtight. If, during injection, negative pressures (suction) develop inside the pump column , air could be forced into the aquifer. Part of the total mass of air might go into solution, but the residual fraction would lodge in the pore spaces, thus reducing the flow of water (transmissivity of the aquifer). At the site, many precautions were taken to minimize this problem; for example, injected water was delivered through the pump column at a depth of 200 ft below the standing water in the well. Moreover, the wellhead was always pressurized before injection in order to reduce the potential for air entrainment.

4. Reduction of hydraulic conductivity. Hydraulic conductivity is a function of the properties of the water (density, viscosity) and of the porous medium (porosity). A 20-degree decline in temperature from 24°C to 4°C would increase the density and consequently the transmissivity, by a factor of 0.27 percent. The same 20-degree change would increase the dynamic viscosity by 179 percent and decrease the transmissivity by 58 percent (Fig. 12). Hence, temperature changes greatly affect the transmissivity of the aquifer. Nevertheless, temperature corrections that were introduced in the predicted specific capacity were not sufficient to explain the observed changes. Temperature effects on the viscosity, therefore, could not be the sole cause of the transmissivity decline.

5. Another possible explanation for high buildup and lower specific capacity was found in the hysteretic behavior of the module of compressibility. The module is a ratio between the volumetric strain and stress; this "strain-stress" relation is neither linear nor elastic. For repeated loading and unloading processes, the aquifer may show some hysteretic changes in its compressibility, which, unlike the one for water, is not constant. Confining-bed compressibility module for expansion is only about one-tenth of that observed during compression (Freeze and Cherry, 1979). Jacob Bear (1972) pointed out that soil response to stress depends on the loading history, particularly in unconsolidated materials. In clay, which is highly plastic, the lag time for a response is significantly long, and therefore the assumption of immediate response of the aquifer to stress is incorrect. This means that a change in the stress of the system, as during injection, will not cause an immediate change in porosity, void ratio, or effective stress but, instead, would cause an increase in buildup rate. After the aquifer equilibrates to the new stress (stress increases and consolidates the porous medium), buildup rates decline.

Storage period. Between June and August, injected water remained stored in the aquifer, except during short periods of water sampling. Soon after injection stopped, water levels declined and reached a new static level. The

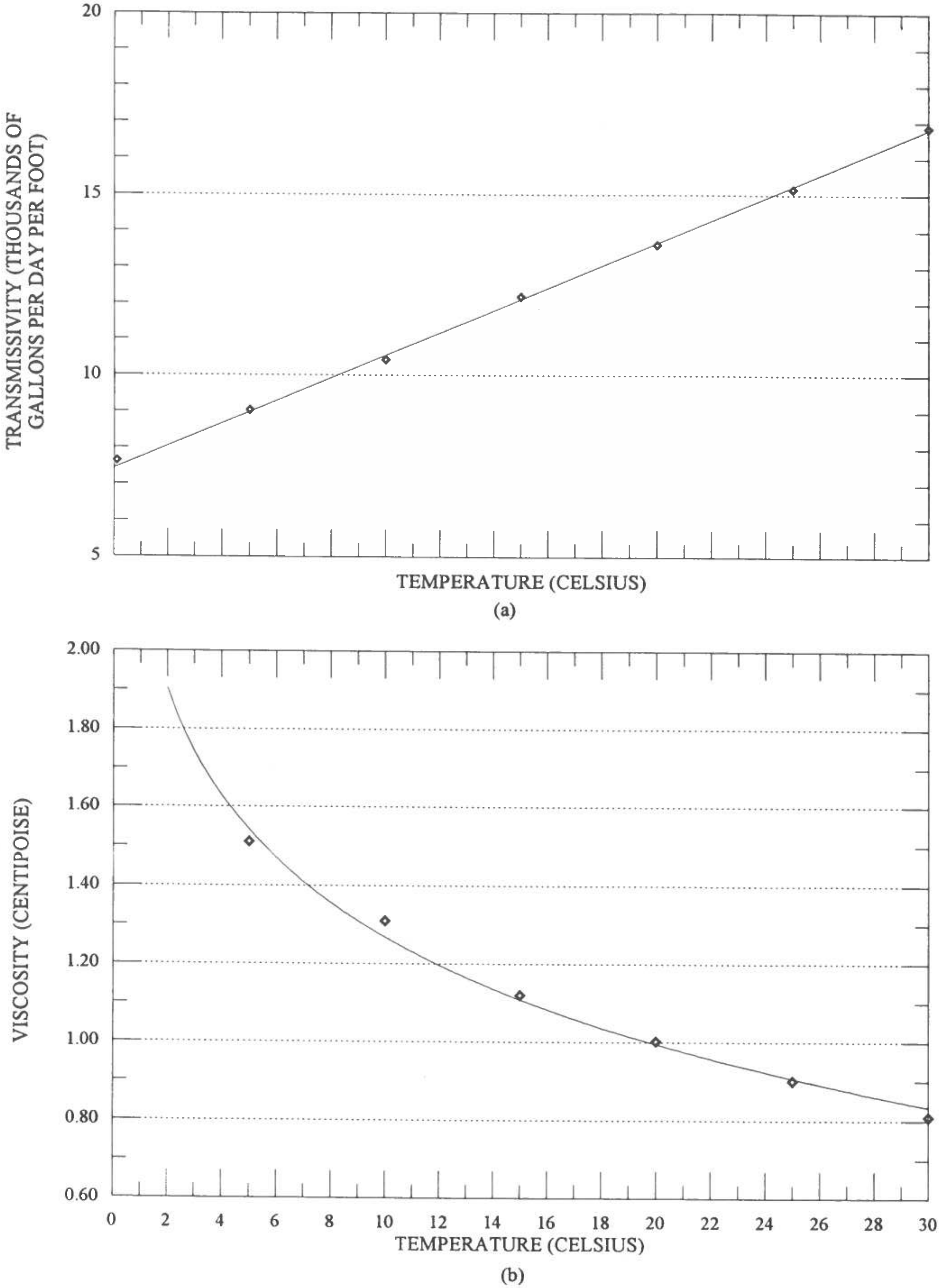


Figure 12. Variations in aquifer transmissivity (a) and water viscosity (b) as a function of temperature.

time required by the aquifer to stabilize was less than a day. After this initial period, the potentiometric surface of the aquifer progressively rose (Fig. 9), due to the regional recovery of the aquifer. Figure 10, which shows water levels in the observation-well zone 2, clearly identifies these gradual changes. The potentiometric head rose from 110.5 ft bls before the test to 104.5 ft after the test.

Recovery period. Beginning in September, the well was continuously pumped for more than 21 days at an average discharge of 310.6 gpm. Close inspection of Figures 9 and 10 show that the water-level rise while pumping was still taking place. This apparent inconsistency can be explained by considering the temperature changes of the recovered water. Lower water temperature induced higher water viscosity and, consequently, lower aquifer transmissivity. During the early part of pumping, most of the injected water that was removed had a lower temperature. The colder water induced a lower transmissivity, and therefore, larger drawdown. Later, a mixture of injected and native ground water was removed. The mixture, which was warmer because of the native ground water, induced a higher transmissivity in the aquifer. The drawdown consequently was reduced and the water level rose, although the well was still discharging at the same rate.

WATER QUALITY

Data collection at the ASR site was an intensive endeavor, where accuracy and consistency of measurements were of foremost importance. The objectives for this part of the study were (a) to obtain information on various processes affecting the injection test, and (b) to comply with monitoring requirements specified by the South Carolina Department of Health and Environmental Control through an Underground Injection Control permit. Cations were measured by inductively coupled plasma chromatography and anions by ion chromatography (Standard Methods, 1985) by XRAL Environmental (Ontario, Canada). Total organic carbon and trihalomethane (EPA method 510.1) were analyzed by General Engineering Laboratories (Charleston, S. C.). Low-level tritium analyses were made by the Environmental Isotopes Branch of the Alberta Environmental Centre (Alberta, Canada). Additionally, bacteriologic analyses were made by Environmental Systems Testing Services, Inc. (Conway, S. C.). Table 4 lists the water quality analyses made in 1992 during the long-term injection test.

The purpose of this section is to present and discuss the significance of the information. The scope of this discussion will be limited to the properties and constituents that are most sensitive to the quality from the drinking-water-standards perspective.

Chemical properties and constituents were plotted against time in order to discern variations that would define trends and anomalies in their behaviors.

From the analysis, it was apparent that chloride behaved conservatively; it did not react in the aquifer. Thus the chloride concentration in the recovered water was a function of the degree of mixing between the injected water and native ground water. This was confirmed by comparing tritium and chloride concentrations. Tritium, in environments not open to the atmosphere, is not significantly affected by reactions other than spontaneous radioactive decay (Freeze and Cherry, 1979); therefore, in ground water studies it is an excellent conservative tracer.

Figure 13(a) shows that tritium activity and chloride concentration are directly proportional to each other. This suggests that the chloride ion did not react with minerals from the aquifer and that chloride behaved conservatively in this system. Chloride, not tritium, was selected as a tracer for the present analysis, because more chloride analyses were available. The low-level tritium analysis was significantly more expensive than the chloride analysis.

Although the time plots provided a convenient way of inferring temporal and spatial variations, plots of ions against chloride clearly highlighted the relative effects of chemical reactions and mixing. In these graphs, concentrations plotting above a conservative mixing line (a line joining initial and final concentrations) indicate the net addition of an ion to the ground water by chemical reaction. Conversely, concentrations plotting below the mixing line represent net removal of an ion from solution by chemical reaction.

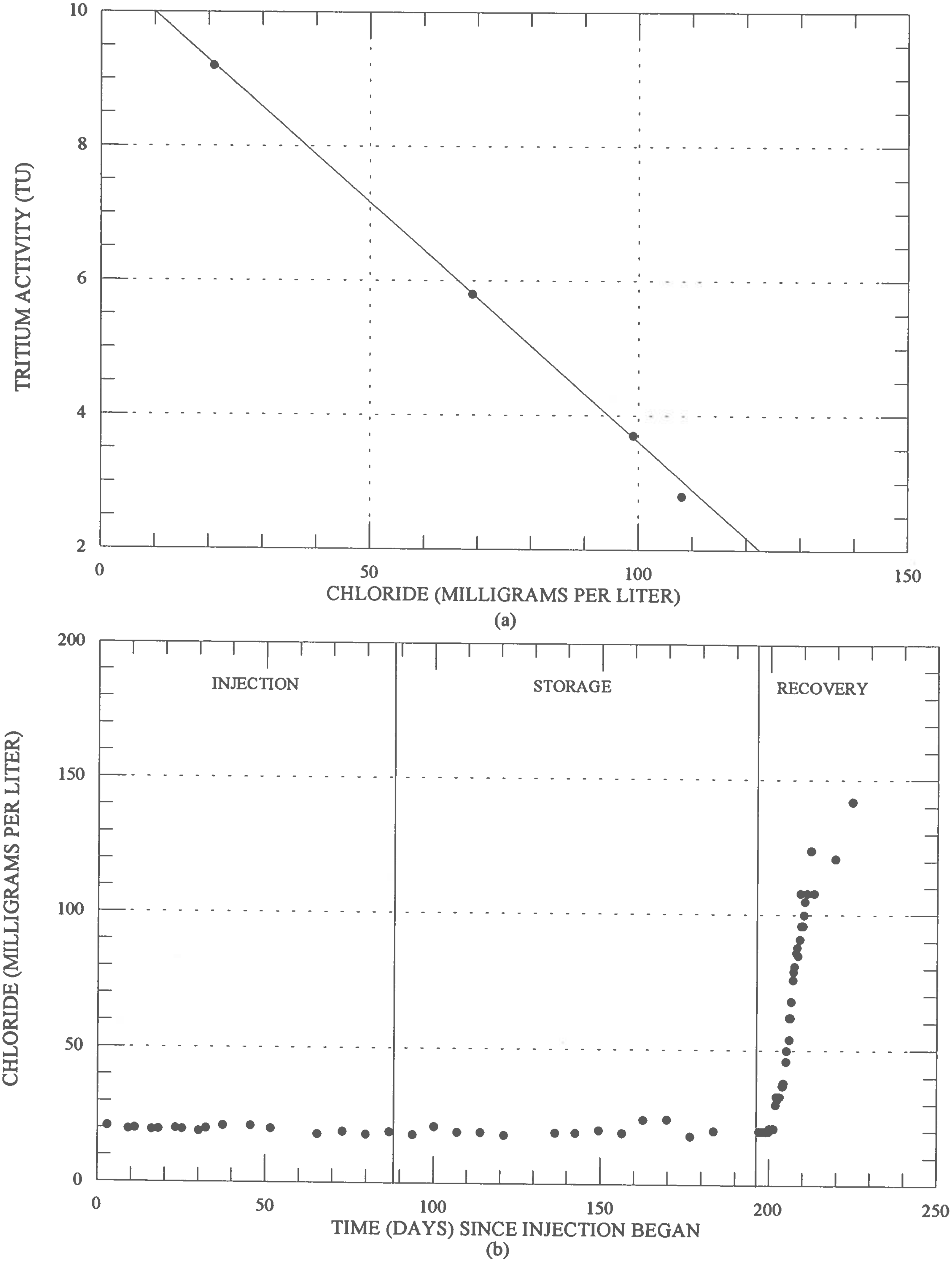


Figure 13. Comparison of chloride concentration with tritium activity (a) and with time (b).

Table 4. Water quality properties and constituents analyzed

Sample ID	Julian	Date	Time	Spec Cond. uS/cm	Temp. deg. C	Field Alk. mg/L	Field pH	Field D.O. mg/L	Lab Alk. mg/L	Lab pH	Aluminum mg/L	Boron mg/L	Calcium mg/L
17tl	**58.5**	**02/27/92**	**1100**	**219**	**15.0**	**34**	**7.3**	**5.90**	**23**	**7.3**	**0.165**	**0.06**	**10.0**
18tl	62.4	03/03/92	1000	202	16.3	22	7.2	5.84	15	7.3	0.572	0.05	15.6
19tl	64.4	03/05/92	0945	208	16.2	21	6.9	5.26	14	7.1	0.726	0.05	15.8
20tl	69.4	03/10/92	1015	204	18.1	21	7.0	5.23	14	7.1	0.316	0.03	14.3
21tl	71.5	03/12/92	1055	198	16.9	22	6.8	4.69	15	7.2	0.349	0.03	14.5
22tl	76.4	03/17/92	1040	208	15.2	19	7.1	5.34	13	7.1	0.501	0.02	14.9
23tl	78.4	03/19/92	1045	211	15.5	19	7.2	5.45	13	7.1	0.394	0.03	15.0
24tl	83.4	03/24/92	1020	221	15.3	27	7.3	7.57	18	7.5	0.380	0.03	15.0
25tl	85.6	03/26/92	1500	211	14.6	22	7.2	5.38	15	7.0	0.410	0.02	16.0
26tl	91.4	04/01/92	1020	211	16.1	25	7.3	6.37	17	7.9	0.430	0.02	16.0
27tl	**99.5**	**04/09/92**	**1230**	**211**	**16.9**	**24**	**7.3**	**7.25**	**16**	**7.5**	**0.390**	**0.02**	**16.0**
28tl	105.5	04/15/92	1100	223	18.4	28	7.4	6.94	19	7.1	0.350	0.02	14.0
29tl	119.5	04/29/92	1100	216	19.5	28	6.9	5.50	19	6.9	0.400	0.02	14.0
30tl	126.5	05/06/92	1255	214	20.9	30	7.3	6.31	20	7.1	0.340	0.02	15.0
31tl	133.5	05/13/92	1130	215	20.9	33	7.9	7.16	22	7.2	0.560	0.02	14.0
32tl	140.5	05/20/92	1135	223	23.0	37	7.9	6.63	25	7.3	0.720	0.02	14.0
33tl	147.5	05/27/92	1100	242	23.2	52	8.0	6.93	35	7.6	0.640	0.05	12.0
34wl	154.5	06/03/92	1230	313	23.5	89	7.1	0.08	60	7.2	0.070	0.07	23.0
35wl	161.5	06/10/92	1115	341	23.5	119	7	0.11	80	7.2	0.060	0.09	25.0
36wl	168.5	06/17/92	1210	320	23.4	105	8.5	0.08	71	7.1	0.060	0.08	26.0
37wl	**175.5**	**06/24/92**	**1110**	**336**	**23.2**	**120**	**6.8**	**0.11**	**81**	**7.2**	**0.050**	**0.08**	**28.0**
39wl	190.6	07/09/92	1500	338	22.8	120	7	0.09	81	7.3	0.050	0.09	29.0
40wl	196.5	07/15/92	1230	332	22.9	113	7	0.09	76	7.9	0.060	0.12	29.0
41wl	203.5	07/22/92	1200	329	22.4	110	7.1	0.14	74	7.4	0.050	0.11	29.0
42wl	210.5	07/29/92	1221	334	22.1	115	7.1	0.21	77	7.4	0.050	0.09	31.0
43wl	217.4	08/05/92	1034	338	22.0	114	7.2	0.20	76.8	7.3	0.050	0.08	29.5
44wl	224.5	08/12/92	1115	334	21.8	119	7.3	0.07	80.2	7.2	0.050	0.07	29.5
45wl	231.5	08/19/92	1225	340	21.6	124	7.3	0.08	84	7.5	0.050	0.08	28.4
46wl	238.5	08/26/92	1054	344	21.6	117	7.3	0.33	79	7.3	0.022	0.06	28.1
48wl	252.6	09/09/92	1418	350	20.7	211	7.7	0.01	141	7.5	0.053	0.14	23.3
48w2	**253.5**	**09/10/92**	**1138**	**364**	**20.5**	**177**	**7.9**	**0.01**	**120**	**7.9**	**0.138**	**0.16**	**18.6**
48w3	253.7	09/10/92	1538	371	20.5	193	8.2	0.01	127	7.9	0.157	0.17	18.1
48w4	254.4	09/11/92	1030	394	20.4	202	8.5	0.01	132	8.3	0.233	0.21	16.4
48w5	254.5	09/11/92	1230	399	20.5	210	8.8	0.01	142	8.3	0.236	0.20	16.3
48w6	254.6	09/11/92	1500	404	20.4	211	8.9	0.01	142	8.4	0.251	0.20	16.2
48w7	255.4	09/12/92	1015	435	20.4	242	8.8	0.01	167	8.3	0.275	0.25	15.3
48w8	255.5	09/12/92	1300	439	20.4	246	9.0	0.01	167	8.2	0.298	0.25	15.2
48w9	255.7	09/12/92	1600	443	20.4	254	9.1	0.01	172	8.2	0.270	0.25	15.3
48w10	256.4	09/13/92	1000	475	20.5	280	8.9	0.01	191	8.3	0.278	0.28	14.3
48w11	256.5	09/13/92	1300	480	20.5	280	9.1	0.01	186	8.2	0.303	0.29	14.3
48w12	**256.9**	**09/13/92**	**2215**	**497**	**20.6**	**288**	**9.1**	**0.01**	**191**	**8.4**	**0.289**	**0.32**	**14.3**
49wl	257.5	09/14/92	1115	521	20.7	291	8.9	0.01	196	8.7	0.283	0.36	13.8
49w2	257.7	09/14/92	1630	532	20.7	300	9.1	0.01	201	8.7	0.256	0.36	13.7
49w3	257.9	09/14/92	2200	543	21.8	312	9.1	0.01	210	8.7	0.261	0.40	13.5
49w4	258.4	09/15/92	0900	565	20.8	326	8.8	0.01	220	8.7	0.242	0.41	13.3
49w5	258.5	09/15/92	1200	573	20.9	330	8.9	0.01	221	8.7	0.239	0.43	13.3
49w6	258.7	09/15/92	1600	581	21.0	343	8.9	0.01	236	8.7	0.242	0.44	13.3
49w7	259.4	09/16/92	0945	625	21.1	410	8.9	0.01	274	8.8	0.221	0.51	12.7
49w8	259.5	09/16/92	1240	635	21.1	417	9.1	0.01	278	8.8	0.220	0.51	12.6
49w9	259.7	09/16/92	1630	645	21.2	425	9.1	0.01	283	8.8	0.194	0.55	12.6
49w10	**260.4**	**09/17/92**	**0940**	**701**	**21.4**	**453**	**8.8**	**0.01**	**301**	**8.7**	**0.181**	**0.63**	**12.5**
49w11	260.5	09/17/92	1230	710	21.4	458	8.9	0.01	308	8.7	0.190	0.64	12.5
49w12	260.7	09/17/92	1600	725	21.4	466	8.9	0.01	313	8.7	0.183	0.70	12.6
49w13	261.4	09/18/92	0930	746	21.5	416	8.9	0.01	278	8.7	0.162	0.70	12.0
49w14	261.4	09/18/92	0930	780	21.6	437	8.5	0.02	291	8.6	0.143	0.80	12.0
49w15	261.5	09/18/92	1300	789	21.8	441	9	0.01	296	8.6	0.172	0.80	11.9
49w16	261.7	09/18/92	1600	801	21.8	437	9.1	0.01	291	8.7	0.153	0.82	11.9
49w17	261.9	09/18/92	2200	814	21.9	459	9.2	0.01	300	8.7	0.161	0.87	11.8
49w18	262.4	09/19/92	1020	846	22.0	465	9.2	0.01	313	8.7	0.145	0.96	11.7
49w19	262.7	09/19/92	1600	859	22.0	468	9	0.01	313	8.7	0.136	1.00	11.7
49w20	**262.9**	**09/19/92**	**2200**	**875**	**22.1**	**475**	**9.1**	**0.01**	**322**	**8.8**	**0.106**	**1.04**	**11.7**
49w21	263.4	09/20/92	1015	901	22.2	486	9.1	0.01	327	8.7	0.135	1.11	11.4
49w22	263.7	09/20/92	1615	910	22.2	490	8.9	0.01	327	8.7	0.110	1.15	11.5
49w23	263.9	09/20/92	2200	925	22.3	501	8.9	0.01	332	8.8	0.109	1.20	11.3
50wl	264.4	09/21/92	1025	944	22.2	525	8.7	0.01	353	8.7	0.121	1.20	11.2
50w2	264.7	09/21/92	1600	955	22.2	531	8.9	0.05	357	8.7	0.118	1.23	11.1
50w3	264.9	09/21/92	2215	966	22.2	541	8.9	0.02	362	8.7	0.124	1.29	11.1
50w4	265.4	09/22/92	1000	981	22.3	548	8.8	0.01	366	8.7	0.103	1.36	10.8
50w5	265.7	09/22/92	1600	992	22.4	555	8.8	0.01	371	8.7	0.120	1.40	10.8
50w6	265.9	09/22/92	2200	1000	22.4	555	8.8	0.01	371	8.8	0.098	1.45	10.6
50w7	**266.6**	**09/23/92**	**1500**	**1016**	**22.5**	**565**	**8.9**	**0.01**	**380**	**8.9**	**0.098**	**1.53**	**10.4**
50w8	267.6	09/24/92	1500	1043	22.6	573	8.8	0.00	383	8.5	0.093	1.71	9.8
50w9	268.6	09/25/92	1510	1070	22.7	580	8.8	0.01	392	8.6	0.049	1.84	9.4
50w10	274.6	10/01/92	1440	1178	22.7	566	8.8	0.01	378	8.7	0.086	1.83	8.1
52wl	279.6	10/06/92	1500	1239	22.9	610	8.7	0.01	410	8.7	0.084	1.93	8.1

Spec Cond., specific conductance in microsiemens per centimeter
TOC, total organic carbon
THM, trihalomethane
THMP, trihalomethane potential
ALK, alkalinity as an equivalent concentration of calcium carbonate

for samples collected during the 1992 injection test

Chloride mg/L	Fluoride mg/L	Iron mg/L	Magnesium mg/L	Manganese mg/L	Nitrate mg/L	Phosphate mg/L	Potassium mg/L	Silica mg/L	Sodium mg/L	Sulfate mg/L	Sulfide mg/L	Tritium TU	TOC mg/L	THM ug/l	THMP ug/l
20.8	**1.0**	**0.029**	**1.30**	**0.010**	**0.23**	---	**2.21**	**4.7**	**22.6**	**36.9**	---	---	**7.0**	**17.9**	**955**
19.6	1.1	0.065	2.29	0.021	0.51	---	2.04	7.3	19.1	35.6	---	---	---	---	---
19.9	1.1	0.075	2.87	0.025	0.23	---	2.35	8.1	21.4	35.7	---	---	---	---	---
19.4	1.0	0.037	1.32	0.012	0.20	---	0.64	3.8	18.5	35.8	---	---	7.0	17.9	805
19.7	0.9	0.049	1.38	0.012	0.50	---	0.63	3.4	17.2	35.4	---	---	---	---	---
20.0	0.9	0.034	1.38	0.009	0.19	---	0.68	2.6	19.1	38.9	---	---	---	---	---
19.7	1.0	0.041	1.38	0.011	0.20	---	0.68	3.2	18.4	37.8	---	---	---	---	---
19.0	1.0	0.041	1.30	0.012	0.00	---	0.70	3.0	21.0	43.0	---	---	8.0	23.8	900
20.0	0.9	0.038	1.30	0.010	0.00	---	2.10	2.5	19.0	42.0	---	---	---	---	---
21.0	1.2	0.038	1.30	0.010	0.00	---	2.00	2.7	19.0	42.0	---	---	---	---	---
21.0	**1.1**	**0.047**	**1.30**	**0.010**	**0.70**	**0.010**	**2.00**	**2.8**	**19.0**	**41.0**	---	---	**6.0**	**20.0**	**955**
20.0	1.1	0.020	1.20	0.006	0.78	0.003	2.30	2.2	22.0	41.0	---	---	---	---	---
18.0	1.0	0.040	1.20	0.010	0.43	0.007	2.50	3.0	19.0	39.0	---	---	8.0	15.4	203
19.0	0.8	0.030	1.10	0.006	0.69	0.007	2.70	3.8	18.0	40.0	---	---	---	---	---
18.0	0.9	0.040	1.30	0.007	0.63	0.003	2.60	4.5	20.0	40.0	---	---	7.9	44.2	440
19.0	0.9	0.060	1.20	0.006	0.54	0.000	2.60	4.4	22.0	40.0	---	---	---	---	---
18.0	0.9	0.039	1.40	0.005	0.58	0.003	3.00	5.1	28.0	41.0	---	---	29.3	5.7	399
21.0	1.0	0.035	2.20	0.023	0.46	0.033	3.10	5.0	32.0	48.0	---	---	---	---	---
19.0	0.9	0.170	2.60	0.042	0.02	0.120	3.30	5.4	32.0	48.0	---	---	4.4	32.5	510
19.0	1.1	0.180	2.30	0.038	0.02	0.029	3.20	5.0	31.0	47.0	---	---	---	---	---
18.0	**1.1**	**0.439**	**2.50**	**0.041**	**0.02**	**0.033**	**3.30**	**5.6**	**30.0**	**47.0**	---	---	**5.3**	**7.9**	**450**
19.0	1.3	0.490	2.40	0.040	0.02	0.033	3.00	5.5	30.0	48.0	---	---	---	---	---
19.0	1.2	0.560	2.40	0.380	0.02	0.036	3.00	5.3	31.0	50.0	---	---	4.0	3.6	597
20.0	1.3	0.470	2.30	0.034	0.02	0.036	2.80	5.2	30.0	51.0	---	---	---	---	---
19.0	1.3	0.580	2.30	0.040	0.02	0.039	2.50	5.4	31.0	51.0	---	---	4.0	<2.00	583
24.0	1.2	0.560	2.30	0.038	0.01	0.117	2.58	5.7	33.3	46.0	---	---	---	---	---
24.1	1.5	0.530	2.33	0.034	0.01	0.114	2.60	5.5	36.3	48.4	---	---	---	---	---
18.0	1.2	0.410	2.21	0.029	0.68	0.046	2.50	5.4	37.7	50.0	---	---	---	---	---
20.0	1.6	0.470	2.17	0.032	0.01	0.055	2.60	5.5	39.1	49.0	<0.01	---	---	---	---
20.0	1.4	0.112	1.66	0.027	0.01	0.907	2.78	4.4	58.4	47.0	<0.01	---	---	---	---
20.0	**1.4**	**0.057**	**1.29**	**0.019**	**0.01**	**0.130**	**2.84**	**4.0**	**68.1**	**44.0**	**<0.01**	---	---	---	---
20.0	1.4	0.057	1.27	0.018	0.01	0.130	2.89	3.9	69.5	44.0	<0.01	---	---	---	---
20.0	1.5	0.048	1.10	0.014	0.01	0.189	2.82	3.9	78.4	46.0	<0.01	---	---	---	---
20.0	1.5	0.045	1.10	0.013	0.01	0.179	2.84	3.9	79.3	43.0	<0.01	---	---	---	---
20.0	1.5	0.046	1.08	0.024	0.01	0.140	2.79	4.0	80.6	43.0	<0.01	---	---	---	---
20.0	1.4	0.050	1.02	0.012	0.01	0.170	2.89	4.2	89.9	42.0	---	---	---	---	---
21.0	1.5	0.060	1.01	0.011	0.01	0.199	2.94	4.2	90.9	42.0	---	---	---	---	---
21.0	1.5	0.053	0.98	0.011	0.01	0.189	2.94	4.3	94.2	42.0	---	---	---	---	---
21.0	1.5	0.053	0.93	0.010	0.01	0.199	3.00	4.4	102.0	40.0	<0.01	---	---	---	---
21.0	1.7	0.060	0.94	0.009	0.01	0.199	3.00	4.4	103.0	41.0	<0.01	9.2	---	---	---
21.0	**1.7**	**0.052**	**0.93**	**0.009**	**0.01**	**0.199**	**3.13**	**4.6**	**110.0**	**40.0**	**<0.01**	---	---	---	---
30.0	1.7	0.045	0.94	0.009	0.01	0.254	3.31	4.7	116.0	43.0	---	---	5.7	<2.00	341
33.0	1.8	0.045	0.95	0.008	0.01	0.280	3.29	4.8	119.0	43.0	---	---	---	---	---
32.0	1.7	0.036	0.92	0.008	0.01	0.277	3.40	4.8	121.0	42.0	---	---	---	---	---
33.0	1.8	0.031	0.94	0.007	0.01	0.274	3.66	5.0	129.0	42.0	<0.01	---	---	---	---
33.0	1.8	0.033	0.91	0.006	0.01	0.264	3.72	5.0	131.0	43.0	<0.01	---	---	---	---
33.0	1.9	0.031	0.90	0.006	0.01	0.261	3.81	5.0	134.0	41.0	<0.01	---	---	---	---
37.0	2.3	0.026	0.92	0.007	0.01	0.179	4.17	5.3	146.0	46.0	---	---	---	---	---
37.0	2.2	0.016	0.95	0.006	0.01	0.179	4.29	5.3	147.0	39.0	---	---	---	---	---
38.0	2.4	0.023	0.96	0.006	0.01	0.160	4.35	5.4	151.0	39.0	---	---	---	---	---
46.0	**2.8**	**0.023**	**0.99**	**0.005**	**0.01**	**0.140**	**4.94**	**5.7**	**165.0**	**38.0**	**<0.01**	---	---	---	---
46.0	2.9	0.013	0.99	0.005	0.01	0.140	5.01	5.7	166.0	36.0	<0.01	---	---	---	---
50.0	2.9	0.014	1.03	0.006	0.01	0.140	5.28	5.8	171.0	36.0	<0.01	---	<100	<2.00	388
54.0	2.9	0.021	1.00	0.006	0.01	0.101	5.11	5.6	169.0	42.0	<0.01	---	---	---	---
62.0	3.1	0.014	1.01	0.006	0.01	0.140	5.55	5.9	179.0	32.0	---	---	---	---	---
62.0	3.2	0.018	1.02	0.006	0.01	0.140	5.68	5.9	180.0	32.0	---	5.8	---	---	---
62.0	3.0	0.016	1.02	0.005	0.01	0.121	5.79	5.9	182.0	30.0	---	---	---	---	---
68.0	3.0	0.023	1.04	0.006	0.01	0.130	5.93	6.0	185.0	30.0	---	---	---	---	---
76.0	3.2	0.018	1.10	0.006	0.01	0.140	6.14	6.2	194.0	28.0	---	---	---	---	---
79.0	3.2	0.016	1.10	0.006	0.01	0.130	6.20	6.3	197.0	28.0	---	---	---	---	---
81.0	**3.4**	**0.019**	**1.14**	**0.007**	**0.01**	**0.121**	**6.27**	**6.4**	**202.0**	**26.0**	---	---	---	---	---
86.0	3.4	0.016	1.15	0.007	0.01	0.101	6.40	6.5	206.0	26.0	<0.01	---	---	---	---
88.0	3.7	0.014	1.17	0.006	0.01	0.091	6.38	6.7	212.0	26.0	<0.01	---	---	---	---
85.0	3.2	0.018	1.18	0.007	0.01	0.121	6.39	6.7	212.0	23.0	<0.01	---	---	---	---
91.0	4.3	0.016	1.23	0.008	0.01	0.130	6.39	7.0	225.0	22.0	---	3.7	26.1	<2.00	384
108.0	4.3	0.023	1.27	0.008	0.01	0.121	6.44	7.0	226.0	20.0	---	---	---	---	---
96.0	4.9	0.010	1.27	0.008	0.01	0.111	6.48	7.1	229.0	20.0	---	---	---	---	---
96.0	4.2	0.010	1.31	0.008	0.01	0.130	6.55	7.2	233.0	18.0	<0.01	---	---	---	---
100.0	5.0	0.013	1.32	0.008	0.01	0.111	6.65	7.3	237.0	18.0	<0.01	---	---	---	---
105.0	5.0	0.013	1.36	0.009	0.01	0.111	6.58	7.4	238.0	18.0	<0.01	---	---	---	---
108.0	**4.3**	**0.016**	**1.37**	**0.009**	**0.01**	**0.121**	**6.62**	**7.6**	**244.0**	**16.0**	**<0.01**	**2.8**	**4.0**	**<2.00**	**285**
124.0	4.6	0.016	1.39	0.015	0.01	0.130	6.28	7.7	243.0	15.0	---	---	---	---	---
108.0	4.4	0.010	1.39	0.012	0.01	0.150	6.31	7.8	252.0	9.0	---	---	---	---	---
121.0	4.3	0.020	1.33	0.006	0.05	0.238	5.76	8.2	253.0	11.8	---	---	3.3	<2.00	341.9
142.0	4.3	0.020	1.39	0.005	0.06	0.218	5.96	8.9	266.0	9.2	<0.01	---	2.9	<2.00	530

$$\alpha = \frac{C-GW}{SW-GW} \qquad (1)$$

The fraction of injected water mixed with native ground water was determined by α (Eq. 1). In this equation SW, GW, and C are chloride concentrations, in milligrams per liter (mg/L), of initial treated surface water, ground water, and sample, respectively. In the field, where in situ measurements of chloride are not always possible, the same relation can be used by replacing chloride with specific-conductance readings.

SAMPLING SCHEDULES

During the injection period, samples of injected water collected at the test site displayed concentration changes with time. These variations were attributed to quality fluctuations of the AICW water. Water quality in the AICW normally displays seasonal variations that are related to changes in flow and water quality in the Pee Dee and Waccamaw Rivers, the two main freshwater contributors. During the summer and spring months, for example, drainage from adjacent swampland into the AICW and feeder streams significantly impacts the pH, temperature, alkalinity, dissolved oxygen, trace-metal concentrations, and amounts of fecal coliform bacteria in the water (State Water Resources Assessment, 1983). The product water of the treatment plant consequently shows relatively small but noticeable variations in concentration of some of these characteristics throughout the year.

During the storage period, samples were collected weekly after the well was operated for a 1- to 2-hour period. Changes in water quality during this period represent the effects of chemical reactions (no mixing) between injected water and the aquifer matrix. Changes observed in the storage period are indicative of reactions that developed rapidly near the well bore.

During the recovery period, the well was pumped continuously and samples were collected daily. The frequency of the sampling was determined by the specific-conductance measurements. Early samples appeared more affected by chemical reactions, and later ones appeared more affected by mixing. Early samples represented water stored close to the well bore, whereas later samples represented water stored farther away in the aquifer.

CONSERVATIVE MIXING

The analysis of water quality began with the study of chloride concentration, as it displayed the simplest chemical behavior, being affected only by mixing. Concentration trends for this ion, therefore, served as a guide in studying the other constituents.

Chloride (Cl^-). Concentration changes with time are shown in Figure 13(b). During injection, storage, and most of the recovery period, the chloride concentration remained nearly constant (20 mg/L). This suggests that mixing between injected water (20 mg/L) and native ground water (141 mg/L) did not occur during most of the injection period and what mixing occurred was restricted to a narrow zone observed upon recovery of nearly 70 percent of the injected water. Concentrations for the other ions, however, did not follow the chloride trend, indicating that chemical reactions and mixing affected their concentrations.

Temperature. A physical property that provided valuable information in understanding and defining trends was temperature. Temperature, like the chloride ion, was strongly influenced by mixing (endothermic or exothermic reactions were assumed unimportant); however, temperature, unlike the chloride ion, changed throughout injection. These fluctuations, which were attributed to seasonal variations, complicated the chemical interpretation by masking trends that resulted from mixing. Thus, it was necessary to understand how the variations during injection affected and controlled the temperature trend during recovery.

Figure 14(a) shows that mixing had a strong influence on the behavior of the temperature curve. During injection, temperature initially changed from 15°C to 18°C; then it decreased to a minimum of 14.5°C; thereafter, temperature progressively increased to a maximum of 23°C

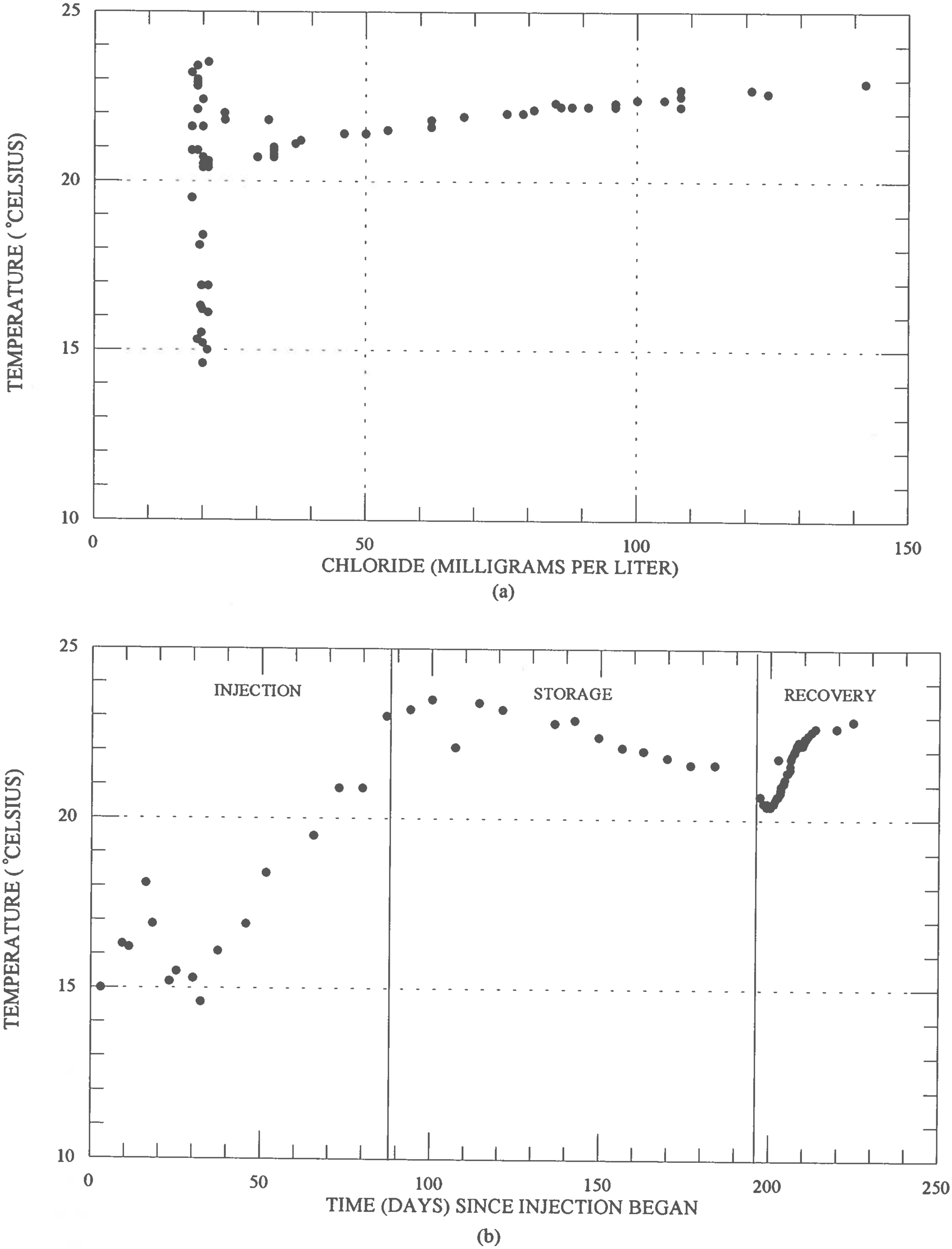


Figure 14. Comparison of temperature with chloride concentration (a) and with time (b).

(Fig. 14). These variations were the result of seasonal ambient temperature changes. During storage and recovery, temperature decreased continuously until it reached a low of 20.5°C; then it increased to a background level of 24°C. Water near the well bore retained a temperature similar to that of water from the last injection. Moreover, water temperature gradually decreased as distance from the well increased, showing that a colder water was injected during the middle of the injection period. After reaching a low point during early recovery, the temperature gradually increased. This increase was due not only to mixing but also to warmer water being injected at the beginning of the injection period. Hence, the storage curve and early portion of the recovery curve became an attenuated mirror image of the injection curve.

CONCENTRATION TREND PATTERNS

The description of the ions is divided into four groups. The classification was based on the concentration trend patterns observed during the storage and recovery periods. Grouping ions helps to visualize related geochemical processes. Group I was characterized by an early decrease, a stable period, a peak, and finally an exponentially decreasing limb that brought ion concentrations close to background levels. Group II had a rapidly increasing limb that plateaued during the middle of the storage period before it progressively increased toward background levels. Group III, which encompassed most of the ions, was defined by a rapidly increasing limb that peaked during the middle of the storage period and decreased rapidly to a minimum level during early recovery before increasing again to background levels. Group IV, a small group, was characterized by an early decrease followed by concentrations below the detection limit. In the following discussion, water quality factors have been graphed in two ways: (a) with chloride, in order to differentiate the effects of mixing from those of chemical reactions, and (b) with time, to illustrate overall trends.

Group I

This group had three constituents and one property: aluminum, total organic carbon, phosphate, and pH. Phosphate was added to this group because it conformed to the overall pattern even though it initially increased rather than decreased.

Aluminum (Al^{3+}). Aluminum concentrations during injection ranged from 0.3 to 0.7 mg/L (Fig. 15), reflecting changes in the source water and adjustments in the treatment process at the treatment plant. During storage, aluminum rapidly decreased to less than 0.05 mg/L and remained at this concentration for the duration of the storage period. Early during recovery, aluminum rapidly increased and then peaked to a maximum of 0.3 mg/L; thereafter, the concentration decreased toward background levels. Aluminum, probably in the presence of silica, precipitated as clay minerals (Hem, 1985). This process, however, did not continue far into the aquifer. Later, the absence of silica caused an apparent increase in aluminum. Farther from the well bore, aluminum decreased because of mixing with the native ground water (Fig. 15(a)). Aluminum concentration in the recovered water was always less than in the injected water and most often below the EPA suggested level (SL).

Total organic carbon (TOC). During injection, the TOC concentration was near 7 mg/L; during storage it sharply decreased to 4 mg/L; and during recovery it initially increased to 5.7 mg/L and then gradually decreased to the aquifer concentration level of 2.0 mg/L (Fig. 16). The TOC might have decreased because of aerobic oxidation of organic matter. Aerobic oxidation, however, was limited to an area close to the well bore, where dissolved oxygen was available. Away from the well, oxidation of organic carbon was limited and TOC concentrations did not change much from initial levels found during injection (this caused the peaks in early recovery). Lastly, TOC decreased to aquifer levels because of mixing (Fig. 16). The recovered water consistently had TOC concentrations lower than those in the injected water (Fig. 16).

Phosphate (PO_4^{3-}). Phosphate ions had an initial concentration of less than 0.03 mg/L; then it increased to 0.1 and then to 0.87 mg/L and, later, sharply decreased to background levels (0.18 mg/L)(Fig. 17). The phosphate

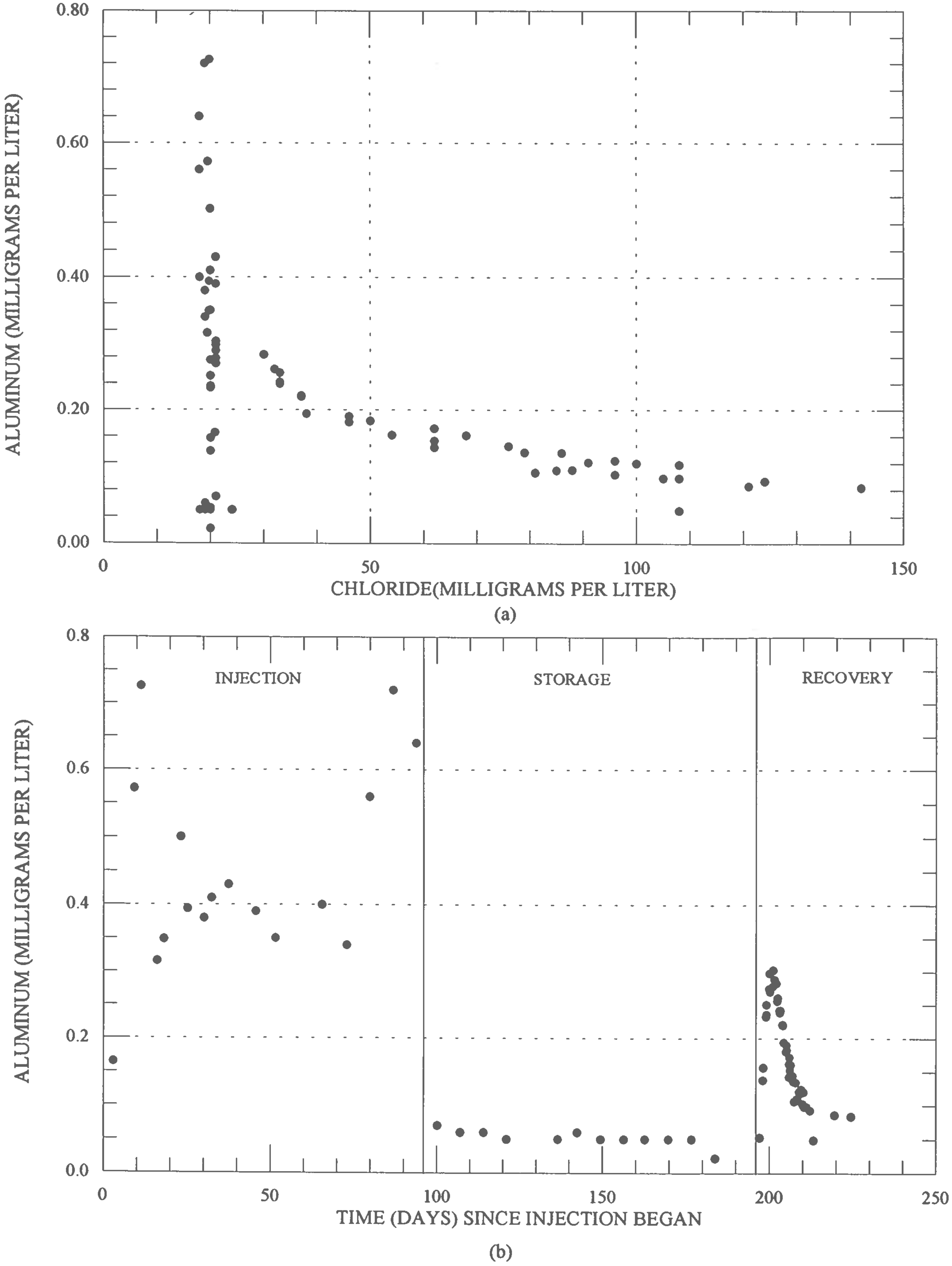


Figure 15. Comparison of aluminum with chloride concentration (a) and with time (b).

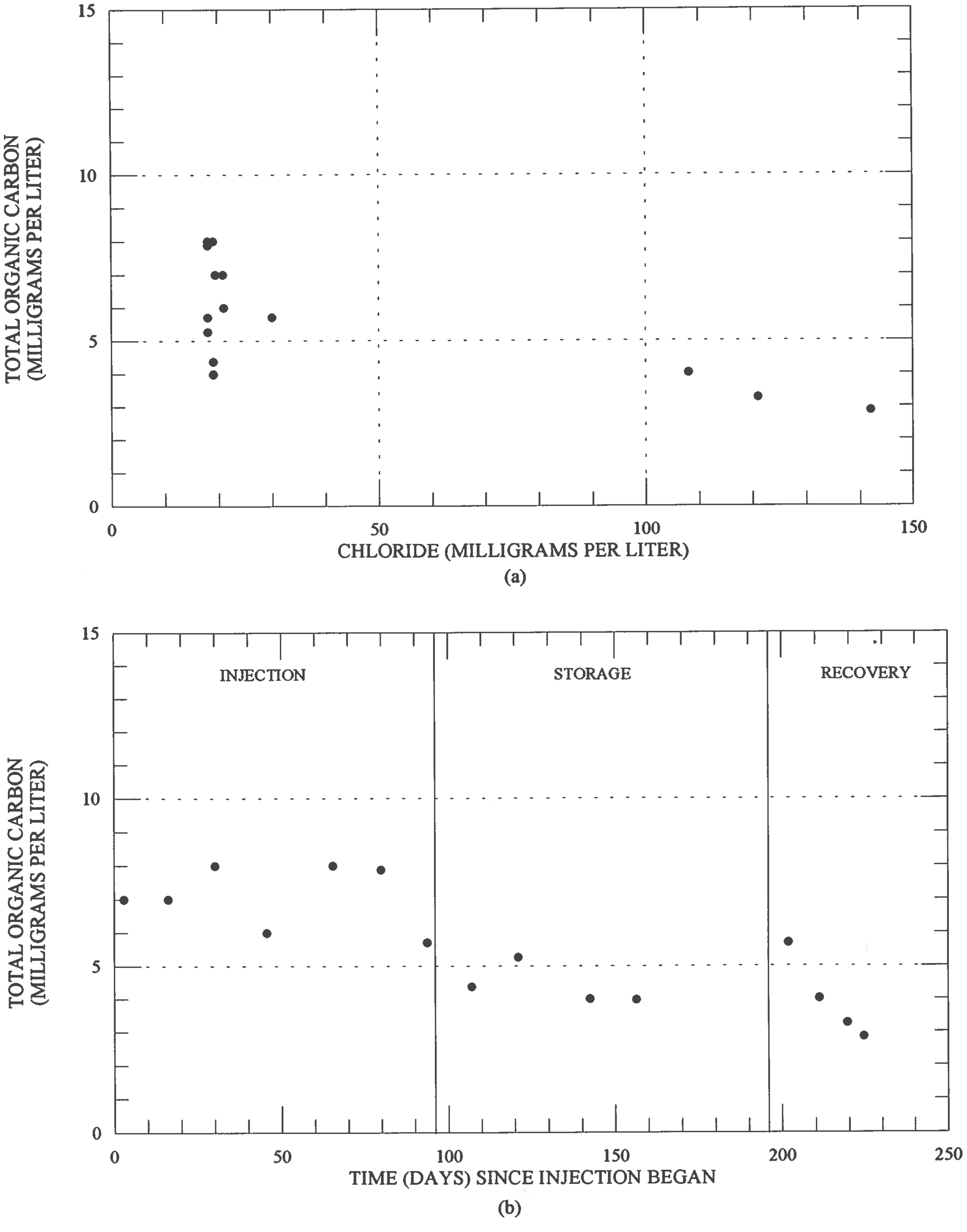


Figure 16. Comparison of total organic carbon with chloride concentration (a) and with time (b).

and fluoride ions might have increased because of the formation of a hydroxyapatite surface complex and not because of the dissolution of fluorapatite; Zack (1980) noted that fluorapatite is insoluble at a pH above 8. The phosphate ion appeared, later on, to have reprecipitated as calcium phosphate. During recovery, the phosphate concentration decreased because of mixing (Fig. 17).

pH. Despite the apparent simplicity with which pH measurements were made, pH was a property difficult to monitor in the field. Consistent values, especially during recovery, were not always obtained. During storage, pH values first decreased and then increased. During early recovery, pH values rapidly increased to a maximum of 9.2 and during late recovery they decreased toward background levels (Fig. 18). The pH changes have been associated with oxidation of organic matter and pyrite (pH decrease), calcite dissolution (pH increase), and mixing (pH decrease).

Group II

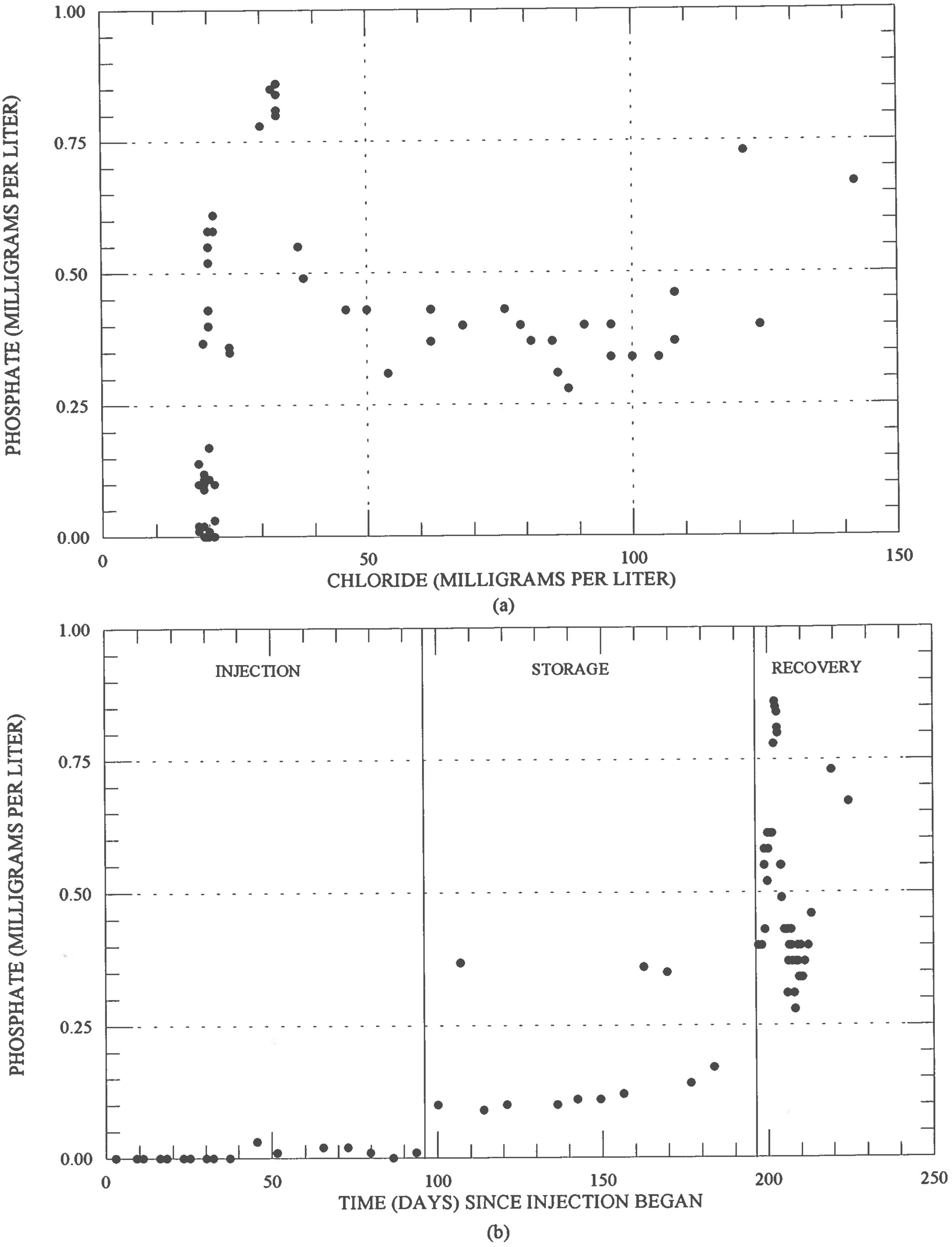
In this group are included alkalinity, fluoride, potassium, sodium, and specific conductance. Potassium did not fully conform with the general trend.

Alkalinity (as $CaCO_3$). Initial values of alkalinity were between 20 and 40 mg/L as $CaCO_3$ (Fig. 19). During early storage, alkalinity increased to about 100 mg/L. During recovery it progressively increased to 500 mg/L (background level). Alkalinity might have increased because of calcite dissolution as a consequence of organic-matter oxidation to carbon dioxide. The rate of increase of alkalinity was noted to be faster during the recovery period (farther away from well bore) and slower during the storage period (close to the well bore). The increase of alkalinity in the recovered water represented a significant improvement in the quality of the water. Higher alkalinity and pH are desirable characteristics in the recovered water, as they decrease the aggressiveness and increase the buffer capacity of the recovered water.

Sodium (Na^+). Concentrations, during injection, remained constant at about 20 mg/L; during storage, sodium increased rapidly to 40 mg/L; and during recovery, sodium progressively increased to 270 mg/L toward the background level (Fig. 20). This increase in sodium was likely a result of desorption from marine clay caused by exchange with calcium. The concentration of the latter was likely raised by calcite dissolution. Sodium concentration, especially during the recovery period, exceeded the EPA's suggested levels; therefore, this constituent would be a limiting factor in determining the volume of usable recovered water.

Fluoride (F^-). During injection, fluoride concentrations fluctuated around an average of 1 mg/L; during late storage, fluoride progressively increased to 1.25 mg/L; and during recovery, fluoride increased rapidly to 4.2 mg/L, which is the background level (Fig. 21). The processes that might have increased the fluoride concentration were: formation of a surface complex on the fluorapatite minerals (see phosphate section); exchange of hydroxyl ions available in solution, with fluoride ions from fluorapatite; and mixing. The secondary drinking water standard for fluoride, as defined by the EPA, is 4 mg/L. Throughout most of the test, fluoride concentrations were below 2 mg/L, but near the end of the recovery, fluoride concentrations increased and exceeded the EPA limit. Fluoride concentration, consequently, would be another limiting factor in determining the volume of potable water that can be recovered.

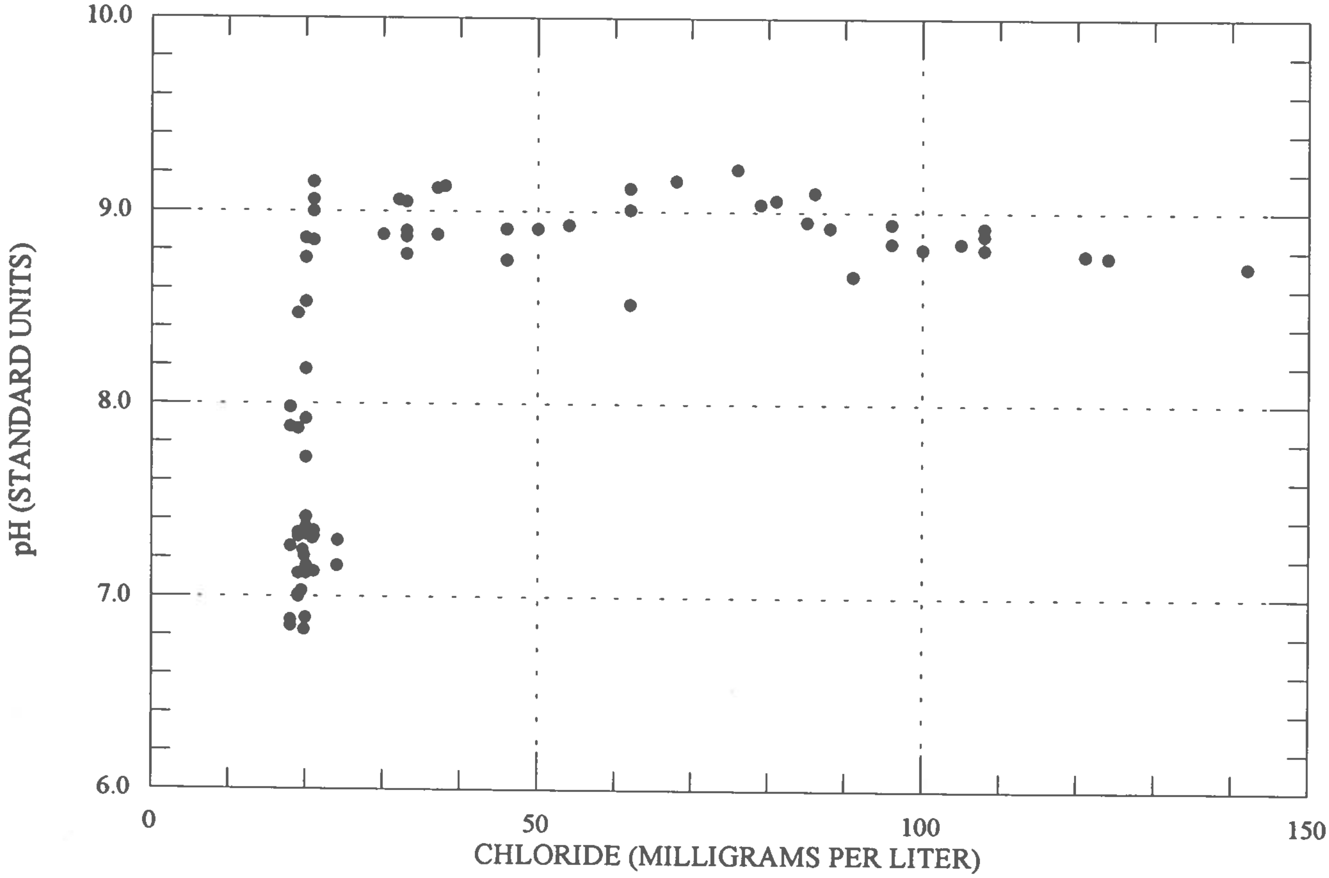
Potassium (K^+). During injection, potassium concentrations varied between 2 and 3 mg/L; during early storage, potassium increased from an initial value of 3 to about 3.3 mg/L; during late storage, potassium decreased to 2.5 mg/L; and during recovery, potassium increased to 6 mg/L the background level (Fig. 22). The fluctuations in concentration during injection clearly influenced the concentration trend observed during storage. The potassium curve during storage and early recovery appeared to be a mirror image of the curve during injection. The increase of potassium in solution observed during the recovery period was attributed to the dissolution of K-feldspar minerals, which were common in the Black Creek core samples. Scanning electron microscope (SEM) photographs of Black Creek core samples show that feldspar



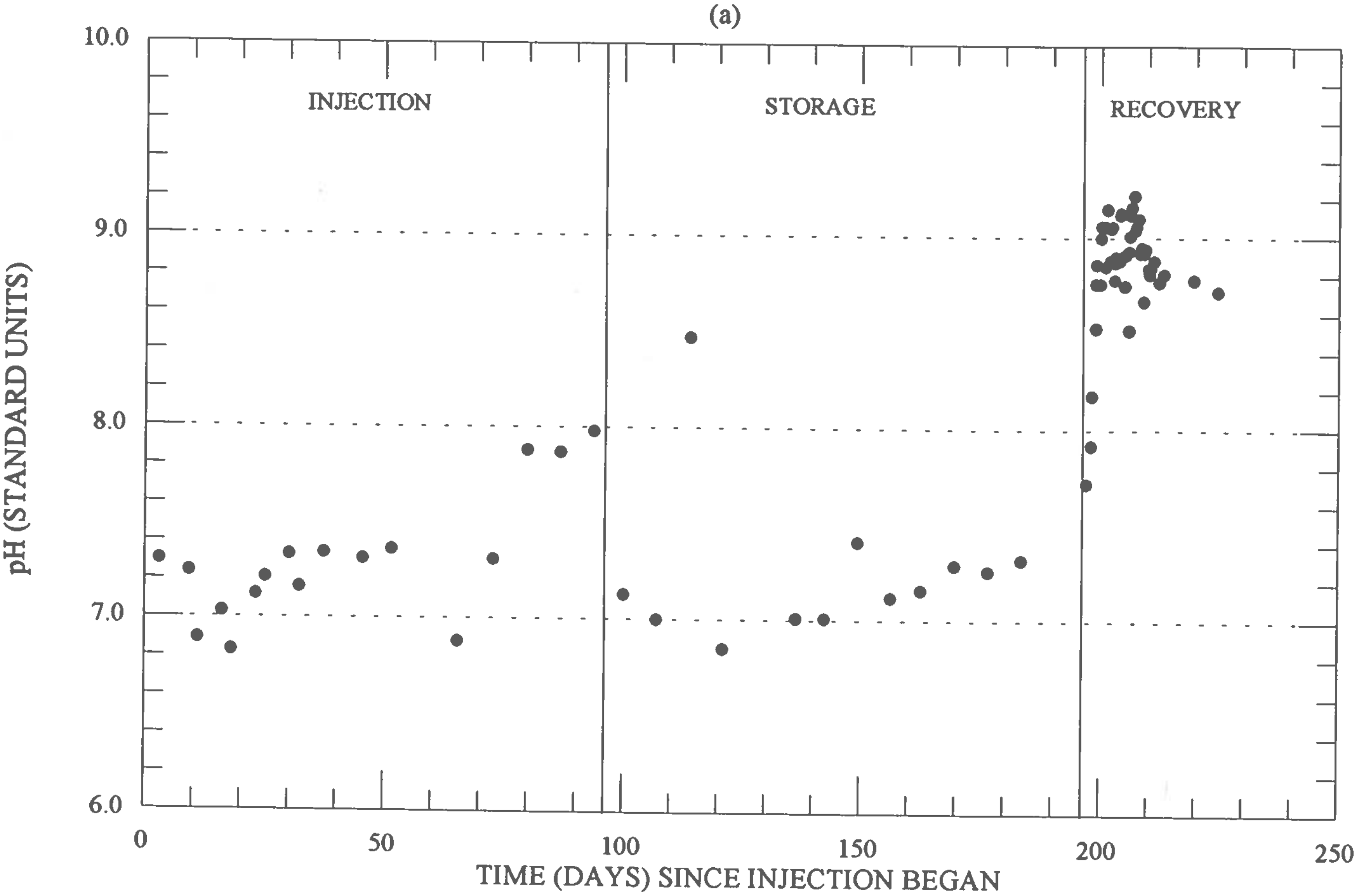
(a)

(b)

Figure 17. Comparison of phosphate with chloride concentration (a) and with time (b).



(a)



(b)

Figure 18. Comparison of pH with chloride concentration (a) and with time (b).

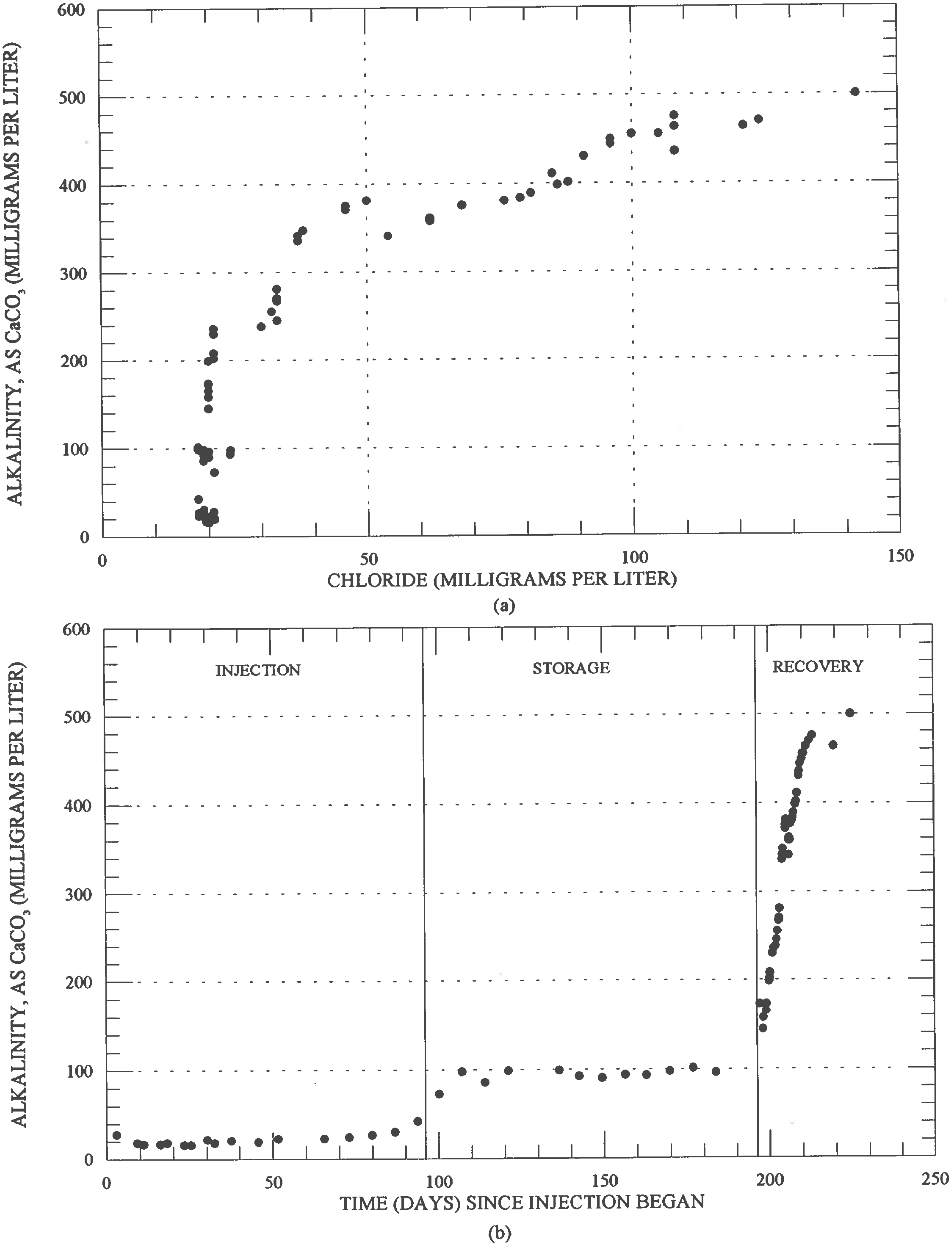


Figure 19. Comparison of alkalinity with chloride concentration (a) and with time (b).

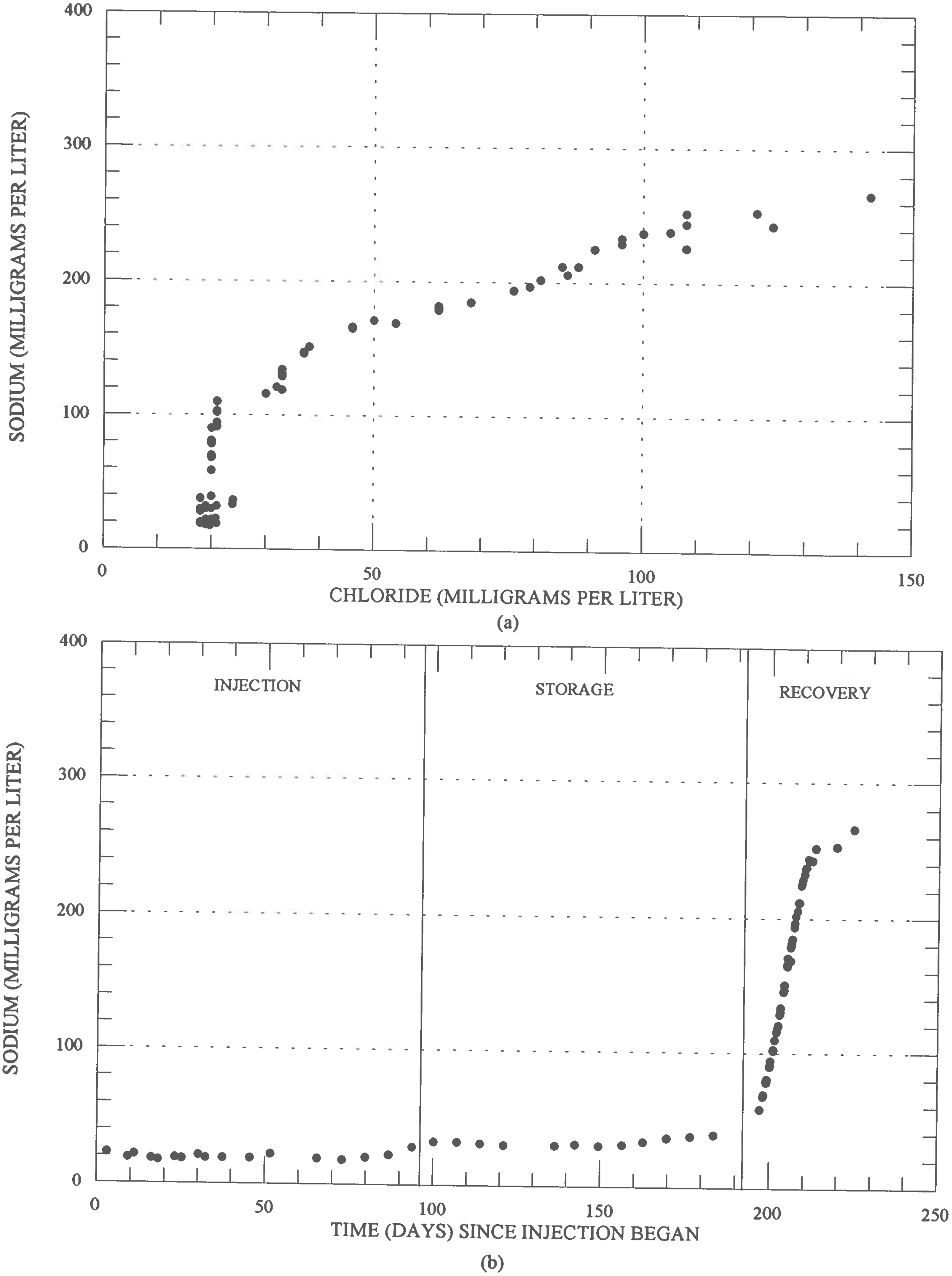
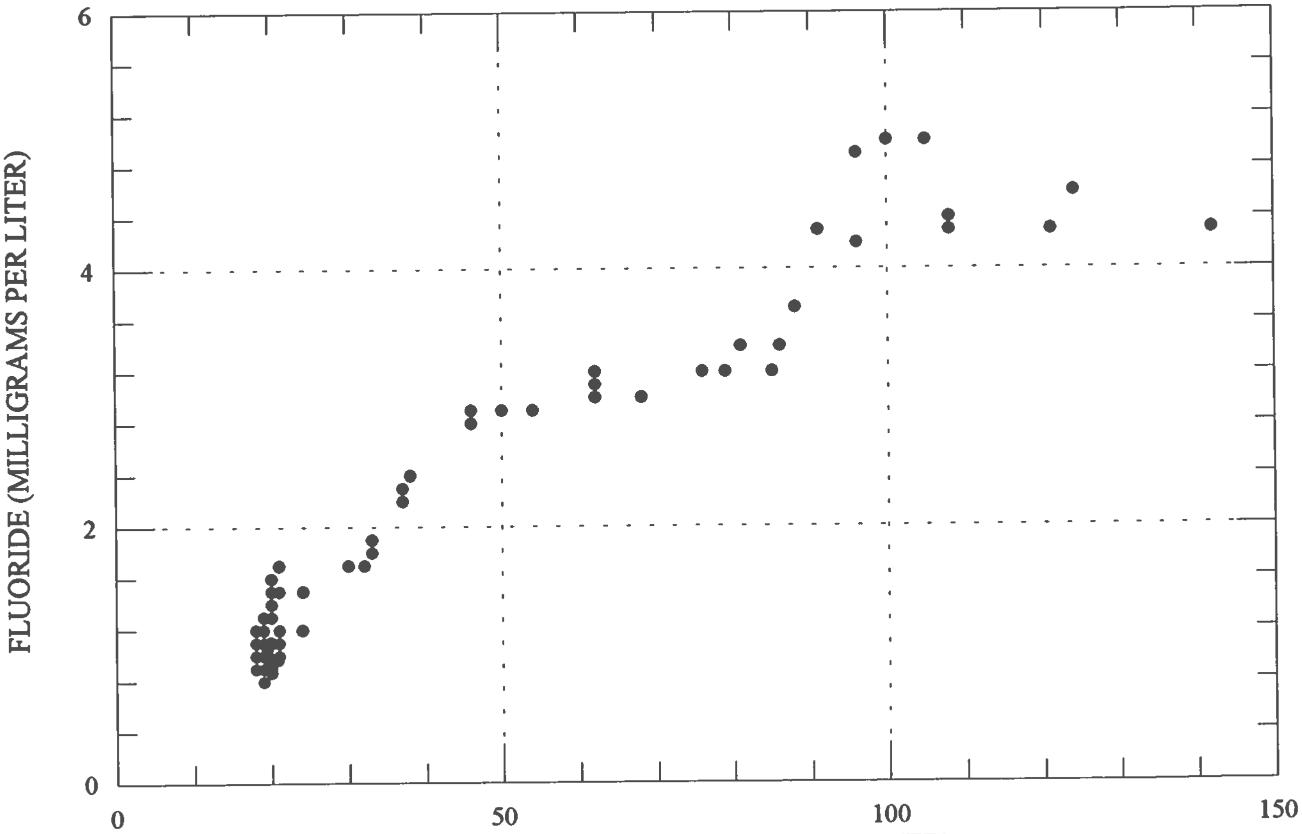
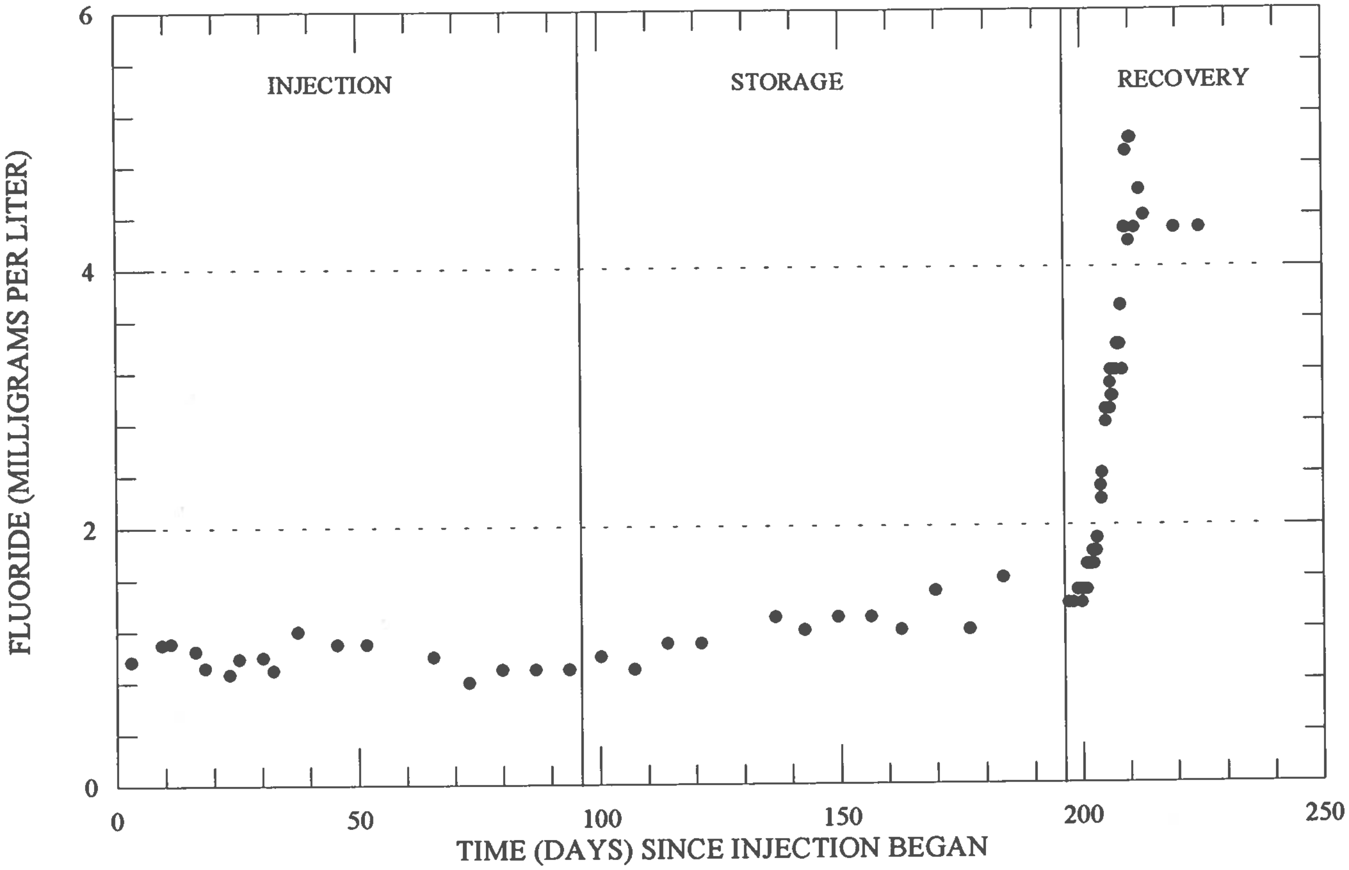


Figure 20. Comparison of sodium with chloride concentration (a) and with time (b).



(a)



(b)

Figure 21. Comparison of fluoride with chloride concentration (a) and with time (b).

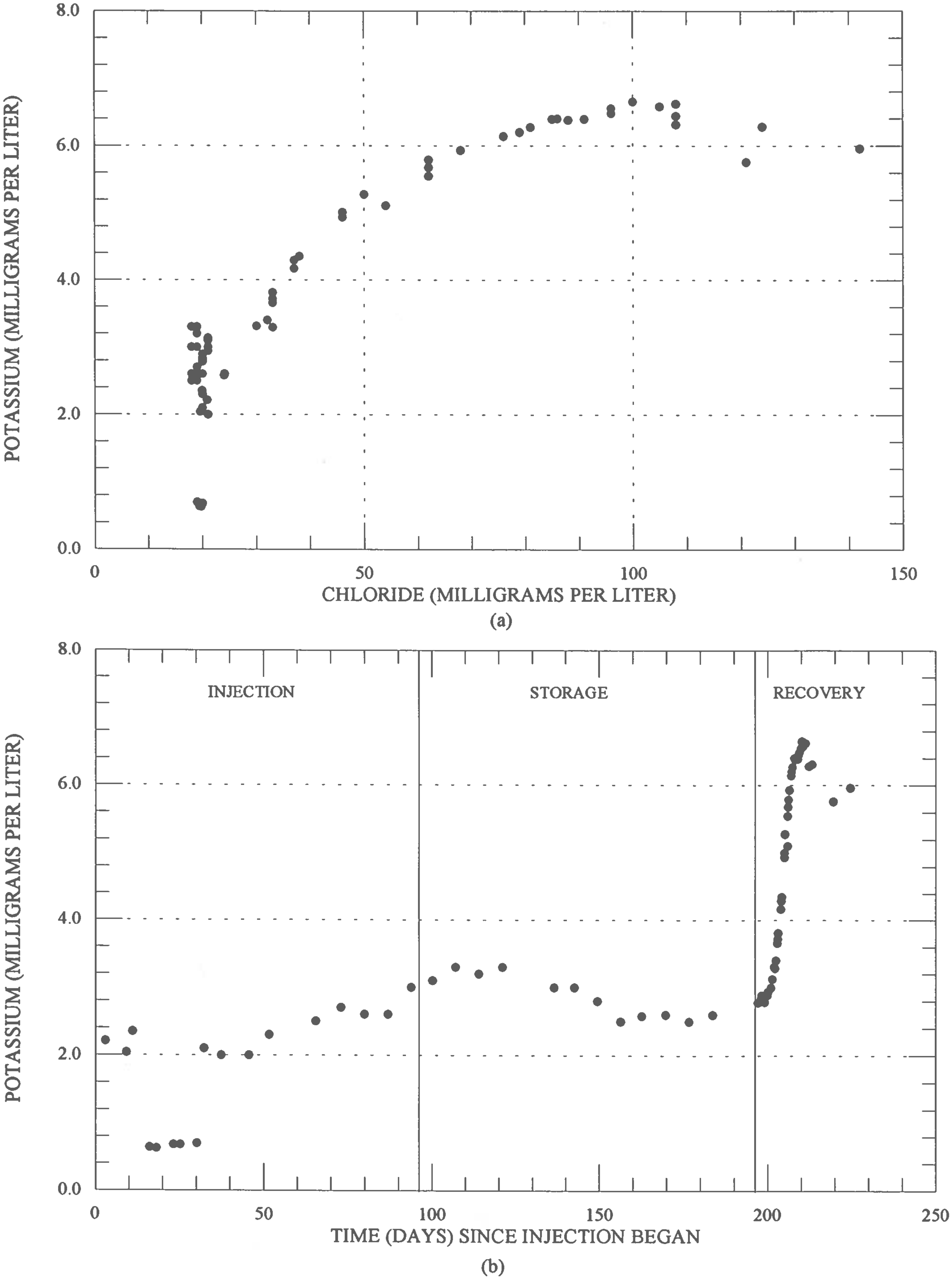


Figure 22. Comparison of potassium with chloride concentration (a) and with time (b).

dissolution is an ongoing process in this system (Castro and others, 1995).

Specific conductance. The specific-conductance curve closely resembled those of alkalinity and sodium. This was expected because bicarbonate and sodium were the major ions in solution and, therefore, had the greatest influence on the specific conductance (Fig. 23).

Group III

This is the largest group and includes calcium, iron, magnesium, silica, and sulfate ions.

Calcium (Ca^{2+}). The calcium concentration, during injection, ranged from 10 to 16 mg/L; during storage, calcium increased to a maximum of 31 mg/L; and thereafter it progressively decreased to 8 mg/L (Fig. 24). Calcium concentration might have increased because of dissolution of calcite (a calcium carbonate phase such as aragonite) and decreased because of calcium-sodium exchange and mixing. If during recovery, the exchange process developed at a lower rate than calcite dissolution, dissolution would have caused a net increase of dissolved calcium. Later on, if exchange was faster, it could have caused calcium concentrations to decrease. Low values of calcium concentration during early storage suggest slower rates of calcite dissolution. Slower calcite-dissolution rates near the well appeared to have been related to the previous injection tests. The area near the well bore was the portion of the aquifer that was exposed to the injected water the longest and, therefore, was more susceptible to chemical reactions. It may be that with each additional injection test, the intensity of chemical reactions decreases in the aquifer as reactants are depleted. This phenomenon has been reported at other injection sites (Pyne, 1995). Calcium concentrations, despite the relatively large fluctuations, were always below the EPA suggested levels.

Magnesium (Mg^{2+}). The concentration of magnesium increased from 1.25 mg/L during injection to 2.5 mg/L during early storage; during late storage, however, magnesium gradually decreased to 2 mg/L; and during recovery it first decreased to a minimum of 0.8 mg/L and then increased to 1.6 mg/L (Fig. 25). Magnesium concentrations in solution may have increased because of dissolution of mica, such as glauconite and phlogopite, and decreased because of sodium exchange. It is noteworthy to mention that magnesium exchanged more readily with sodium than did calcium.

Iron (Fe^{2+}). Dissolved ferrous ions increased from an initial value of 0.05 mg/L during injection to a maximum of 0.58 mg/L during storage; thereafter it rapidly decreased to the background level of 0.02 mg/L (Fig. 26). Dissolved ferrous iron might have increased (near the well bore) because of : (a) pyrite oxidation and (b) glauconite dissolution. Ferrous iron might have decreased (farther from the well) because of: (a) ferric hydroxide precipitation and (b) mixing. SEM photographs of Black Creek core samples show the formation of pyrite framboids (Castro and others, 1995). Since the EPA secondary drinking water standard for iron is 0.2 mg/L, during storage, the concentration of dissolved iron exceeded this limit. Because the total volume of water with high iron concentration was only a small fraction of the injected volume, an appropriate blending proportion may eliminate this problem altogether.

Sulfate (SO_4^{2-}). Sulfate ions, during injection, increased from 36 to 40 mg/L, reflecting changes in the treatment process; during storage, sulfate increased to a maximum of 50 mg/L, and thereafter it decreased toward background levels (Fig. 27). Sulfate ions might have increased because of pyrite oxidation and decreased because of reduction to sulfide and mixing (Fig 27(a)). The lower sulfate concentration closer to the well bore once again tend to support the idea that previous injection tests have affected those areas closer to the well bore. The EPA secondary drinking water standard for sulfate is 250 mg/L; thus, the sulfate concentrations observed during the test were well below the suggested maximum.

Silica (SiO_2). Silica concentrations, during injection, varied from 2 to 5 mg/L; during storage, silica progres-

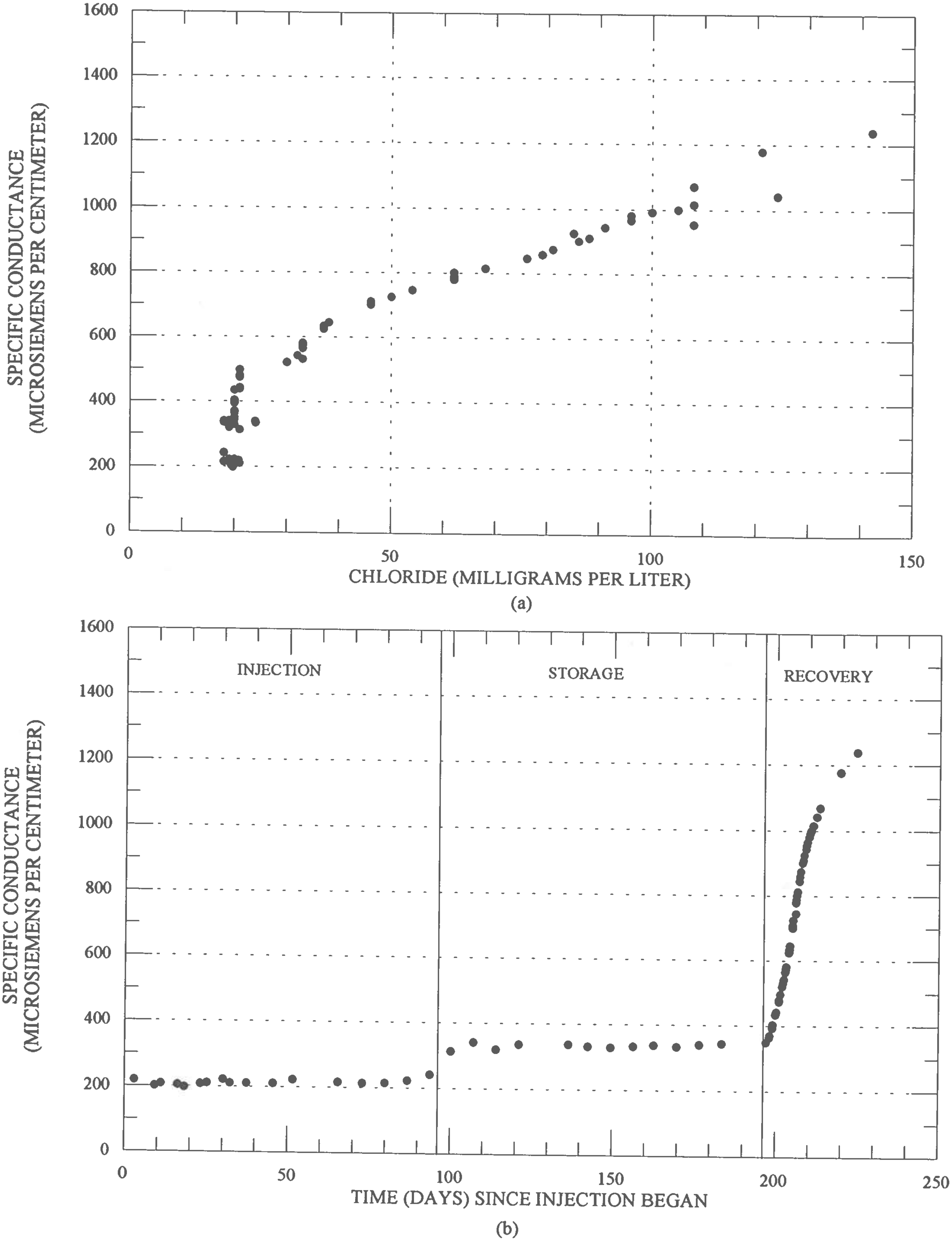
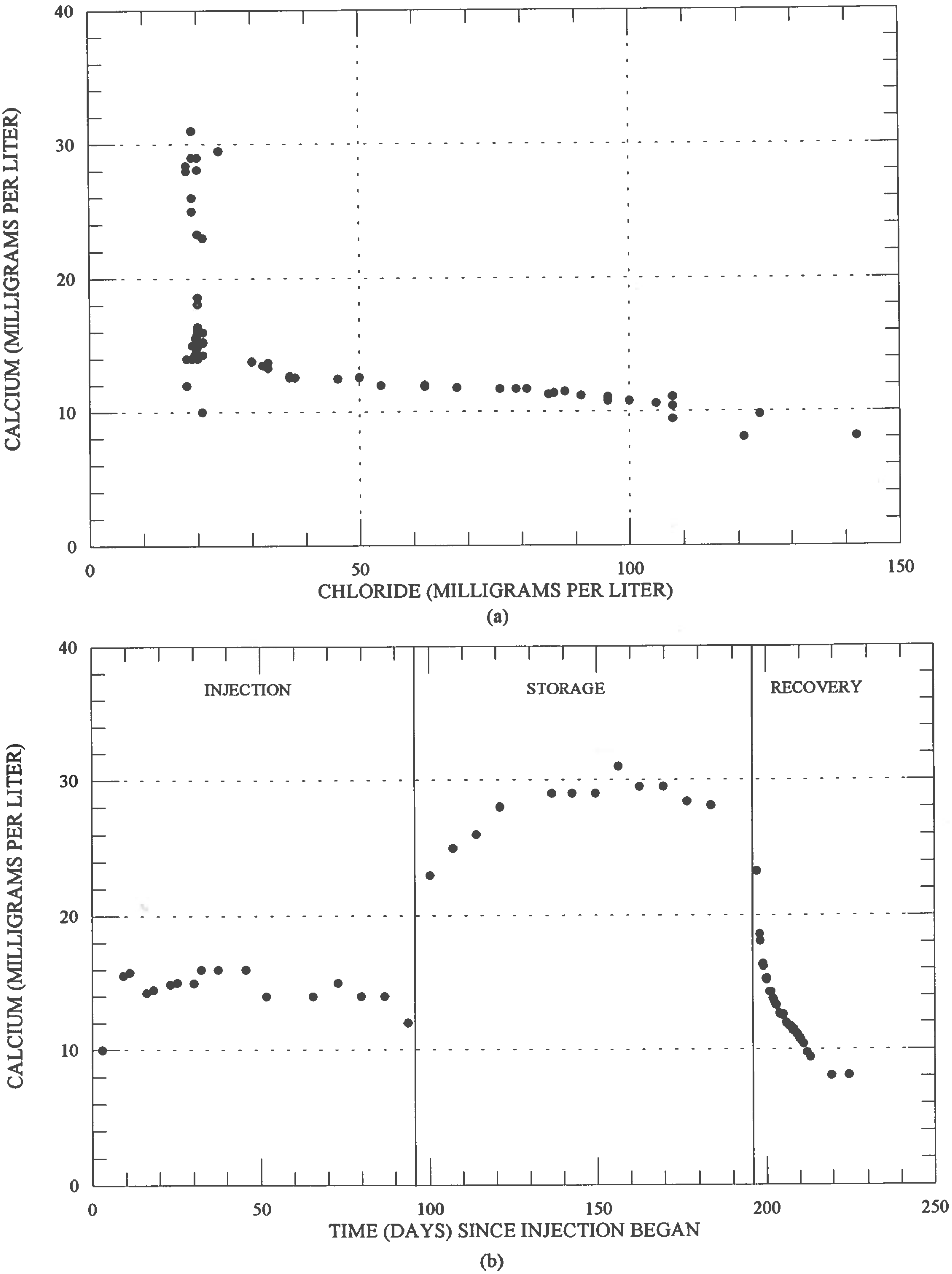


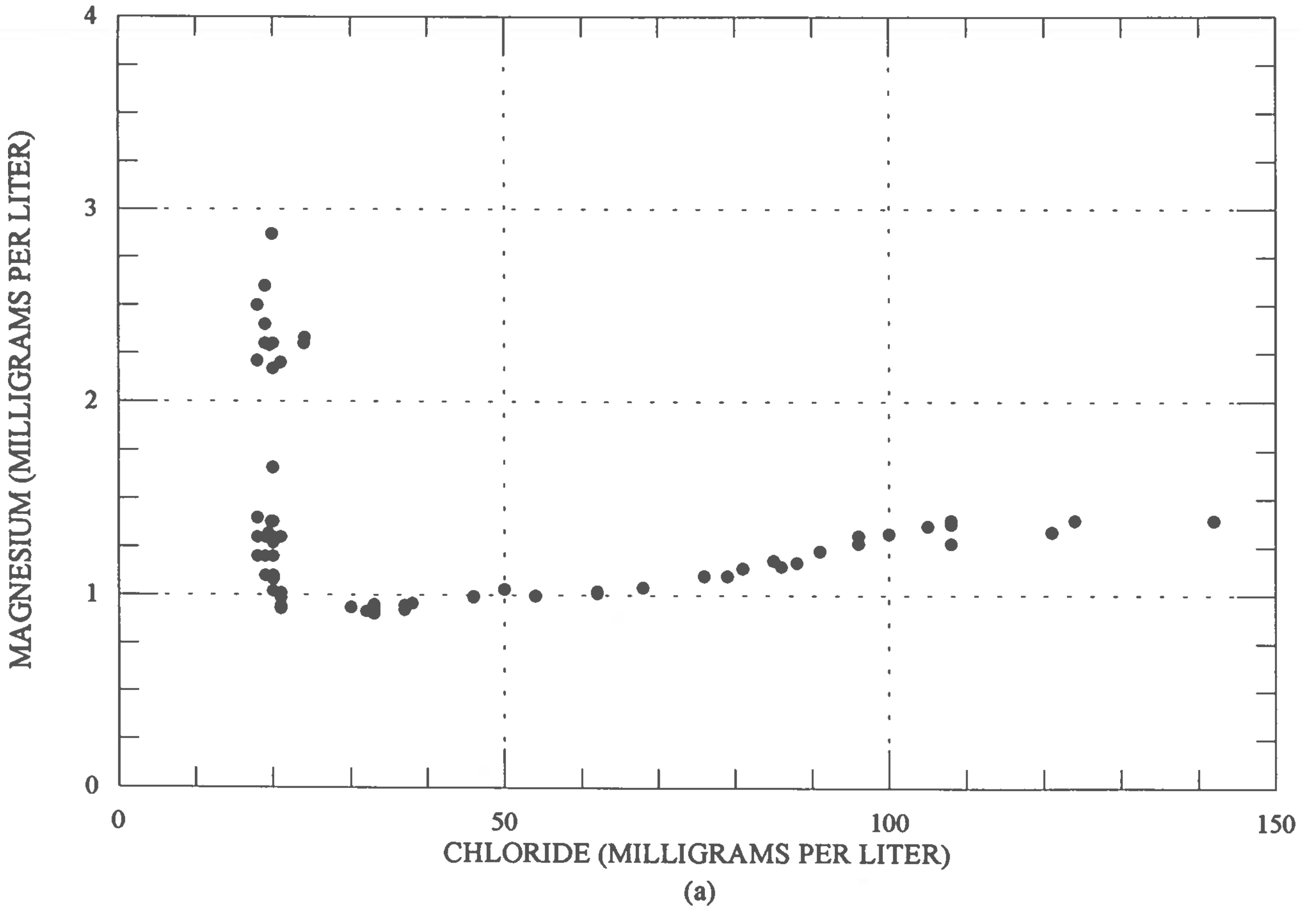
Figure 23. Comparison of specific conductance with chloride concentration (a) and with time (b).



(a)

(b)

Figure 24. Comparison of calcium with chloride concentration (a) and with time (b).



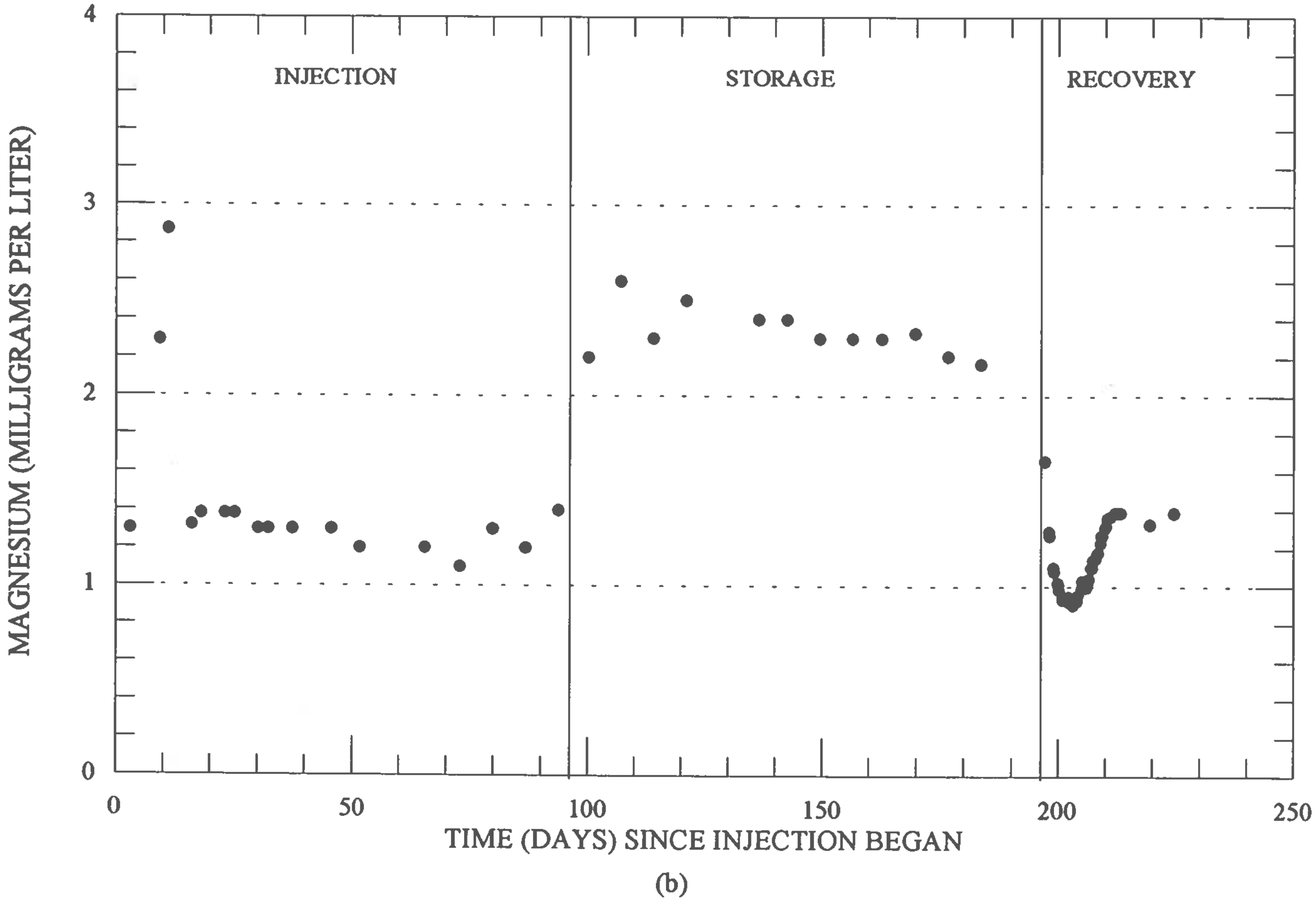
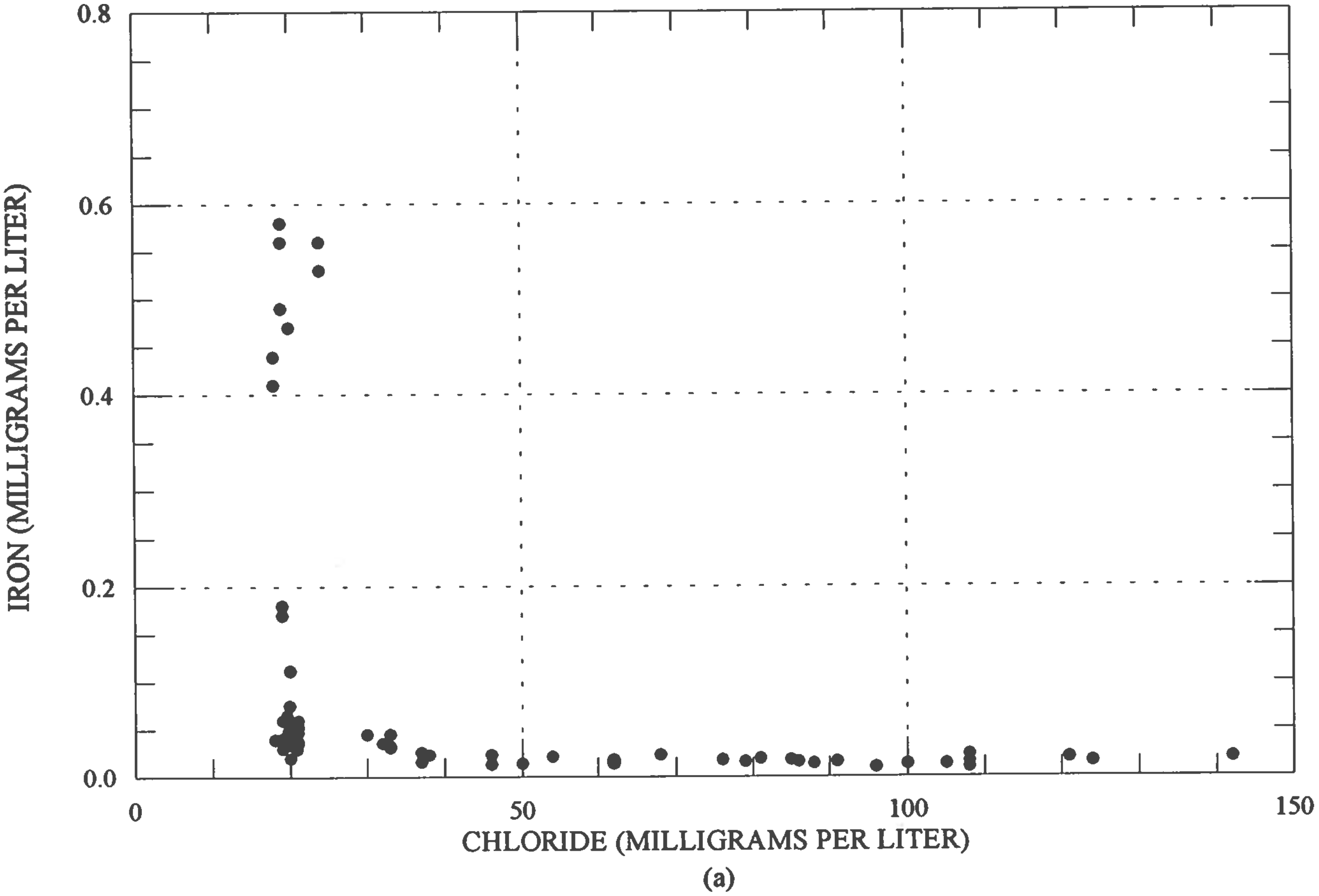


Figure 25. Comparison of magnesium with chloride concentration (a) and with time (b).



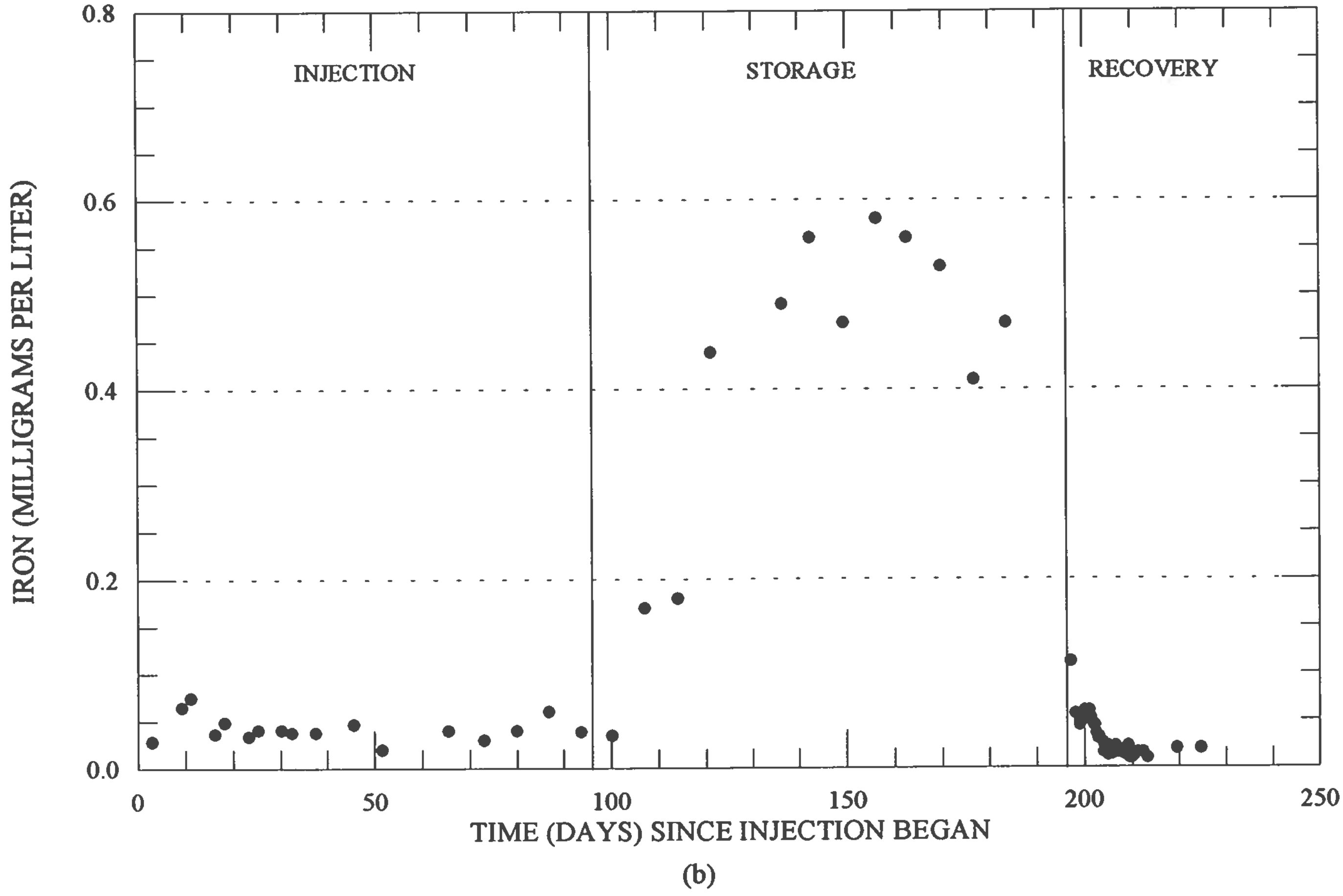


Figure 26. Comparison of iron with chloride concentration (a) and with time (b).

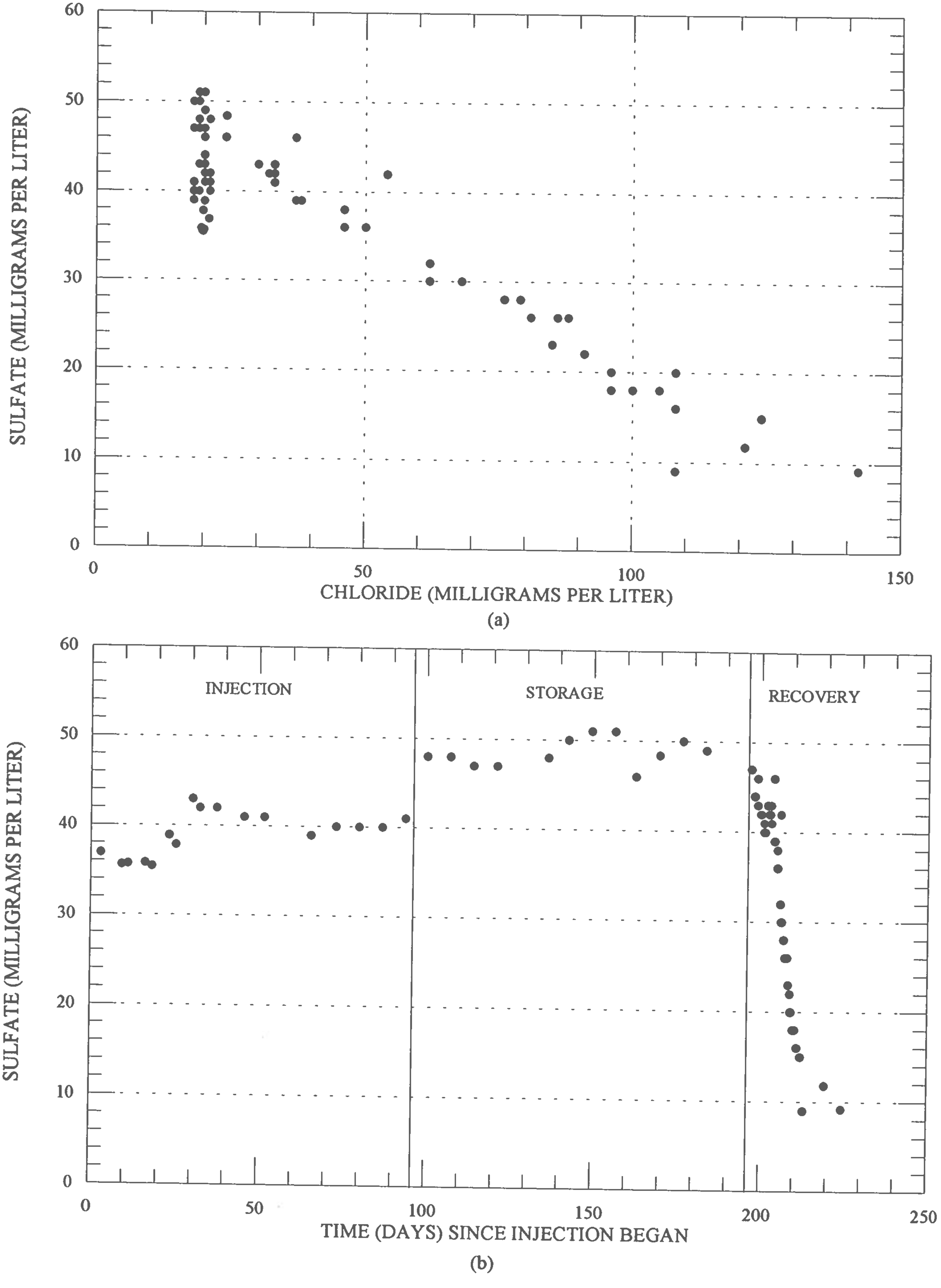
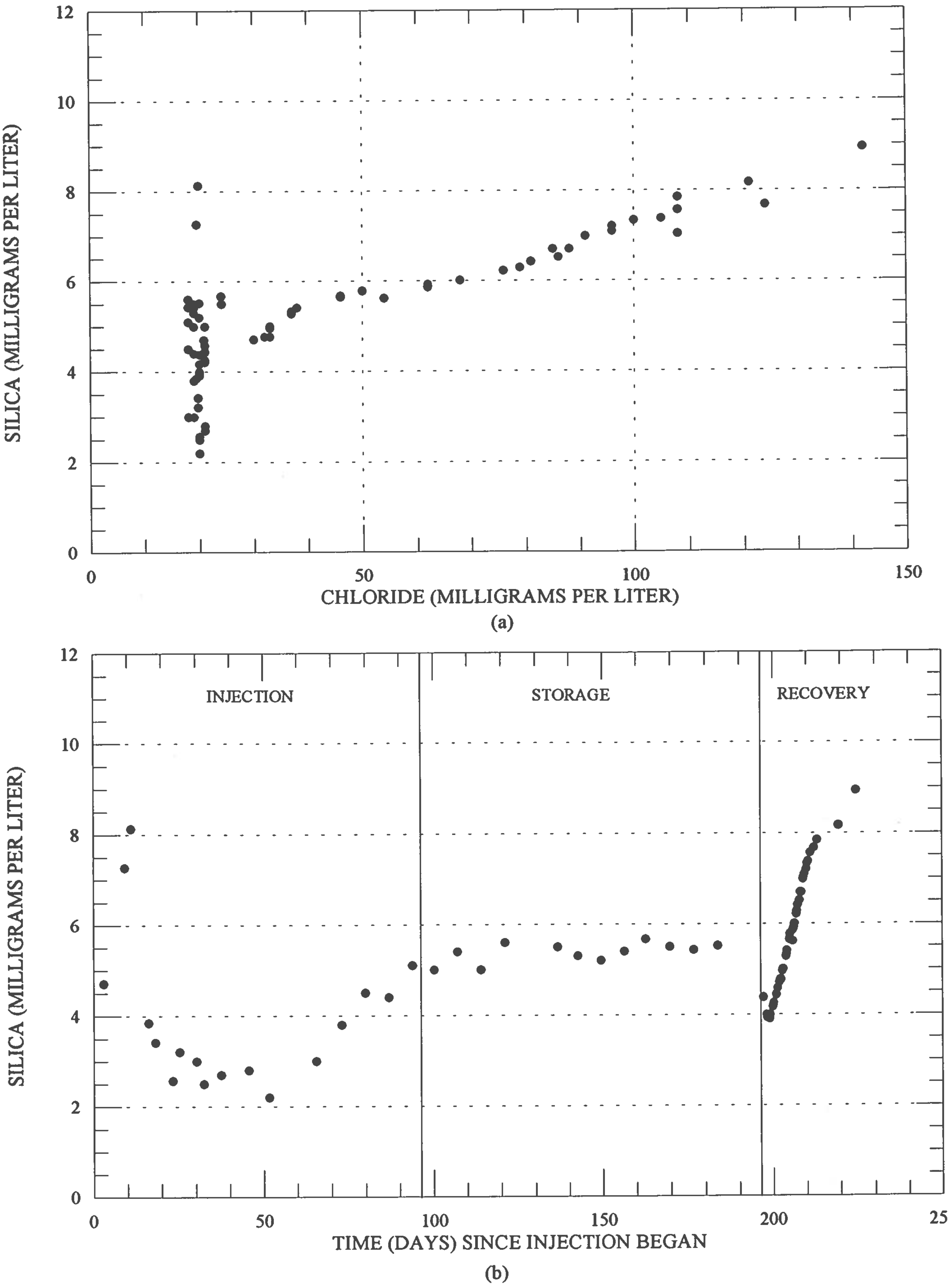


Figure 27. Comparison of sulfate with chloride concentration (a) and with time (b).



(a)

(b)

Figure 28. Comparison of silica with chloride concentration (a) and with time (b).

sively increased to 5.5 mg/L; during early recovery, it rapidly decreased to 4 mg/L and then increased toward background levels (Fig. 28). Silica might have increased in solution because of feldspar and glauconite dissolution during storage and because of mixing during recovery (Fig. 28(a)). The apparent decrease of silica during the early recovery probably reflects concentration variations in the injected water.

Group IV

The fourth group has two constituents, oxygen and nitrate. Their concentration decreased rapidly.

Dissolved oxygen (O_2). The dissolved-oxygen concentration in the injected water varied from 4.5 to 7.5 mg/L. These changes can be attributed to ambient temperature fluctuations; lower temperatures were associated with higher concentrations. Shortly after injection, the dissolved oxygen concentration decreased and remained at less than 0.1 mg/L (Fig. 29). This suggests that the oxidation of organic matter and pyrite quickly consumed most of the oxygen available in the system. These rapid reactions probably were confined to the vicinity of the well bore.

Nitrate (NO_3^-). During injection, nitrate ranged from 0.2 to 0.8 mg of nitrogen per liter. During storage, nitrate decreased rapidly and remained below detection limits throughout the recovery phase (Fig. 30). Nitrate appears to have reduced to nitrogen gas during the anaerobic oxidation of organic matter (denitrification).

GEOCHEMICAL MODELING

The chemical composition of water samples collected throughout the ASR test varied continually, and not always gradually. Initial samples, nevertheless, resembled injected water and later samples resembled native ground water. The two most important mechanisms controlling water quality are mixing and chemical interaction. Treated surface water, as it is injected into the aquifer, displaces the native ground water and forms a freshwater zone around screened portions of the aquifer (Fig. 31). In the periphery of this zone, treated surface water and ground water mixes, creating a transition zone. While chemical reactions were observed to predominate in the freshwater zone, mixing controlled concentrations in the transition zone. Both processes, nonetheless, impacted the quality of the treated water stored in the aquifer.

To identify and quantify these processes, the geochemical system was simulated with an inverse method (Plummer, 1984). In this approach, chemical reactions are inferred from observed water composition. The mass-balance model "NET geochemical reactions along a flow PATH" (NETPATH) and the speciation model PHREEQE were used for the simulations.

In modeling the geochemical evolution of the injected water, it was important to recognize and determine the effects that each type of mechanism had in the composition of the resulting water. In the freshwater zone and very close to the well bore, chemical reactions occurred in an aerobic environment. Farther away from the well in the same freshwater zone, where oxygen had been depleted, chemical reactions occurred under anaerobic conditions. Finally, in the transition zone, treated surface water and native ground water mixed under anoxic conditions. Consequently, the geochemical modeling was conceptualized in three stages. These stages were associated with three physical stages of the test: (a) early storage period; (b) late storage period; (c) recovery period. These stages and processes are listed in Table 5.

In each modeling stage, one water sample, selected from a pool of samples collected during the corresponding physical stage, was chosen to be simulated. This sample was designated *final water*. One or more samples of the treated surface water were selected to represent the chemical composition of the injected water. These samples were labeled *initial water 1 and initial water 2* (in Tables 6 through 9). *Initial water 3* was always the native ground water sample.

The underlying assumption in the modeling effort was that the *final water* evolved from *initial water* (two or more). *Initial water* (treated surface water), once injected into the ground water system, reacted with mineral phases present in the aquifer and mixed with the native ground

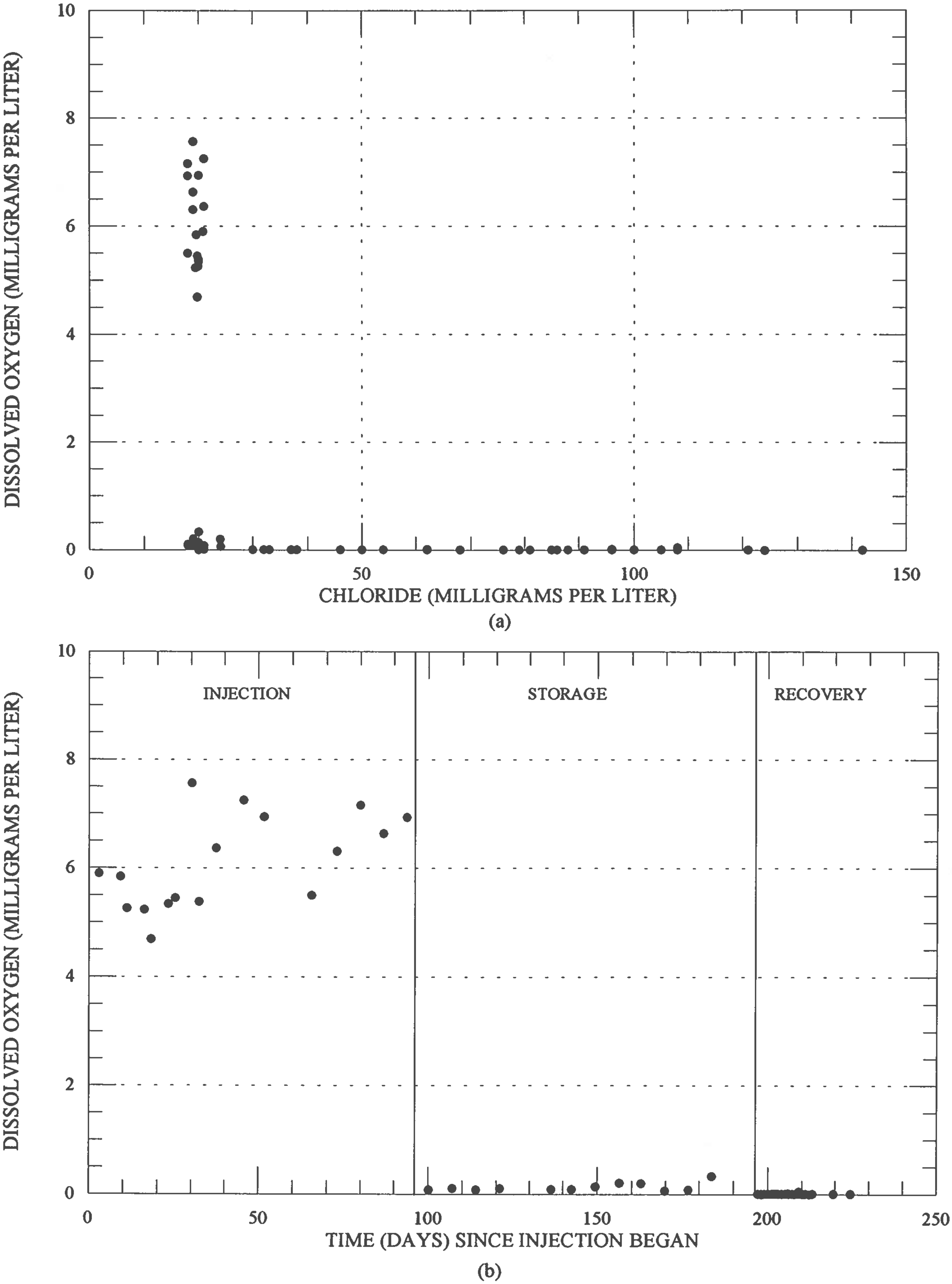


Figure 29. Comparison of oxygen with chloride concentration (a) and with time (b).

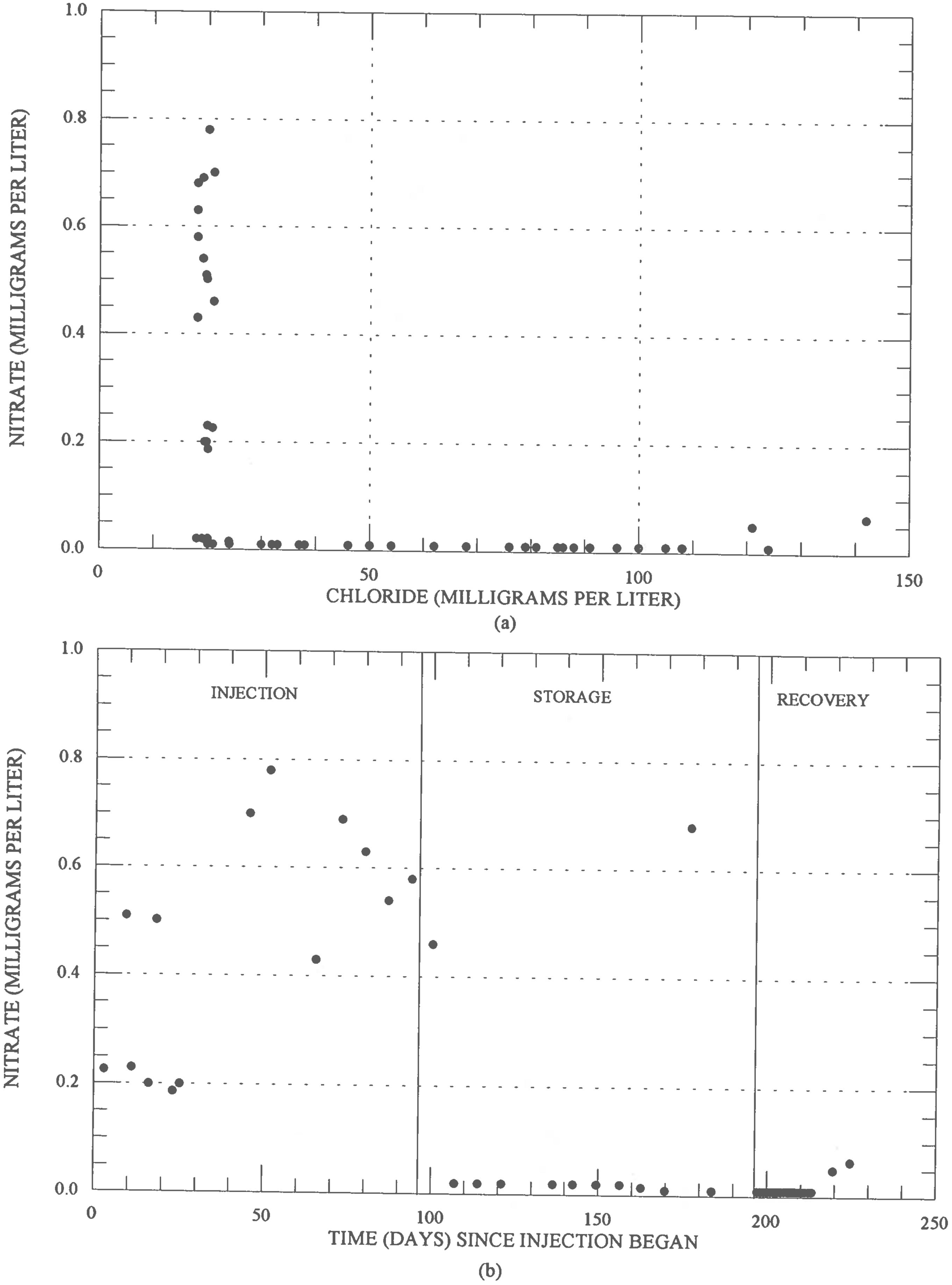


Figure 30. Comparison of nitrate with chloride concentration (a) and with time (b).

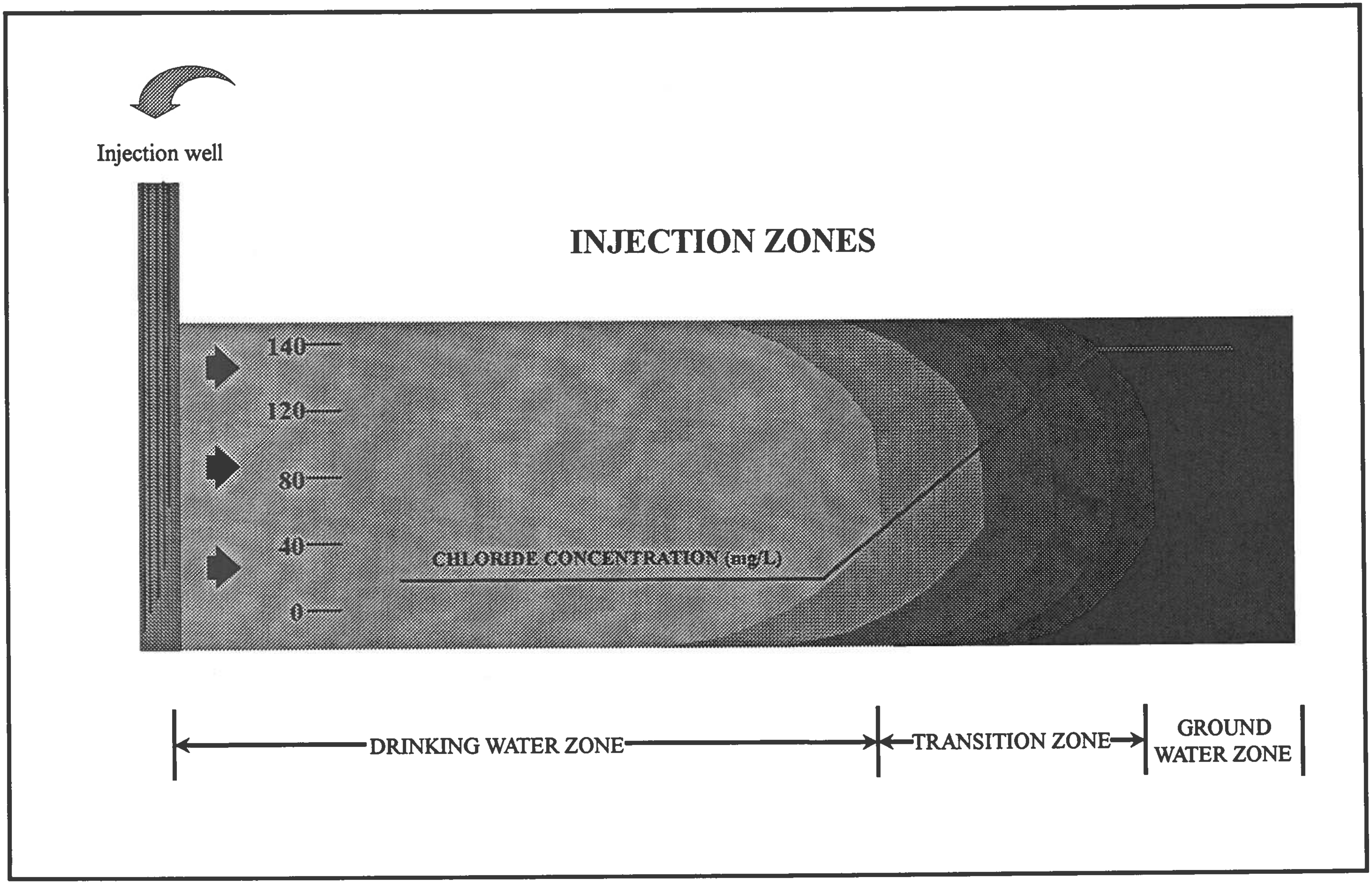


Figure 31. Schematic representation of injection effects on water quality, as characterized by chloride concentration.

water to become the *final water*. Moreover, the chloride ion, because of its nonreactive character in the system, provided an accurate estimate of the mixing between end members.

Table 5. Stages and processes of the geochemical system

MODELING STAGE	AEROBIC CONDITIONS	ANAEROBIC CONDITIONS	PHYSICAL STAGE
1	Chemical reaction		Early storage
2		Chemical reaction/ mixing	Late storage
3		Mixing/ chemical reaction	Recovery

EARLY STORAGE STAGE

Water samples 18t1 and 33t1, collected during injection, were designated *initial water 1* and *initial water 2*, respectively. Sample 33t1 was the last sample collected during injection and was representative of the average composition of the injected water. Sample 18t1 was taken in the middle of the injection period and represented a water composition somewhat different from that of the sample 33t1. The difference might be due to seasonal variations of the source water. A representative sample from the native ground water was selected and labeled *initial water 3*. Sample 42w1, collected just past the storage period midpoint, was selected as *final water* and, therefore, was the sample simulated to examine reactions occurring during early storage.

Although the *final water* (42w1) resembled the *initial water 1* and *2* (18t1 and 33t1), differences in ionic concentration were readily observed (Table 4). Calcium ion concentration, for example, increased by 138 percent, carbon ion by 96 percent, and sulfate ion by 30 percent. Dissolved oxygen, in contrast, decreased by more than 97 percent.

According to transport simulations (Petkewich, 1992), oxygen concentrations decrease sharply within a short distance away from the well. Therefore, aerobic oxidation were assumed to be confined to an area in the immediate vicinity of the well screens. In this aerobic environment, pyrite and organic-matter oxidation probably predominated. Once the system became anoxic, organic-matter oxidation coupled with nitrate and sulfate reduction probably prevailed. Calcite dissolution was also considered a potentially important process during this first stage.

Table 6 summarizes the modeling results for the early storage period. The table has been divided into two sections; the upper section (rows A through F) provides information on the chemical composition of the water samples, and the lower section (rows G through R) lists the chemical processes modeled. Rows A, B, and C provide the ionic concentration in millimoles per kilogram (mmol/kg) of each constituent in *initial water 1*, *initial water 2*, and *initial water 3*, respectively. Row D shows the chemical composition that would be expected in a conservative mixture of the *initial water* in proportions indicated in the PERCENT column. Row E shows the chemical composition of the *final water*, which is the sample being modeled. Comparison of row E *(final water)* with row D *(simple mixture)* shows that the latter sample accounts for most but not all of the *final water* chemical composition. The ionic difference between these two samples, therefore, has to be the result of in situ geochemical processes. This difference, which is given in row F, is what the model tries to explain by invoking several plausible chemical reactions.

The model results are given in the lower part of the table under the heading GEOCHEMICAL MODELING. Rows G through Q list the chemical processes involved in the modeling. Row R is the total contribution from chemical processes for each ion. Positive values indicate that geochemical processes have caused a net

Table 6. Geochemical modeling of water sample 42W1 (early storage period)

	SAMPLE	PERCENT	CONCENTRATION (mmol/kg)											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
A	**Initial water 1 (18t1)**	30.73	0.9242	0.3885	0.8309	0.0942	0.0522	0.1210	0.3703	0.0012	0.0212	0.0579	0.0003	0.0628
B	**Initial water 2 (33t1)**	68.86	1.0870	0.2995	1.2181	0.0576	0.0767	0.0849	0.4269	0.0007	0.0237	0.0474	0.0003	0.1007
C	**Initial water 3 (Black Creek)**	0.41	11.0326	0.0610	14.1533	0.0667	0.1536	0.2250	0.0219	0.0004	0.0019	0.2213	0.0019	0.0004
D	**Simple mixture (initial water 1, 2, and 3)**		1.0777	0.3259	1.1521	0.0689	0.0695	0.0966	0.4078	0.0009	0.0228	0.0513	0.0003	0.0886
E	**Final water (42w1)**		2.1110	0.7736	1.3488	0.0946	0.0640	0.0899	0.5310	0.0104	0.0019	0.0684	0.0013	0.0014
F	MASS DIFFERENCE (ROW E - ROW D)		**1.0333**	**0.4477**	**0.1967**	**0.0257**	**-0.0055**	**-0.0067**	**0.1232**	**0.0095**	**-0.0209**	**0.0171**	**0.0010**	**-0.0872**

	PROCESS	MASS TRANSFER (mmol/kg)	GEOCHEMICAL MODELING											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
G	**CO_2(g), bacteriologic -- ingassing**	0.51841	0.5184											
H	**Calcite, $Ca_{.95}Mg_{.05}CO_3$ -- dissolution**	0.51496	0.5150	0.4892		0.0257								
I	**Ca/Na exchange**	0.04298		-0.0430	0.0860									
J	**Fe/Na exchange**	0.05439			0.1088					-0.0544				
K	**F/OH exchange**	0.01711										0.0171		
L	**Glauconite, $K_2Fe_2Al_6(Si_4O_{10})_3(OH)_{12}$) -- dissolution**	0.00167					0.00334	0.02004		0.00334	0.01002			
M	**K-mica, ($KAl_3Si_3O_{10}(OH)_2$) -- precipitation**	-0.01106					-0.0111	-0.0332			-0.0332			
N	**Pyrite, FeS_2 -- oxidation**	0.06160							0.1232	0.0616				
O	**Hydroxyapatite, $Ca_5(PO_4)_3OH$ -- dissolution**	0.00031		0.0016									0.0009	
P	**K-spar, $KAlSi_3O_8$ -- dissolution**	0.00217					0.0022	0.0065			0.0022			
Q	**Nitrogen (g) -- degassing**	-0.04361												-0.0872
R	MASS ADDED OR REMOVED (ROWS G TO Q)		**1.0334**	**0.4478**	**0.1947**	**0.0257**	**-0.0056**	**-0.0066**	**0.1232**	**0.0106**	**-0.0210**	**0.0171**	**0.0009**	**-0.0872**
S	NET MASS BALANCE (ROW F - ROW R)		-0.0001	-0.0001	0.0019	-0.0000	0.0001	-0.0000	-0.0000	-0.0010	0.0000	-0.0000	0.0001	-0.0000

(-) minus--> removed from solution

(+) plus --> added to solution

increase in concentration in the solution. Negative values signify a net removal from solution. Row S provides an indication of accuracy of the model in predicting the mass transfers.

Under the heading PROCESS, the table lists a phase name, the chemical composition of the phase involved in the reaction, and the mass transfer that resulted from the reaction. To the right of this section, the table shows how much the chemical reactions have contributed to the concentration of each ion affected by this process. The addition (+) or removal (-) of mass is given in millimoles per kilogram (mmol/kg).

The objective of the model in the EARLY STORAGE stage was to explain the large increase of total carbon concentration (CO_2) and the decrease of dissolved oxygen. Both were assumed to be the result of the oxidation of pyrite, aerobic and anaerobic oxidation of organic matter, and calcite dissolution. The model (Table 6) shows that the oxidation of 0.0616 mmol of pyrite (row N) consumed 0.2156 mmol of dissolved oxygen and produced 0.1232 mmol of sulfate ion and 0.0616 mmol of ferrous iron per kilogram of solution (Eq. 2). The model thus required more dissolved oxygen than was available in the mixture, which was 0.205 mmol/kg. The difference, 0.0106 mmol/kg, might indicate that additional oxygen became available to the system, perhaps due to air entrainment.

$$FeS_2+\frac{7}{2}O_2+H_2O\leftrightarrow Fe^{2+}+2SO_4^{2-}+2H^+ \qquad (2)$$

Sodium was assumed to be produced by desorption from clay as a result of exchange with ferrous iron produced by reaction (2) (row J). Another process assumed to be important was the oxidation of organic matter, a microbial catalyzed reaction that requires oxidizing agents. Bacteria, in these processes, display a selective preference for agents (Chapelle and others, 1993). These agents, in order of preference, are: dissolved oxygen (O_2), nitrate (NO_3^-), manganese oxyhydroxide (MnO_2), ferric hydroxide ($Fe(OH)_3$), and sulfate ion (SO_4^{2-}). If a system becomes strongly reducing, organic matter can even act as its own oxidizing agent, producing carbon dioxide (CO_2) and methane (CH_4) (Freeze and Cherry, 1979).

Since all oxygen was removed, the system became anoxic, and oxidation of organic matter should have proceeded through the reduction of available agents. The anaerobic oxidation of organic matter was not directly modeled; the resulting inorganic carbon phase (CO_2) was simply assumed to be present as a consequence of this process. This assumption is supported by McMahon and Chapelle (1991) who suggested that the inorganic carbon is produced in situ by anaerobic processes such as fermentation. They concluded that, in Atlantic Coastal Plain Formations, anaerobic respiration of organic matter in confining clay beds contributes nearly one-third of the total carbon present in aquifers.

The proposed nitrate model assumes removal via reduction to nitrogen gas, using organic matter as an oxidant (row Q). Organic matter, for the purpose of modeling, has been represented by CH_2O. According to this reaction (Eq. 3), 0.042 mmol of nitrogen gas was generated.

$$CH_2O+\frac{4}{5}NO_3^-\longrightarrow\frac{2}{5}N_{2\,(g)}$$
$$+HCO_3^-+\frac{1}{5}H^+\frac{2}{5}H_2O \qquad (3)$$

For calcium, important processes present at this stage assumed calcite dissolution and exchange with sodium (row H and row I). Calcium provided from calcite dissolution was assumed to exchange with sodium adsorbed on marine clay (Eq. 4). Although the actual calcium carbonate phase undergoing dissolution is aragonite and not calcite, it is referred to here as calcite. Both phases have the same chemical composition and their crystallographic differences are irrelevant in mass-balance calculations.

$$CaCO_3+Na_2-CLAY+H_2O+CO_2$$
$$\leftrightarrow 2Na^++Ca-CLAY+2HCO_3^- \qquad (4)$$

Other, less important, processes, as indicated by their small mass transfers, were hydroxyapatite dissolution (row O) and fluoride-hydroxyl exchange (row K), glauconite dissolution (L), potassium-mica precipitation (row M), and potassium-feldspar dissolution (row P).

Table 7. Geochemical modeling of water sample 48W1 (late storage period)

	SAMPLE	PERCENT	CONCENTRATION (mmol/kg)											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
A	**Initial water 1 (18t1)**	0.00	0.9242	0.3885	0.8309	0.0942	0.0522	0.1210	0.3703	0.0012	0.0212	0.0579	0.0003	0.0628
B	**Initial water 2 (33t1)**	98.37	1.0870	0.2995	1.2181	0.0576	0.0767	0.0849	0.4269	0.0007	0.0237	0.0474	0.0003	0.1007
C	**Initial water 3 (Black Creek)**	1.63	11.0326	0.0610	14.1533	0.0667	0.1536	0.2250	0.0219	0.0004	0.0019	0.2213	0.0019	0.0004
D	**Simple mixture (Initial water 1, 2, and 3)**		1.2492	0.2956	1.4290	0.0578	0.0780	0.0872	0.4203	0.0007	0.0233	0.0502	0.0003	0.0991
E	**Final water (48w1)**		2.6424	0.5815	2.5410	0.0683	0.0711	0.0729	0.4894	0.0020	0.0020	0.0737	0.0042	0.0004
F	**MASS DIFFERENCE (ROW E - ROW D)**		**1.3932**	**0.2859**	**1.1120**	**0.0105**	**-0.0069**	**-0.0143**	**0.0691**	**0.0013**	**-0.0213**	**0.0235**	**0.0039**	**-0.0987**

	PROCESS	MASS TRANSFER (mmol/kg)	GEOCHEMICAL MODELING											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
G	CO2(g), bacteriologic -- ingassing	0.5828	0.5828											
H	Calcite, Ca.95Mg.05CO3 -- dissolution	0.8109	0.8109	0.7704		0.0405								
I	Ca/Na exchange	0.4909		-0.4909	0.9818									
J	Mg/Na exchange	0.0300			0.0600	-0.0300								
K	F/OH exchange	0.0235										0.0235		
L	Fe/Na exchange	0.0354			0.0707					-0.0354				
M	Glauconite, K2Fe2Al6(Si4O10)3(OH)12 -- dissolution	0.0011					0.0021	0.0127		0.0021	0.0064			
N	Gibbsite, Al(OH)3 -- precipitation	-0.0008									-0.0024			
O	Pyrite, FeS2 -- oxidation	0.0346							0.0691	0.0346				
P	K-mica, KAlSi3O10(OH)2 -- precipitation	-0.0090					-0.0090	-0.0270			-0.0270			
Q	Hydroxyapatite, Ca5(PO4)3OH -- dissolution	0.0013												
R	Nitrogen (g)	-0.0493												-0.0987
S	**MASS ADDED OR REMOVED (ROWS G TO R)**		**1.3937**	**0.2795**	**1.1125**	**0.0106**	**-0.0069**	**-0.0143**	**0.0691**	**0.0013**	**-0.0230**	**0.0235**	**0.0000**	**-0.0987**
T	NET MASS BALANCE (ROW F - ROW S)		-0.0004	0.0064	-0.0005	-0.0000	0.0000	-0.0000	-0.0000	-0.0000	0.0016	-0.0000	0.0039	0.0000

(-) minus--> removed from solution

(+) plus --> added to solution

In summary, the model suggested that 50 percent of the carbon increase was due to oxidation of organic matter and 50 percent to calcite dissolution. Excess calcium was produced by calcite dissolution and most was exchanged for sodium from clay. Pyrite oxidation increased the sulfate concentration in solution. The anaerobic oxidation of organic matter was not directly modeled; instead a "$CO_{2(g)}$ phase" was assumed to be present in the system to provide the inorganic carbon necessary to balance the system.

LATE STORAGE STAGE

In this stage, the *final water* was sample 48w1. The *initial water* samples remained the same as in the previous stage (18t1 and 33t1). Sample 48w1 was chosen as the *final water* because it was the last sample collected during the *storage period,* which was the period simulated.

Results of the modeling are presented in Table 7. Inspection of the table reveals that total carbon and sodium concentrations increased significantly while the other constituents changed only slightly (row F). The model, consequently, was primarily developed to explain the large increases of carbon and sodium. A carbon dioxide phase (row G) was assumed to be available to the system through a pool of $CO_{2(g)}$ generated by anaerobic respiration.

According to the model, approximately 42 percent of the total inorganic carbon was provided by this pool of $CO_{2(g)}$ and the rest by calcite dissolution (row H). The excess calcium from the dissolution of calcite was exchanged with sodium (row I). Additionally, sodium was generated by exchange with magnesium (row J) and ferrous iron (row L). Glauconite dissolution provided iron, silica, aluminum, and all of the potassium needed to balance the system (row M). Excess aluminum present in the mixture and from glauconite dissolution was precipitated as gibbsite (row N) and potassium-mica (row P). The oxidation of pyrite (row O) increased the concentration of sulfate and ferrous iron ions in solution. Potassium-mica precipitation removed from solution the excess potassium, silica, and aluminum (row P). Hydroxylapatite dissolution explained most of the increase of phosphate ions in solution (row Q). The last process shown in Table 7 is the reduction of nitrate to nitrogen gas in the oxidation of organic matter (row R).

RECOVERY STAGE

This stage simulated the chemical reactions and mixing observed in samples collected during the recovery phase. Mixing appeared to be the dominant process determining the composition of the recovered water. Sample 50w8 was selected as the *final water*. This sample was collected when a volume equivalent to approximately 112 percent of the injected water was pumped. The *initial-water* samples were injected water samples 18t1 and 33t1 and the native ground water sample.

In simulating sample 50w8, it was assumed that the final water evolved from chemical reactions and mixing of the three initial water types. After a few model trials, it became evident that the chloride concentration in sample 50w8 was in error. This was confirmed when chloride concentrations of adjacent samples were reviewed. Inspection of Table 8 clearly shows that not only the chloride concentration of sample 50w8 was in error but also that of sample 50w9.

Although corrections introduced in Table 8 are arbitrary, they were chosen to provide a smoother transition between chloride values of adjacent samples. These corrections were based on interpolations from the tritium-chloride relationship shown in Figure 13(b).

Table 8. Corrected chloride for sample 50w8

SAMPLE	CHLORIDE (mg/L)	CORRECTED VALUE (mg/L)
50w5	100	
50w6	105	
50w7	108	
50w8	124	111
50w9	108	116
50w10	121	
52w1	142	

Table 9 lists chemical compositions of the samples and modeling results for the recovery phase. Contrary to previous stages, the inorganic carbon and sodium concentrations decreased. Calcium in solution increased as before. The rest of the constituents appear to have been less reactive and to have played only a small role in explaining the final composition of the simulated sample.

The model developed for this stage considered a $CO_{2(g)}$ phase; however, to achieve mass balance it had to act as an inorganic carbon sink (row G). In previous models, the inorganic carbon phase was always a source.

To explain the sodium decrease, the model postulated a sodium-calcium exchange that removed sodium from solution and added it to the clay (row I). This reverse-exchange process increased the calcium in solution that was removed by calcite precipitation (row H). Sodium exchanged with magnesium (row J) and ferrous iron (row L). This process added magnesium in solution and removed iron from solution.

Another predicted process was the exchange of fluoride with hydroxyl (row K), which added fluoride to solution and removed hydroxyl ions. Silica precipitated to remove the excess dissolved silica from solution (row M). Pyrite oxidation added to solution all of the sulfate needed to maintain mass balance as well as all of the iron that exchanged with sodium (row O). Potassium-feldspar dissolution added excess potassium and silica in solution (row P), which precipitated illite, thus removing magnesium, potassium, silica, and aluminum from solution (row N). The dissolution of hydroxyapatite added calcium and phosphate ions to the solution (row Q). Finally, nitrogen gas was produced from the oxidation of organic matter (nitrate reduction, row R).

In summary, degassing of $CO_{2(g)}$, calcite precipitation, and sodium-calcium exchange were the predominate geochemical processes. Nearly 79 percent of the inorganic carbon was removed by the $CO_{2(g)}$ sink and 21 percent by the precipitation of calcite. Most of the chemical makeup of the *final water* sample, however, was explained by the *simple mixture*. The sample was a mixture of 24 percent injected water and 76 percent ground water. The calcite precipitation and reverse exchange of calcium for sodium processes invoked by this model may not be entirely feasible in this environment. The apparent over-saturation of calcite may be an artifact of degassing. If carbon dioxide degassed during sample collection, the pH could have decreased, causing the speciation model to predict an oversaturation of calcite. This problem requires further investigation.

It is interesting to note that the sodium concentration in the *simple mixture* was larger than that found in the *final water* sample, which is somewhat unexpected. Assuming that the sodium came from cation exchange only, it would be reasonable to anticipate that the maximum sodium concentration would be similar to that found in the ground water, but no higher. This would indicate that the corrected chloride concentration for sample 50w8 was still too large.

ECONOMIC CONSIDERATIONS AND BENEFITS

The analysis selected to evaluate the economic benefits of an ASR system compares the cost of an injection well with costs of conventional facilities. The facilities used in the analysis were those that provide either additional storage (storage tank) or additional flow (expansion of treatment plant). Costs were computed as annual cost per 1,000 gallons of water produced and were amortized over 30 years at an interest rate of 7 percent. Most of the data used in the analysis were obtained from the City of Myrtle Beach and the Grand Strand Water and Sewer Authority, the two major water suppliers in Horry County, S.C.

CONVENTIONAL ALTERNATIVES

Storage tanks. Peak-daily and emergency water demands might be satisfied by increasing the storage capacity of the distribution system. In a public supply system with varying seasonal demand, surplus water could be accumulated in storage tanks and later used to augment daily pumpage during periods of high water demand. The most common types of storage facilities are elevated and ground-level storage tanks.

Treatment plant expansion. Another alternative for increasing the production capacity of a plant is to expand

Table 9. Geochemical modeling of water sample 50W8 (recovery period)

	SAMPLE	PERCENT	CONCENTRATION (mmol/kg)											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
A	**Initial water 1 (18t1)**	0.00	0.9242	0.3885	0.8309	0.0942	0.0522	0.1210	0.3703	0.0012	0.0212	0.0579	0.0003	0.0628
B	**Initial water 2 (33t1)**	24.41	1.0870	0.2995	1.2181	0.0576	0.0767	0.0849	0.4269	0.0007	0.0237	0.0474	0.0003	0.1007
C	**Initial water 3 (Black Creek)**	75.59	11.0326	0.0610	14.1533	0.0667	0.1536	0.2250	0.0219	0.0004	0.0019	0.2213	0.0019	0.0004
D	**Simple mixture (initial water 1, 2, and 3)**		8.6049	0.1192	10.9958	0.0645	0.1348	0.1908	0.1208	0.0005	0.0072	0.1789	0.0015	0.0249
E	**Final water (50w8)**		8.0810	0.2440	10.5798	0.0572	0.1608	0.1279	0.1563	0.0003	0.0035	0.2424	0.0042	0.0004
F	MASS DIFFERENCE (ROW E - ROW D)		**-0.5239**	**0.1248**	**-0.4160**	**-0.0073**	**0.0260**	**-0.0629**	**0.0355**	**-0.0002**	**-0.0037**	**0.0635**	**0.0027**	**-0.0245**

	PROCESS	MASS TRANSFER (mmol/kg)	GEOCHEMICAL MODELING											
			Carbon	Calcium	Sodium	Magnesium	Potassium	Silica	Sulfate	Iron	Aluminum	Fluoride	Phosphate	Nitrate
G	CO2(g), bacteriologic -- degassing	-0.4150	-0.4150											
H	Calcite, Ca.95Mg.05CO3 -- precipitation	-0.1086	-0.1086	-0.1032		-0.0054								
I	Ca/Na exchange	-0.2235		0.2235	-0.4470									
J	Mg/Na exchange	-0.0025			-0.0051	0.0025								
K	F/OH exchange	0.0635										0.0635		
L	Fe/Na exchange	0.0179			0.0358					-0.0179				
M	Silica (a), SiO2 -- precipitation	-0.1109						-0.1109						
N	Illite, KO.6MgO.25Al2.3Si3.5O10(OH)2 -- precipitation	-0.0174				-0.0044	-0.0105	-0.0610			-0.0401			
O	Pyrite, FeS2 -- oxidation	0.0178							0.0355	0.0178				
P	K-spar, KAlSi3O8 -- dissolution	0.0364					0.0364	0.1091			0.0364			
Q	Hydroxyapatite, Ca5(PO4)3OH -- dissolution	0.0009		0.0045									0.0027	
R	Nitrogen (g) -- degassing	-0.0122												-0.0245
S	MASS ADDED OR REMOVED (ROWS G TO R)		**-0.5236**	**0.1248**	**-0.4163**	**-0.0073**	**0.0259**	**-0.0628**	**0.0355**	**-0.0001**	**-0.0038**	**0.0635**	**0.0027**	**-0.0245**
T	NET MASS BALANCE (ROW F - ROW S)		-0.0003	-0.0001	0.0002	-0.0000	0.0001	-0.0001	-0.0000	-0.0000	0.0000	0.0001	-0.0000	-0.0000

(-) minus--> removed from solution

(+) plus --> added to solution

chloride concentration was corrected

the treatment capacity. This would be done by enlarging plant structures and treatment facilities. Plant treatment capacity is designed for a 30-year water demand, that for economic reasons, is implemented in a series of steps throughout the life of the facility. At Myrtle Beach, for example, the treatment plant completed in 1988 had a capacity of 20 mgd. It was expanded in 1990 to a capacity of 25 mgd, and more recently to 29.5 mgd. The expansion was necessary in order to supply the city of North Myrtle Beach with potable water.

COST ANALYSIS

Storage tank alternative. According to information provided by the city of Myrtle Beach in 1992, the construction cost of a 2-million gallon ground tank was $ 440,000. The construction of an elevated 0.5-million gallon tank was $ 610,000. The annual costs for the ground tank and elevated tank per 1,000 gallons were $31 and $161 dollars, respectively (Table 10). The costs, especially for the elevated tank, are high because the tanks serve other needs besides storage. For example, the elevated tank aids in regulating the pressure of main and secondary lines. This additional benefit, essential for a public supply system, was not factored into the analysis because it does not provide storage benefits to this study. Owing to the unrealistically high unit costs of these alternatives, they are not being considered in the study. They are mentioned in this report in order to include all alternatives.

Treatment-plant expansion. On the basis of information provided by the two largest utilities in Horry County, the annual cost per 1,000 gallons for treatment plant expansion was estimated to be $ 1.13, of which $ 0.59 and $ 0.54 were capital and operation-and-maintenance costs, respectively. The amortization period was, again, 30 years and the interest rate 7 percent.

Aquifer storage and recovery (ASR) well. Inasmuch as Horry County has a substantial number of Black Creek wells that could be converted to ASR wells, the cost analysis also considered the upgrade of existing wells in addition to the construction of new wells (Table 11). Cost estimates were made for a single, dual-purpose, Black Creek well. These figures reflect conditions that were encountered at the Myrtle Beach test site.

Table 10. Comparison of elevated and ground storage tank costs

COST	STORAGE TANK ELEVATED (0.5 mil. gal.)	 GROUND (2.0 mil. gal.)
Capital	$610,000	$440,000
Annual	$80,000	$65,000
Annual per 1,000 gallons	$161	$32

Table 11 provides a breakdown of items and costs for a single Black Creek well capable of delivering 36 million gallons of water per cycle. The analysis provided for the purchase of monitoring equipment, minor repairs, and the installation of a well pressure seal. In computing the total annual cost of these alternatives, two important assumptions were made. First, the unit cost for operating and maintaining an ASR well was considered fixed, although it will decrease as the number of wells increases (economy of scale). Second, the volume recovered was assumed to be equal to the volume injected (100-percent recovery).

Costs for other recovery efficiencies are shown in Figure 32. It is important to emphasize that the unit cost is inversely related to the production capacity of a well; the more it produces, the lower the unit cost. Therefore, wells delivering twice as much water would have one-half the unit cost.

Annual-cost comparison. Figure 33 shows that the cost of both ASR alternatives is less than that of expanding a surface-water treatment plant. The difference lies in the annual capital cost. Whereas the annual capital cost per 1,000 gallons is $ 0.06 and $ 0.55 for an upgrade and a new

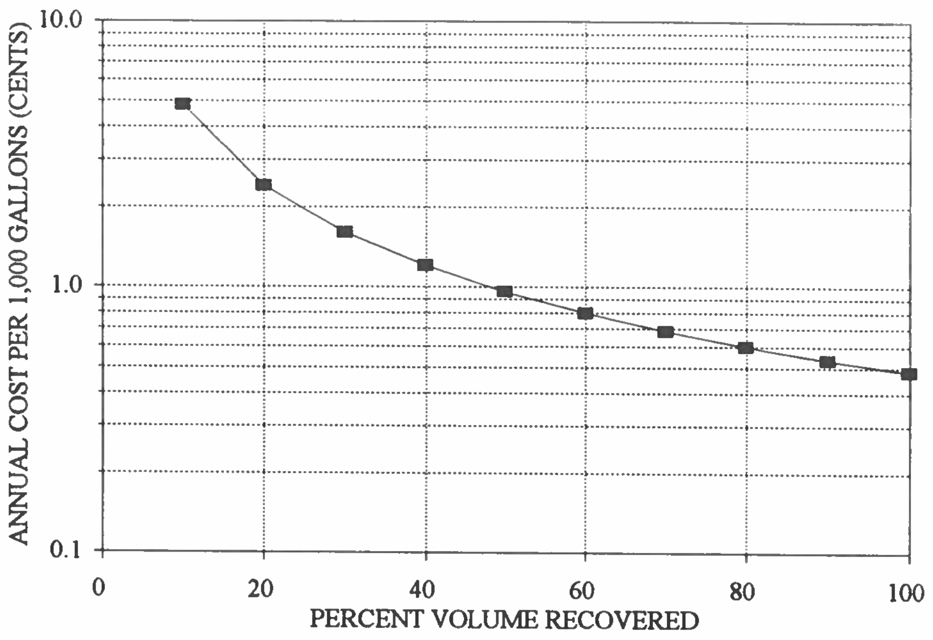


Figure 32. Annual cost of injecting, recovering, and disinfecting 36 million gallons of treated surface water, using a dual-purpose Black Creek well.

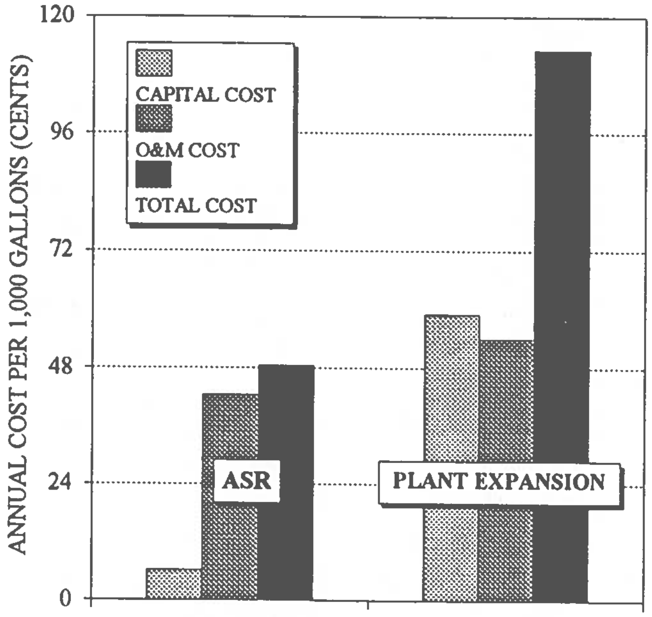


Figure 33. Annual-cost comparison for an ASR system and a treatment plant expansion alternative.

Table 11. Annual costs (dollars) for an ASR alternative producing 36 million gallons of water

COST	DESCRIPTION	EXISTING WELL	NEW WELL
Capital	Well upgrade/construction	$15,000	$250,000
	Equipment	$10,000	
	Contingency (10% capital)	$2,500	
	Total	$27,500	$250,000
	Debt service	$2,200	$20,000
Annual	Operation and maintenance		
	Power	$6,600	$6,600
	Parts and chemical	$4,200	$4,200
	Personnel	$2,800	$2,800
	Overhead (10% OM)	$1,400	$1,400
	Total	$17,200	$35,000
Unit annual (1,000 gallons)	Capital	$0.06	$0.55
	Operation and maintenance	$0.42	$0.42
	Total unit annual	$0.48	$0.97

well, respectively, it is $ 0.59 for the expansion. Even if an upgraded ASR well had a recovery of less than 50 percent, it would still be the most cost-effective alternative.

ADDITIONAL ECONOMIC CONSIDERATIONS

ASR projects could fulfill several additional objectives while still providing increased storage capacity (see page 7). The following example was prepared so that different options could be explored and evaluated. The example focused on designing and evaluating the operation schedule of an ASR-treatment-plant system for the city of Myrtle Beach. The operating schedule was computed by using an optimization technique that (a) minimized the cost of operation, (b) selected the optimum number of wells, and (c) provided an optimum schedule of injection, recovery, and plant production.

Data used in the example were obtained from the city's monthly surface water treatment reports for 1991 and 1992. Chemical costs were obtained from suppliers in the region. Operation-and-maintenance costs were obtained from the previous unit-cost analysis. Therefore, expenses computed here do not represent the City's real expenses or the treatment plant operating budget.

Statement of problem. Design an optimum production schedule for an ASR-and-treatment-plant (combined) facility that would provide the same or greater production capacity as a treatment plant (with or without an expansion) while operating at the same or lower cost.

Methodology. On the basis of the chemical-dosage information, monthly treatment costs were computed for the treatment plant. These treatment costs were added to those of capital and operation-and-maintenance to estimate total monthly costs. For the ASR-and-treatment-plant alternative, similar steps were taken, but with the difference being that for the ASR-and-treatment-plant option an optimization routine was used to minimize the production cost while satisfying the water demand. This procedure was repeated for different scenarios where water demand progressively increased by 10 percent.

In the analysis, the following assumptions were made:

- The maximum treatment capacity of the plant was fixed at 29.5 mgd;
- A single expansion was introduced to double the existing capacity of the plant;
- 30 wells were considered available to the system. These were former public supply wells that had been upgraded to ASR standards;
- The pumping capacity of each well is estimated at 0.648 mgd and the injection capacity at 0.214 mgd —⅓ the pumping capacity;
- In predicting future water demands, past trends and seasonalities were expected to continue.

Solution. A study of the monthly chemical dosage for the treatment plant revealed that there were months when water was easier to treat, and therefore less expensive to process (Table 12). This finding was used to develop a schedule that utilized more of the less-expensive water and less of the more-expensive water. Results of the study are presented in Table 13.

Table 12. Monthly cost per 1,000 gallons for chemical treatment

MONTH	COST
Jan	$1.13
Feb	$1.11
Mar	$1.09
Apr	$1.13
May	$1.15
Jun	$1.20
Jul	$1.17
Aug	$1.19
Sep	$1.23
Oct	$1.19
Nov	$1.15
Dec	$1.17

Table 13 is divided into two sections; the left side shows the operation of the plant as normally scheduled, and the right side shows the optimum operation of the plant with an ASR system. Beginning at the left side of the table: The first column lists the months; the second column lists the average monthly demand in million gallons per day for 1992; the third column shows the total monthly demand in millions of gallons for 1992; the fourth column is a projected monthly demand and was calculated by multiplying the monthly demands by a demand factor; and the fifth column is the cost for treatment, in dollars, obtained by multiplying the projected demand by the monthly cost per 1,000 gallons from Table 12. On the right side of the table: The first column is the optimum daily production of the plant in millions of gallons; the second column is the optimum monthly production in millions of gallons; the third column is the cost for treatment, obtained by multiplying the monthly production by the monthly cost; the fourth column shows the optimum number of ASR wells required to operate; the shaded area shows the months when recovery takes place; and the fifth column is the total monthly cost of the ASR system. Shown below the table are the treatment capacity and average demand for both alternatives, the total cost of each alternative, the cost difference between alternatives, and the demand factor. The cost difference is positive when the ASR-treatment-plant system is less than the treatment plant alone. Although both alternatives produce the same volume of water per year, in some scenarios the plant may not need to operate for several months at a time in the ASR-treatment-plant alternative. If this is an undesirable condition, the optimization routine could be modified to restrict the minimum operating flow of the plant to a value greater than zero.

Table 13 shows the results of the optimization model for a scenario where the water demand is 2.4 times the demand in 1992. According to these results, the plant, if operating without any ASR wells, would have to increase its capacity by 16.6 mgd to meet the demand for this scenario (10 billion gallons per year). Interestingly, the ASR-treatment-plant alternative requires 5.6 million dollars less to operate than the expanded plant.

TABLE 13. Optimum production of a plant-and-ASR alternative

MONTH	PLANT (PRESENT OPERATION)				PLANT AND ASR (OPTIMUM OPERATION)				
	PLANT PRODUCTION (MILLIONS OF GALLONS)			TREATMENT COST DOLLARS	PLANT PRODUCTION (MILLIONS OF GALLONS)		TREATMENT COST DOLLARS	A S R	
	AVG. DAILY	MONTHLY	PROJECTED		AVG. DAILY	MONTHLY		NUMBER OF WELLS	COST DOLLARS
JAN	18.5	238	572	129,746	24.9	771	174,919	30	43,000
FEB	21.3	249	597	125,887	27.7	777	163,851	30	43,000
MAR	23.9	309	740	140,992	29.5	914	174,145	26	37,581
APR	28.2	353	847	197,987	29.5	885	206,879	6	8,485
MAY	28.9	374	896	228,194	29.5	915	232,801	3	3,907
JUN	34.4	430	1031	318,147	29.5	885	273,147	7	10,750
JUL	46.1	595	1429	390,793	29.5	915	250,140	26	36,691
AUG	41.9	542	1300	372,729	29.5	914	262,136	19	27,529
SEP	27.4	342	821	272,542	22.2	666	221,151	8	11,408
OCT	22.7	293	704	207,122	29.1	903	265,737	30	43,000
NOV	19.1	239	573	141,182	25.5	766	188,691	30	43,000
DEC	15.9	205	492	131,564	22.3	692	184,782	30	43,000
		TOTAL	10,002	2,656,886	TOTAL	10,002	2,598,380		351,351

MAX. TREATMENT CAPACITY (MGD)	46.1
AVERAGE DEMAND (MGD)	27.4

MAX. TREATMENT CAPACITY (MGD)	29.5
AVERAGE DEMAND (MGD)	27.4

TREATMENT PLANT COST	$17,560,223
TREATMENT-PLANT-AND-ASR COST	$11,951,747
DIFFERENCE	$5,608,476
DEMAND FACTOR	2.4

Figure 34 shows the results of these simulations. The simulations began with a demand of 19.2 mgd, reflecting conditions in 1992, and progressively increased to 46.1 mgd, the maximum demand that the ASR-treatment-plant system can satisfy. Owing to simplifications introduced in the analysis, the cost is proportional to the demand; no economy-of-scale effects were considered. The cost of the ASR-treatment-plant alternative is slightly larger than the cost of the plant alone during the early part of the operational period considered. This means that until the maximum capacity of the plant (29.5 mgd) is exceeded, the plant alone can operate at a lower cost. If the demand in the system increases past the maximum capacity of the system, the cost of the ASR-treatment-plant alternative is significantly less than the cost of the plant and expansion.

SUMMARY AND CONCLUSIONS

Information collected during field tests at the Myrtle Beach Aquifer Storage and Recovery project has proven the feasibility of ASR systems for sediments of the Atlantic Coastal Plain. Moreover, the cost analysis has shown that an ASR system is an inexpensive alternative for augmenting daily flows in a public supply system. Consequently, ASR systems could be an effective and innovative management alternative. They could help utilities and municipalities control rising water costs by reducing initial capital investment in treatment plant expansion, by improving the operation of existing treatment plants, and by providing additional benefits from investments already made in existing wells.

On the basis of the geochemical and hydraulic analyses, aquifers of the Black Creek Formation are the most suitable units for storing treated surface water in Horry County. After more than 2 years of testing and over 10 injection and recovery cycles, well efficiency and aquifer properties have not been adversely affected by injection. In general, the recovered water retains most of the injected-water characteristics, although the water is subjected to chemical reactions that modify its chemical makeup. Some of these modifications are desirable, such as the increase in pH and alkalinity, decrease in trihalomethanes and calcium concentrations, and more stable temperature. Some of these changes are of concern, such as the increases in concentration of sodium, fluoride, and iron. The water quality, however, is expected to improve gradually with each cycle of injection and recovery as pyrite, calcite, apatite, and exchangeable sodium are removed from the aquifer.

The three most important geochemical processes identified by modeling are (a) pyrite oxidation, (b) calcite dissolution, and (c) Ca-Na exchange.

During recovery, degassing persisted for most of the test. Carbon dioxide gas appears to be released to the atmosphere when water is brought to the surface. If this condition persists, water would have to be aerated before it is pumped into a distribution system.

For the conditions at Myrtle Beach, the annual costs in dollars per 1,000 gallons, are: $ 0.48 for an ASR system (upgraded well) versus $1.13 for the expansion and operation of a treatment plant. These figures include capital and operation-and-maintenance costs. While the operation-and-maintenance costs for both alternatives are similar, the capital costs are not. The ASR alternative has a capital cost of $ 0.06 per 1,000 gallons, and the treatment plant expansion alternative has a cost of $ 0.59 per 1,000 gallons.

For systems where optimization of plant-operations is of interest, ASR wells have the most beneficial application when water demand exceeds the treatment plant capacity. The plant alone (without ASR wells) normally would operate more efficiently if the water demand is less than the treatment plant capacity. If the demand exceeds the plant capacity, a combination of ASR wells and plant is preferred. This alternative is better because the ASR-treatment-plant alternative is substantially less expensive than the expansion of the treatment plant.

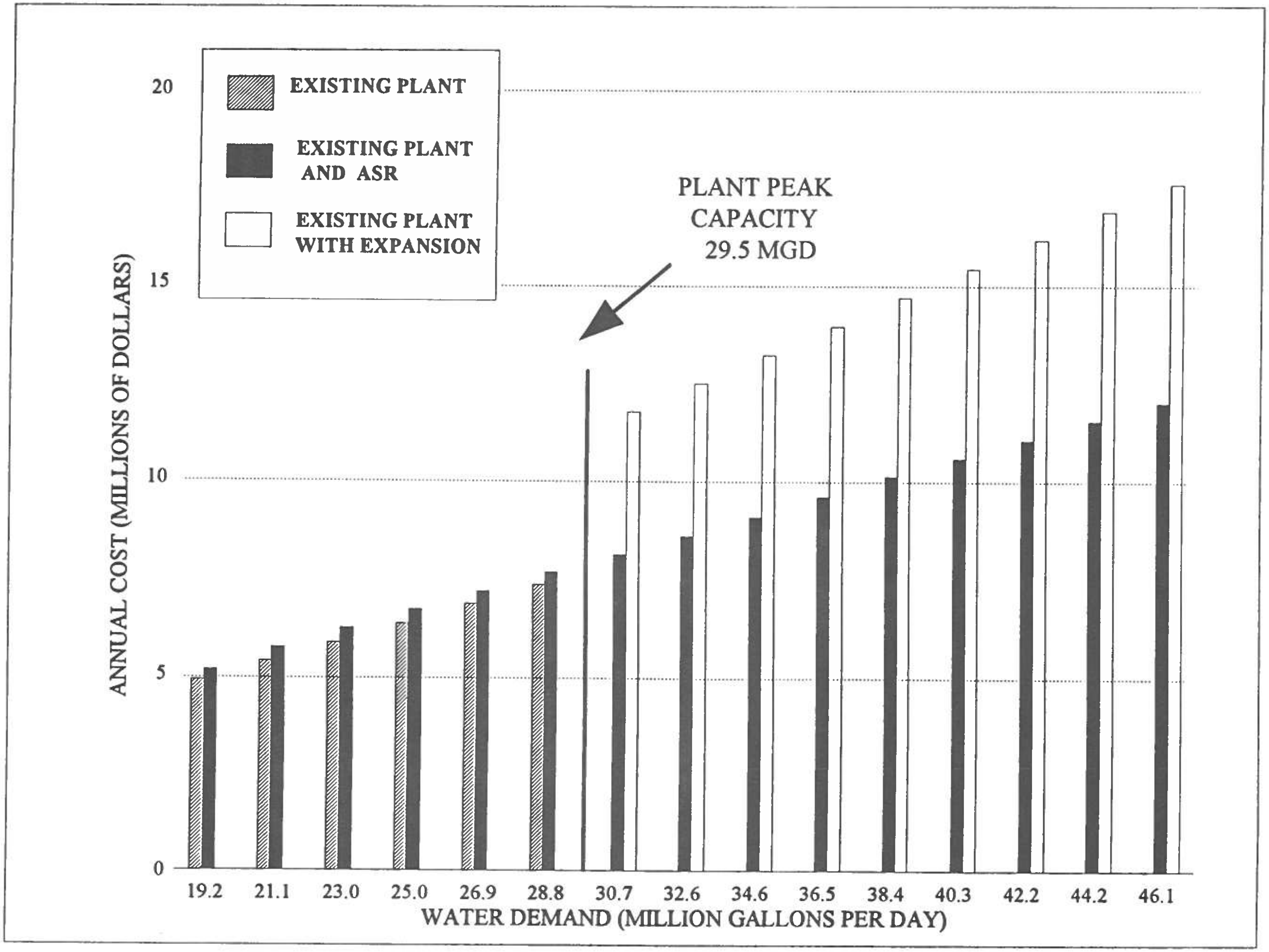


Figure 34. Comparison of the costs of implementing an ASR and a treatment-plant alternative and treatment-plant expansion.

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