Geochemistry and Microbiology of Iron-Related Well-Screen Encrustation and Aquifer Biofouling in Suffolk County, Long Island, New York

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CONVERSION FACTORS, ABBREVIATIONS, AND VERTICAL DATUM

Multiply	By	To Obtain
	Length	
inch (in.)	2.54	centimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
	Area	
square mile (mi^2)	2.590	square kilometer
	Hydraulic conductivity	
foot per day (ft/d)	0.3048	meter per day
	Flow	
gallon per minute (gal/min)	3.7854	liter per minute
	Gradient	
foot per mile (ft/mi)	0.1894	meter per kilometer
	Volume	
million gallons (Mgal)	3,785.00	cubic meter
	Chemical Concentration	
	milligram per kilogram (mg/Kg)	
	milligram per liter (mg/L)	
	microgram per liter (µg/L)	
	Specific Conductance	
	microsiemens per centimeter at 25˚C (µS/cm)	
	Other abbreviations used in this report	
	millivolts (mv)	
	minutes (min)	
	µm (micron)	
	<, less than	
	square liter per minute•atmosphere•mole squared (l^2/min•atm•mol^2)	

Sea level: In this report, “sea level” refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)--a geodetic datum derived from a general adjustment of the first-order level nets of the United States and Canada, formerly called Sea Level Datum of 1929.

GEOCHEMISTRY AND MICROBIOLOGY OF IRON-RELATED WELL-SCREEN ENCRUSTATION AND AQUIFER BIOFOULING IN SUFFOLK COUNTY, LONG ISLAND, NEW YORK

By Donald A. Walter

Abstract

Iron-related well-screen encrustation and aquifer biofouling has decreased the specific capacity of several production wells in Suffolk County, N.Y., and has forced the Suffolk County Water Authority to adopt a costly well-reconditioning and replacement program. The specific-capacity declines are the result of the precipitation of iron oxyhydroxides and the growth of iron bacteria on the well screens and in the pore spaces of the surrounding formation. Mineralogic and chemical analyses indicate that the inorganic part of the encrusting material consists primarily of amorphous ferric hydroxide ($Fe(OH)_3$); minor components of the material include goethite (FeOOH), hematite (Fe_2O_3), and quartz (SiO_2). The weight percent of ferric hydroxide in the material ranged from 32.3 to 98.6 percent and averaged 64.3 percent. Equilibrium modeling indicated that during pumping the well waters were supersaturated with respect to goethite, hematite, magnetite, and quartz and were undersaturated with respect to ferric hydroxide. Theoretical Eh values computed for the ferrous/ferric-iron redox couple and the oxygen/water redox couple averaged 390 millivolts and 810 millivolts, respectively, indicating that the waters were in a state of redox disequilibrium. The disequilibrium condition arises from the mixing of ground water with a low dissolved-oxygen concentration with oxygenated ground water during operation of the well. The low pH of the ground water contributes to the disequilibrium condition by slowing the rate of iron oxidation after the introduction of oxygen.

Chemical and mineralogical data indicate that most of the encrusting material in the wells was deposited while the wells were shut down, probably in response to the use of treated water of higher pH to keep pump turbines wet while the wells were not in operation; the increased pH of water in the static water column increases the rate of ferrous-iron oxidation and causes the well water to become increasingly saturated with respect to ferric hydroxide. The median half-time of oxidation in samples of untreated ground water (pH 4-5) was 4.19 days, whereas the average half-time of oxidation in treated water (pH 7-8) was 11.9 minutes Equilibrium modeling indicated that treated waters generally were supersaturated with respect to ferric hydroxide, whereas untreated well waters were not.

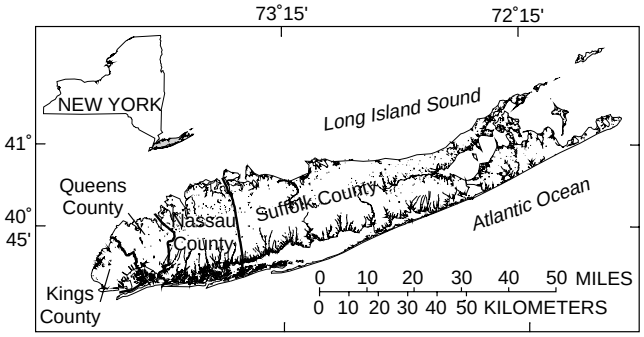
Field and laboratory data indicate that iron bacteria play an important role in the encrustation and biofouling process in Suffolk County. Filamentous iron bacteria were common in the affected wells. The most common species was *Gallionella ferruginea*, an effective biofouling agent that prefers water with low, but detectable, dissolved-oxygen concentrations and high dissolved-iron concentrations; this species was more common in biofilm samples from the Magothy aquifer than in those from the upper glacial aquifer. Iron bacteria also were found in sediment cores from several locations in the aquifer and in drilling water. Lignite could act as a carbon source for heterotrophic iron bacteria, which could accelerate the formation of iron-bacteria biofilms in

wells screened in some parts of the Magothy aquifer. Iron-bacteria biofilms alter the chemistry of well water by removing iron, manganese, and sulfate from solution and by increasing the pH. Sulfur-reducing bacteria and iron-sulfide mineral phases were observed in some samples of encrusting material, indicating that these bacteria could contribute to well-screen encrustation in some geochemical environments.

INTRODUCTION

The unconsolidated sand and gravel aquifers underlying Long Island have been designated as a sole-source aquifer system by the U.S. Environmental Protection Agency (USEPA). The Suffolk County Water Authority (SCWA), which is the primary water supplier for Suffolk County, N.Y. (fig. 1), operated about 400 public-supply wells and supplied water to about 1.2 million people in 1990. Declines in the specific capacity of several production wells, primarily along the southern shore and in the central part of the county, have compelled the SCWA to conduct a costly well-redevelopment and replacement program. The declines in specific capacity are most common in areas where ground water is discharging, such as along the coast and near streams and wetlands (Walter, 1997).

The declines in specific capacity are associated with elevated concentrations of iron and phosphorus and low dissolved-oxygen concentration in ground water and are the result of iron-related well-screen encrustation and aquifer biofouling. Iron-related specific-capacity declines have been reported in similar hydrogeologic environments (Driscoll, 1986).



Base from U.S. Geological Survey digital data 1:24,000, 1990
Universal Transverse Mercatur projection Zone 18

Figure 1. Location of Suffolk County, N.Y.

In 1990, the U.S. Geological Survey (USGS), in cooperation with the SCWA, began a 3-year study of iron-related well-screen encrustation and aquifer biofouling in the aquifers underlying Suffolk County. The purpose of the study was to investigate the occurrence of well-screen encrustation on (1) a countywide basis, by assembling and analyzing chemical and hydrologic data from wells throughout the county, and (2) a site-specific basis, by analyzing water samples and iron-biofilm samples from affected wells and sediment cores from potential well sites.

Purpose and Scope

This report summarizes and interprets geochemical and microbiological data collected from test borings and affected wells to define the geochemical processes that cause well-screen encrustation at specific sites. The report describes the role of iron-related bacteria in the encrustation process and relates their occurrence to chemical conditions in the wells; it also discusses the role of well operation in well-screen encrustation and the effect of well-screen encrustation on well-water quality. A companion report (Walter, 1997) describes the distribution of well-screen encrustation within Suffolk County and the relation of specific-capacity declines to spatial trends in water quality.

Iron-Related Well-Screen Encrustation and Aquifer Biofouling

Iron-related well-screen encrustation and aquifer biofouling and the subsequent loss in well yield are costly problems that are a concern locally, nationally, and internationally (Cullimore, 1986). Iron-related specific-capacity declines occur in a wide variety of geologic settings and are common in wells screened in unconsolidated sediments, such as those underlying Suffolk County (Driscoll, 1986). Previous investigations of iron-related well-screen encrustation and aquifer biofouling have been reported by Applin and Zhao (1989), van Beek and Kooper (1980), Cullimore (1986), Gaffney (1986), and Walter (1989). Specific-capacity declines observed in the SCWA well network are typically associated with high concentrations of iron in the ground water; the well screens and casings in affected wells are typically coated with iron precipi-

tates (iron oxyhydroxides), which results in a decreased specific capacity and well yield.

Iron-related specific-capacity declines also are caused by the growth of iron-bacteria biofilms on well screens and in the surrounding formation; the iron bacteria biofilms consist of iron oxyhydroxides and cellular matter. The ecology of iron bacteria and their role in well-screen encrustation has been discussed by Harder (1919), Ghiorse (1984 and 1986), Pringsheim (1949), Alcalde and Castronovo de Knott (1986), and Halbeck and Pederson (1986). Iron-related bacteria grow naturally in environments with steep Eh gradients, such as ground-water seeps and wetlands. Similar conditions can develop in the vicinity of a well when oxygenated ground water mixes with anoxic ground water that contains ferrous iron. In this report, the terms "iron-related well-screen encrustation" and "aquifer biofouling" are used to describe the geochemical and microbial processes that contribute to declines in the specific capacity of a well. Manganese-related well-screen encrustation, which involves the oxidation of manganese(II) and the precipitation of manganese oxyhydroxides, is less common, but can occur in some hydrogeologic environments. Most iron bacteria also can form biofilms consisting of manganese oxyhydroxides and cellular material.

The most common well-redevelopment procedure entails removing and steam cleaning the pump, and adding sulfamic acid to the well for about 72 hours to dissolve the iron biofilm. The well is then surged and pumped until the water is of background water quality; the amount of water pumped varies with well depth. The specific capacity of the wells is measured before and after the redevelopment procedure, and the procedure is repeated if recovery is insufficient. Production wells are typically 16 to 24 in. in diameter, have a screen length of 30 to 90 ft and range from 55 to 830 ft deep.

Acknowledgments

Gratitude is extended to Steven Colabufo, Frank Alescio, and Paul Kuzman of the SCWA for their assistance in coordinating field activities, and to Roy Cullimore (University of Saskatchewan) and George Alford (ARCC Inc.) for advice and assistance in iron-bacteria identification and analysis. The author also thanks Eleanor Robbins of the USGS for instruction in iron-bacteria identification, and Alfred Mikell of the University of Mississippi at Oxford for assistance in microbiological analyses.

HYDROGEOLOGIC SETTING

Long Island, at the northern extent of the Atlantic Coastal Plain, is underlain by a sequence of unconsolidated gravel, sand, silt, and clay units that were deposited on crystalline bedrock. The sand and gravel units form an aquifer system that is the sole source of potable water for Nassau and Suffolk Counties.

The bedrock surface dips southeastward from a maximum elevation of sea level in the northwestern corner of Queens County to 2,000 ft below sea level in the south-central part of Suffolk County (Smolensky and others, 1989). The average dip of the bedrock surface is about 65 ft/mi (McClymonds and Franke, 1972). The overlying unconsolidated sediments, which range in thickness from 0 to more than 2,000 ft, consist of Pleistocene glaciofluvial gravel, sand, silt, and clay and underlying Cretaceous fluvial and nonmarine deltaic deposits associated with Atlantic Coastal Plain sediments (Jensen and Soren, 1971). Marine sediments underlie the Pleistocene deposits near the southern shore. The stratigraphy of the unconsolidated sediments is summarized in table 1; a generalized north-south hydrogeologic section representing an area near the Nassau-Suffolk County border is shown in figure 2.

The Upper Pleistocene deposits consist primarily of sand and gravel with interbedded lenses of silt and clay (fig. 2). The sediments are mostly quartz and feldspar with minor amounts of biotite, muscovite, and rock fragments (Smolensky and others, 1989); they were deposited as terminal-moraine deposits along the northern shore and as glacial outwash to the south. The glacial deposits are as much as 700 ft thick in areas where the surface of the underlying Cretaceous deposits is heavily eroded. The Pleistocene unit contains two major clay units—the Smithtown clay in central Suffolk County and the "20-foot" clay along part of the southern shore.

Underlying the Pleistocene deposits along the southern shore is the Gardiners Clay (fig. 2), which is of marine origin and consists of glauconitic clay with interbedded silt and sand. The underlying Matawan Group-Magothy Formation consists primarily of quartz sand with interbedded gravel, silt, and clay. These sediments were deposited as part of a large delta

Table 1. Description of major hydrogeologic and stratigraphic units underlying Suffolk County, N.Y.
[Modified from Jensen and Soren, 1971, table 1]

Hydrogeologic unit	Stratigraphic unit	Description and water-bearing character
Upper glacial aquifer	Holocene and Upper Pleistocene deposits	Mainly brown and gray sand and gravel of moderate to high hydraulic conductivity; also includes deposits of clayey till and lacustrine clay of low hydraulic conductivity. A major aquifer.
Gardiners Clay	Gardiners Clay	Green and gray clay, silt, clayey and silty sand, and some interbedded clayey and silty gravel. Low hydraulic conductivity. Unit confines water in underlying aquifer.
Monmouth greensand	Monmouth Group, undifferentiated	Interbedded marine deposits of dark-gray, olive-green, dark-greenish-gray, and greenish-black glauconitic and lignitic clay, silt, and clayey and silty sand. Unit has low hydraulic conductivity and tends to confine water in underlying aquifer.
Magothy aquifer	Matawan Group and Magothy Formation, undifferentiated	Gray and white fine to coarse sand of moderate hydraulic conductivity. Generally contains sand and gravel beds of low to high hydraulic conductivity in basal 100 to 200 feet. Contains much interstitial clay and silt, and beds and lenses of clay of low hydraulic conductivity.
Raritan confining unit (Raritan clay)	Unnamed clay member of the Raritan Formation	Gray, black, and multicolored clay and some silt and fine sand. Unit has low hydraulic conductivity and tends to confine water in underlying aquifer.
Lloyd aquifer	Lloyd Sand Member of the Raritan Formation	White and gray fine to coarse sand and gravel of moderate hydraulic conductivity and some clayey beds of low hydraulic conductivity.
Bedrock	Undifferentiated crystalline rocks	Mainly metamorphic rocks of low hydraulic conductivity. Surface generally weathered. Considered to be the bottom of the aquifer system.

and show a fining-upward sequence. The unit, which pinches out in the northwestern corner of Suffolk County, generally is finer grained than the upper Pleistocene deposits and ranges from 0 to more than 1,000 ft in thickness. Lignite is common throughout the unit as discontinuous beds and lenses.

Beneath the Magothy Formation-Matawan Group, throughout Long Island, is the unnamed clay and Lloyd Sand Member of the Raritan Formation. The Lloyd Sand Member consists primarily of quartz sand with interbedded gravel, silt, and clay, whereas the overlying unnamed clay member consists primarily of silt and clay with interbedded sand. The maximum thickness of the unnamed clay member and Lloyd Sand Member of the Raritan Formation are about 200 ft and 500 ft, respectively.

Ground-Water Flow

The Upper Pleistocene, Magothy Formation-Matawan Group, and Lloyd Sand Member of the Raritan Formation constitute the upper glacial, Magothy, and Lloyd aquifers, respectively. The upper glacial and Magothy aquifers are the two principal aquifers in Suffolk County; the SCWA operates only about four wells screened in the Lloyd aquifer, which is a major source of supply in neighboring Nassau County. The upper glacial aquifer is moderately to highly permeable, whereas the Magothy aquifer is moderately permeable. Average reported horizontal hydraulic conductivity of the upper glacial and Magothy aquifers is 270 ft/d and 50 ft/d, respectively (Smolensky and others, 1989).

Precipitation is the sole source of recharge to the ground-water system of Long Island. About 50 percent of precipitation is lost through evapotranspiration and as surface runoff, and about 50 percent recharges the aquifer at the water table (Warren and others, 1968).

The water table lies within the upper glacial aquifer throughout Suffolk County. The upper glacial aquifer is mostly unconfined, but confined conditions are found near extensive clay units. The Magothy aquifer is confined along the southern shore by the Gardiners Clay, and the Lloyd aquifer is confined by the Raritan confining unit, which is the hydrogeologic equivalent of the unnamed clay member of the Raritan Formation (fig. 2).

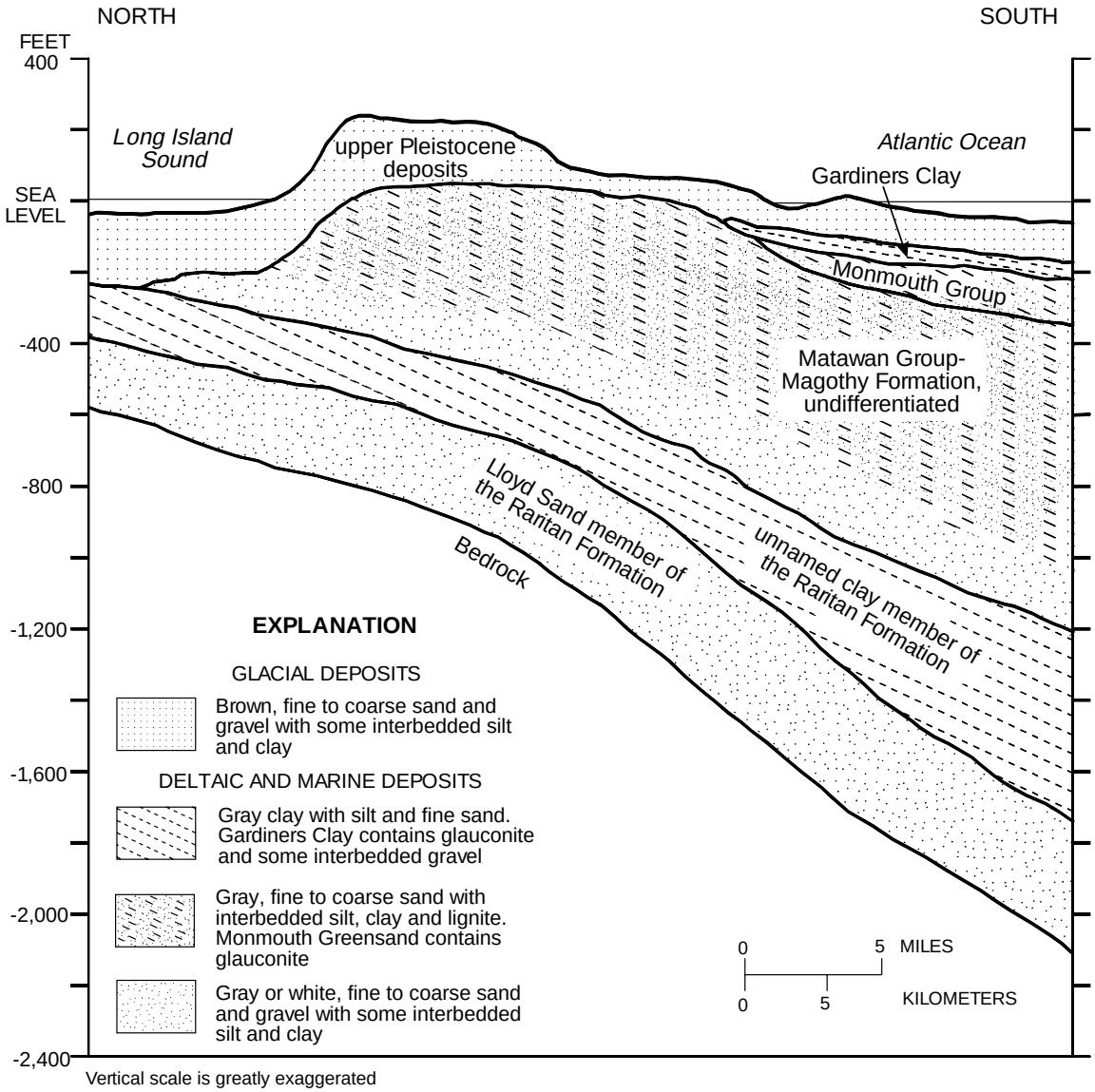


Figure 2. Stratigraphy along a generalized north-south vertical section near the Nassau-Suffolk County border, Long Island, N.Y. (Modified from McClymonds and Franke, 1972, fig. 3.)

A ground-water divide forms an east-west-trending line along the center axis of the island. The hydraulic gradient is steeper north of the divide than south of it because (1) the aquifer is thinner, (2) the topographic relief is greater, and (3) the hydraulic conductivity of the moraine deposits is lower than that of the outwash to the south. A generalized section showing directions of ground-water flow is given in figure 3. Water entering the ground-water system near the ground-water divide, where flow is nearly vertical, recharges the deep aquifers and discharges offshore, whereas water entering near the coast flows in shallow-flow subsystems and discharges to streams and saltwater embayments (Nemickas and others, 1989). The distribution of ground-water discharge areas in Suffolk County, and the relation between ground-water-flow patterns and water quality and the occurrence of iron-related well-screen encrustation and aquifer biofouling, is discussed in Walter (1997).

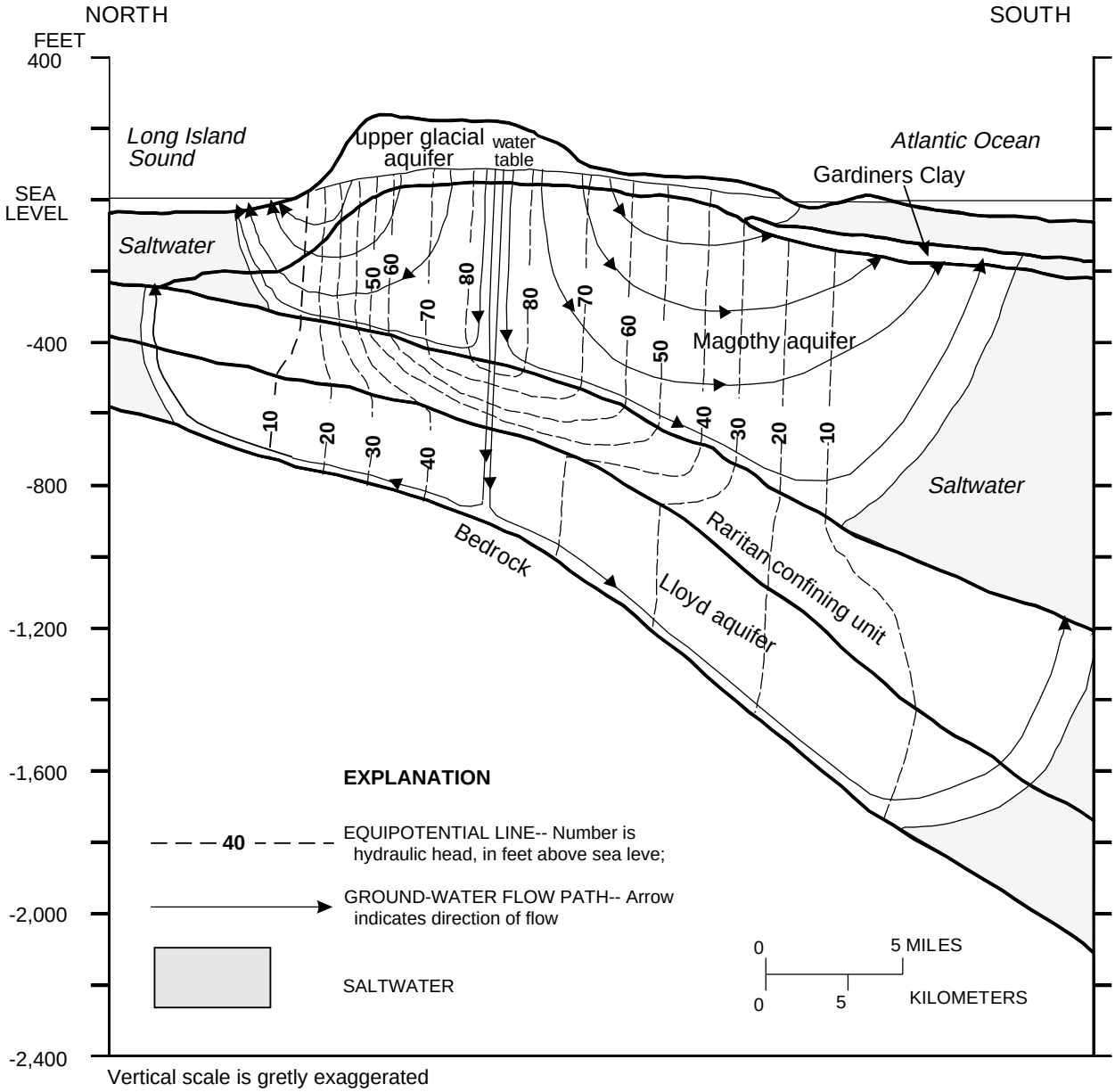


Figure 3. Hydraulic heads and ground-water flow paths along a generalized north-south vertical section near the Nassau-Suffolk County border, Long Island, N.Y. (Modified from Reilly and others, 1983, fig. 4.)

Ground-Water Quality

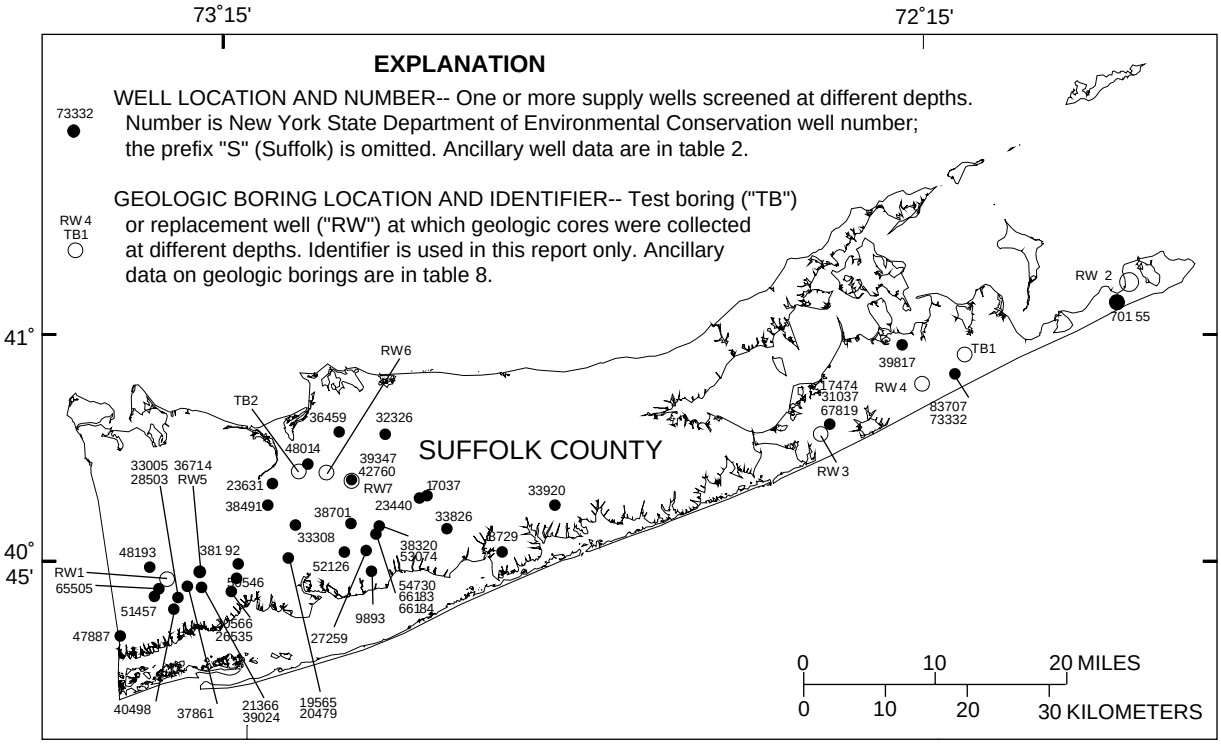
Natural ground-water quality is a function of the geochemical conditions in the aquifer, but ground-water quality in the upper glacial aquifer and the shallow part of the Magothy aquifer has been affected by human activity and land use (Eckhardt and others, 1988). As oxygenated ground water flows through the aquifers, oxygen is consumed through the oxidation of organic matter, including lignite, and geochemical conditions become more reducing. The continued oxidation of lignite and other organic matter, and the subsequent dissolution of iron oxyhydroxide coatings on the aquifer material, increase dissolved-iron concentrations. This mechanism has been identified as the cause of high alkalinity of ground water elsewhere in the Magothy aquifer (Knobel and Phillips, 1988). As reducing conditions intensify, sulfate is reduced to sulfide, and pyrite (FeS_2) can precipitate. Authigenic pyrite is common in the Magothy aquifer along the southern shore of Long

Island. The pumping-induced downward migration of oxygenated water into the Magothy aquifer in southwestern Suffolk County, and the oxidation of authigenic pyrite, could explain the low pH observed in parts of the aquifer (Walter, 1997).

Ground water in shallow parts of the upper glacial aquifer, which is recharged by precipitation, is characterized by relatively low pH (less than 6.0), high dissolved-oxygen concentrations (greater than 10 mg/L), low specific conductance (less than 100 µS/cm), and low concentrations of iron and manganese (less than 0.05 mg/L) (de Laguna, 1964; Scorca, 1990; Walter, 1997). In downgradient areas, where conditions are reducing, ground water typically has lower dissolved-oxygen concentrations (less than 1 mg/L), higher phosphorus concentrations (greater than 0.1 mg/L), and higher dissolved-iron and manganese concentrations (greater than 0.5 mg/L) than elsewhere (Walter, 1997). The iron-related declines in specific capacity of wells have occurred primarily in ground-water discharge zones along the southern shore and in the central part of Suffolk County, where concentrations of dissolved iron and phosphorus are higher, and concentrations of dissolved oxygen lower, than in the recharge areas.

DATA COLLECTION AND ANALYSIS

Water samples, biofilm samples, and sediment cores were collected from affected wells and test borings to identify the geochemical and microbiological mechanisms involved in the well-screen encrustation process. Locations of data-collection sites are shown in figure 4. Well abbreviations, SCWA well names, and the New York State Department of Environmental Conservation well numbers are given in table 2 with the corresponding aquifers and depths. A list of abbreviations for sediment-core collection sites, either replacement wells or test borings, is given in table 8 (p. 24).



Base from U.S. Geological Survey digital data 1:24,000, 1990
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Figure 4. Location of Suffolk County Water Authority wells and geologic borings used in iron-encrustation study, Suffolk County, N.Y. (Well numbers, names, and abbreviations are listed in tables 2 and 8.)

Table 2. Depths of Suffolk County Water Authority wells used in biofouling investigation, Suffolk County, N.Y., and geologic units tapped

[SCWA, Suffolk County Water Authority; NYSDEC, New York State Department of Environmental Conservation; glacial, upper glacial. Depths are in feet below land surface. Locations are shown in fig. 4]

Well identifier				
SCWA name	Abbreviation	NYSDEC number	Aquifer tapped	Well depth
Astor Ave - 3	AA-3	S48014	Magothy	343
August Rd - 4	AR-4	S37861	Magothy	636
Beechnut Ave - 1	BA-1	S23440	Glacial	165
Bellmore Rd - 1	BR-1	S19565	Glacial	129
Bellmore Rd - 2	BR-2	S20479	Glacial	128
Bicycle Path- 1	BCR-1	S32326	Glacial	160
Blue Point Rd - 1	BPR-1	S38320	Glacial	173
Blue Point Rd - 4	BPR-4	S53074	Glacial	169
Bridgehampton Rd - 2	BHR-2	S83707	Glacial	123
Bridgehampton Rd - 5	BHR-5	S73332	Glacial	150
Brook Ave - 3	BA-3	S36714	Magothy	308
Church St (Bohemia) -1	CSB-1	S52126	Glacial	156
Church St (Holbrook) - 1	CSH-1	S27259	Glacial	190
Circle Dr - 4	CD-4	S48193	Glacial	534
Division St - 2	DS-2	S38917	Glacial	173
East Forks Rd - 4	EFR-4	S38192	Magothy	605
Fifth Ave - 32	FA-32	S20566	Magothy	775
Fifth Ave -33	FA-33	S26535	Magothy	776
Gordon Ave - 2	GA-2	S65505	Magothy	650
Green Belt Pkwy - 1	GBP-1	S54730	Magothy	259
Green Belt Pkwy - 3	GBP-3	S66183	Magothy	543
Green Belt Pkwy - 4	GBP-4	S66184	Magothy	384
Greene Ave - 8	GA-8	S47887	Magothy	648
Harvest La - 1	HL-1	S21366	Magothy	470
Harvest La - 3	HL-3	S39024	Magothy	622
Lafayette Rd - 1	LR-1	S28503	Magothy	676
Lafayette Rd - 2	LR-2	S33005	Magothy	681
Lakeview Ave - 3	LKA-3	S09893	Glacial	103
Lincoln Ave - 1	LA-1	S38701	Glacial	202
Long Springs Rd - 3	LSR-3	S17474	Glacial	103
Long Springs Rd - 5	LSR-5	S31037	Magothy	287
Long Springs Rd - 6	LSR-6	S67819	Magothy	280
Margin Dr - 1	MD-1	S18729	Magothy	356
Montauk Point State Blvd - 1	MPSB-1	S70155	Glacial	243
New York Ave - 5	NYA-5	S23631	Magothy	623
Nicolls Rd - 2	NR-2	S33308	Glacial	132
Pleasant Ave - 1	PA-1	S39347	Glacial	176
Pleasant Ave - 2	PA-2	S42760	Glacial	173
Race Ave - 1	RA-1	S17037	Glacial	155
Railroad Ave - 1	RRA-1	S33920	Magothy	629
Sawyer Ave - 3	SA-3	S40498	Magothy	748
Station Rd - 1	SR-1	S33826	Glacial	163
Stem La - 2	SL-2	S36459	Magothy	523
Thomas Ave - 2	TA-2	S50546	Magothy	668
Twelfth St - 4	TS-4	S51457	Magothy	625
Wheeler Rd - 4	WR-4	S38491	Magothy	403

Water Quality

A total of 66 water samples were collected from 40 affected wells from January 1991 through August 1992. Additional water samples were collected from the lubrication systems of 16 affected wells. Treated system water is trickled into the well during idle periods to keep pump turbines wet—this process is referred to as "well lubrication." Water samples were collected before and after the reconditioning of 26 affected wells. A sufficient volume of ground water was evacuated after reconditioning to ensure that no residual effects of the reconditioning process remained; withdrawal volumes ranged from 0.84 to 5.29 Mgal. An additional 14 water samples were collected from wells that had histories of iron-related production declines but were not reconditioned during the study. Water samples were collected with a plastic drop line that was placed in the discharge stream of the well and sent to the SCWA Central Laboratory in Oakdale, N.Y., for analysis for total and dissolved constituents. Dissolved-constituent samples were filtered through a 0.45–µm filter in the field. Duplicate samples were collected from nine sites. Samples collected for dissolved-organic-carbon, dissolved-silica, and duplicate dissolved-iron analysis were sent to the USGS Central Laboratory in Arvada, Colo. Dissolved oxygen was measured in a flow-through cell with a Yellow Springs Instruments Company[1] dissolved-oxygen meter during sampling. Temperature was field measured with a mercury thermometer, specific conductance was field measured with an Orion specific-conductance meter, and pH was field measured with an Orion pH electrode and meter. Alkalinity was field measured with a Hach field-titration unit; samples were titrated to a fixed end point of pH 4.5. Ferrous and ferric iron were field measured in triplicate by the 2,2´-bipyridine method (Arthur Horowitz, U.S. Geological Survey, written commun., 1980).

[1] The use of trade, product, or firm names in this report is for identification or location purposes only and does not imply endorsement of products by the U.S. Geological Survey, nor impute responsibility for any present or potential effects on the natural resources of the United States.

Encrusting Material

Samples of encrusting material were collected from the pump columns of 43 affected wells during reconditioning; the samples were collected with a sterilized metal scraper and distilled water. At some wells, material was collected from the screen zone of the well during well replacement. The material was analyzed with a light microscope to identify any filaments characteristic of iron-producing bacteria (Eleanor Robbins, U.S. Geological Survey, oral commun., 1990). Chemical analysis of solid-phase material from 20 wells entailed digesting the samples in heated nitric acid (HNO_3) and analyzing the solution for trace elements by atomic absorption. An additional five solid-phase samples were analyzed with a microprobe equipped with a KEVEX energy-dispersion system, and another 16 samples were analyzed for mineral content by x-ray diffraction. Energy-dispersion and x-ray diffraction analyses were done by the Department of Earth Sciences at the State University of New York, Stony Brook, N.Y.

Sediment Cores

A total of 118 sediment cores were collected from 7 replacement wells and 2 test borings to detect the presence of iron bacteria in the aquifer sediments. The outer parts of the sediment cores were removed with a sterilized metal scraper to minimize contamination from the drilling fluid. The middle parts of the cores were mixed with triple-distilled water and placed in a sterilized steel filter press. The water was forced through a filter membrane by nitrogen gas. The effluent was collected in a sterilized test tube and placed in an iron-specific Biological Activity Reaction Test (BART). The BART test uses a modified Winogradsky's medium to test for the presence of iron-related bacteria; an intercedant ball allows an oxygen gradient to form in the tube and enables several species of iron bacteria to grow in the same test (George Alford, ARCC Inc., oral commun., 1992). Field blanks were tested with triple-distilled water. Samples of drilling fluid also were analyzed to ensure that any positive reactions in the cores did not result from contamination by drilling fluid.

GEOCHEMISTRY

Iron-related well-screen encrustation is, at least in part, an abiotic process that involves the oxidation of ferrous iron to ferric iron and the precipitation of insoluble ferric-iron mineral phases, such as amorphous ferric hydroxide ($Fe(OH)_3$). This process is kinetically controlled and is a function of redox conditions and water chemistry. Samples of ground water and of encrusting material were collected from several wells that had iron-related encrustation to determine the geochemical controls on the encrustation process on the basis of the bulk chemistry and mineralogy of the material. Chemical data from water samples were used to determine redox conditions in well waters, the kinetics of ferrous-iron oxidation, and the state of saturation of selected iron-oxyhydroxide minerals in well water and in treated system water.

Mineralogy and Solid-Phase Chemistry

The encrusting material generally was orange or brown and became powdery when dry; the encrusting material from some wells was black.

Chemical Digestions

Results of 20 chemical digestions (table 3) indicate that the encrusting material consists primarily of iron (Fe) with trace amounts of manganese (Mn), copper (Cu), zinc (Zn), lead (Pb), selenium (Se), and arsenic (As). Manganese concentrations ranging from <100 mg/Kg to 1,000 mg/Kg indicate that manganese is only a minor constituent of the encrusting material and not an important biofouling agent in Suffolk County. Arsenic and selenium were detected in 19 samples; solid-phase concentrations of arsenic ranged from <10 to 785 mg/Kg, and concentrations of selenium ranged from <20 to 653 mg/Kg. Lead was detected in 18 of the 20 samples; concentrations were as high as 530 mg/Kg. Copper and zinc were detected in 17 and 12 samples, respectively; maximum concentrations were 1,000 mg/Kg and 600 mg/Kg, respectively. Concentrations of trace elements in the encrusting material, except for manganese, were typically below detection limits in the water samples. The presence of these trace elements in encrusting material, despite their absence above detection limits in ground water could be due to the high flow rates of the water around the well and to the ability of iron oxyhydroxides to effectively coprecipitate with and sorb trace elements (Hem, 1985). The data indicate that trace elements, except for iron, constitute only a small fraction of the encrusting material. The total percentages of trace elements (by weight) are included in table 3. The weight percents ranged from 0.07 to 1.19 percent and averaged 0.26 percent.

The concentration of iron in the encrusting material ranged from 168,000 mg/Kg to 513,000 mg/Kg. On the assumption that the iron was present as ferric hydroxide, $Fe(OH)_3$, the solid-phase concentrations of iron were converted to ferric hydroxide concentrations by adding the equivalent weight of hydrogen and oxygen that would be present in the hydroxide. The equivalent concentrations and weight percents of ferric hydroxide are included in table 3. The weight percent of ferric hydroxide in the encrusting material ranged from 32.3 to 98.6 percent and averaged 64.3 percent.

The weight percent of material not accounted for in the digestions was determined from the weight percents of ferric hydroxide and trace elements. The remaining material is assumed to consist primarily of organic matter and aluminosilicate mineral phases — concentrations of carbon, hydrogen, and oxygen (which are the primary components of organic matter) could not be determined, and silica and aluminum were not included in the analysis. The weight percentages of remaining material (table 3) ranged from 1.3 to 67.5 percent and averaged 34.6 percent.

Energy-Dispersion Analysis

Five samples of encrusting material were analyzed through energy-dispersion spectrometry with an electron microbeam analyzer to determine whether silica was present in significant amounts. When atoms of a given element are bombarded with electrons, they emit atomic particles of a known energy; the emitted energy shows as a characteristic peak on the energy spectrum, and the height of the peak is a function of the concentration of the element in the sample. Energy dispersion can be used to identify elements with atomic numbers equal to or greater than 13 (aluminum) that are present in a sample in significant amounts. The method cannot be used to determine whether organic material is present because it cannot detect carbon (C), hydrogen (H), or oxygen (O).

Table 3. Iron and trace-element content of screen-encrusting material from selected wells in Suffolk County, N.Y., as determined by chemical digestion[1]

[Concentrations are in milligrams per kilogram (mg/Kg). <, less than. Well locations are shown in fig. 4. Analyses by Suffolk County Water Authority Central Laboratory, Oakdale, N.Y.]

Well	Concentrations of iron		Weight percent ferric hydroxide	Concentrations of trace elements										Total weight percent trace elements[3]	Total weight percent remaining material[4]
	Iron	Iron as ferric hydroxide[2] [$Fe(OH)_3$]		Manganese	Copper	Zinc	Chromium	Arsenic	Barium	Cadmium	Lead	Selenium	Gold		
AA-3	214,000	411,000	41.1	300	400	600	<10	785	416	8	40	481	<25	0.31	41.4
AR-4	272,000	523,000	52.3	<100	200	300	<10	284	<100	6	185	407	<25	.16	47.5
BA-3	355,000	682,000	68.2	500	500	400	<10	75	<100	5	27	264	<25	.19	31.6
EFR-4	380,000	730,000	73.0	<100	<200	<200	<10	121	<100	<5	34	11	<25	.08	26.9
FA-32	420,000	808,000	80.8	<100	400	<200	45	373	<100	<5	134	21	<25	.14	19.1
FA-33	350,000	673,000	67.3	100	700	<200	<10	536	<100	<5	163	358	<25	.22	32.5
GA-2	168,000	323,000	32.3	100	500	300	<10	208	<100	<5	530	185	<25	.20	67.5
GA-8	293,000	563,000	56.3	<100	700	400	<10	124	<100	7	255	479	<25	.22	43.5
GBP-1	314,000	603,000	60.3	2,800	400	400	<10	122	<100	16	680	653	<25	.51	39.2
HL-1	296,000	569,000	56.9	100	600	400	<10	646	<100	<5	59	408	<25	.22	42.9
HL-3	290,000	557,000	55.7	600	800	300	<10	392	<100	6	85	438	<25	.28	44.0
LR-1	328,000	630,000	63.0	1,000	900	<200	<10	29	<100	<5	23	<20	<25	.23	36.8
LR-2	350,000	673,000	67.3	1,100	900	200	<10	79	<100	4	23	235	<25	.27	32.4
LSR-5	513,000	986,000	98.6	400	<200	<200	<10	<10	<100	<5	<10	24	<25	.14	1.3
LSR-6	266,000	511,000	51.1	8,900	1,000	600	<10	207	648	7	115	381	<25	1.19	47.7
MD-1	318,000	611,000	61.1	1,700	500	400	<10	113	<100	7	12	590	<25	.35	38.6
MSPB-1	433,000	832,000	83.2	<100	<200	<200	26	30	<100	<5	<10	7	<25	.07	16.7
NYA-5	430,000	826,000	82.6	<100	700	<200	<10	289	<100	22	13	23	<25	.15	17.3
SA-3	350,000	673,000	67.3	<100	300	<200	<10	130	<100	<5	23	257	<25	.12	32.6
TS-4	350,000	673,000	67.3	300	400	200	<10	212	<100	6	436	508	<25	.22	32.5

[1] Analyses do not include silica or aluminum.

[2] Calculated, on assumption that iron is present as $Fe(OH)_3$.

[3] Nondetection values were assumed to be detection limit.

[4] Unaccounted for material—assumed to be organic matter and siliceous material.

Four of the five samples of encrusting material were orange or brown and had energy peaks for iron (Fe) and silica (Si); the absence of other peaks indicated that iron and silica were the primary inorganic components of the material. Trace-element concentrations in the encrusting material were too low to be detected by energy-dispersion analysis. Silica content was variable in each sample, and silica was the most abundant element in parts of the four samples. Electron-microprobe images of sample LSR-5 showed zones of iron-rich crystalline material containing no silica, alternating with a conchoidal phase consisting almost entirely of silica; this suggests that silica can be a major component of the encrusting material. A fifth sample of black encrusting material from well HL-1 was found to consist primarily of iron and sulfur with small amounts of copper, phosphorus, and silica.

X-ray Diffraction

X-ray diffraction was used to determine the mineralogy of 17 samples of encrusting material; the results are summarized in table 4. Sixteen of the 17 samples were orange or brown; the remaining sample was black. X-ray diffraction uses reflected x-rays of a known wavelength to determine the characteristic d-spacing (spacing in the crystal lattice) of crystalline mineral phases present in a sample. The presence of a certain mineral phase will cause a characteristic peak at an angle that corresponds to the d-spacing in the crystal lattice of the mineral. The size of the peak is a function of the abundance of the mineral phase in the sample; small peaks indicate that the material is weakly crystalline, and no peaks indicate that the material is amorphous.

The data in table 4 indicate that goethite (FeOOH) was present in all 16 orange or brown samples, and hematite was present in only 2. A mineral phase that was either magnetite (Fe_3O_4) or maghemite (Fe_2O_3), which is a polymorph of hematite whose crystal structure is similar to that of magnetite, was detected in 7 of the orange or brown samples. Quartz was detected in 10 orange or brown samples. Although crystalline mineral phases were present in the encrusting material, the x-ray diffraction peaks were small, indicating that the crystalline phases were present in small amounts. The data indicated that all 16 orange or brown samples were weakly crystalline and that most of the material was amorphous. Chemical-digestion, energy-dispersion, and x-ray diffraction data indicate that the organic part of the orange or brown encrusting material is primarily amorphous ferric hydroxide with lesser amounts of crystalline iron oxyhydroxide mineral phases, quartz, and trace elements. The black sample showed peaks for quartz but not for iron minerals. X-ray diffraction and energy-dispersion spectrometry data suggest that the organic part of the black material was composed mostly of an amorphous or weakly crystalline iron-sulfide mineral phase with lesser amounts of quartz. Some quartz peaks could be from sand grains entrained in the encrusting material.

Table 4. Mineralogy of screen-encrusting well material as identified by x-ray diffraction

[X denotes detection. All peaks were small, indicating trace amounts. Well locations shown in fig. 4]

Well	Mineral phases				
	Goethite	Hematite	Maghemite	Magnetite	Quartz
AA-3	X				X
BA-3	X				
CSB-1	X		X	X	X
EFR-4	X				X
FA-32	X				
FA-33	X				X
GA-2	X		X	X	X
HL-1*					X
LA-1	X		X	X	X
LR-2	X		X	X	
LSR-5	X		X	X	X
MD-1	X		X	X	X
MSPB-1	X				X
NYA-5	X				
PA-1	X	X	X	X	X
SL-2	X				
TA-2	X	X			

* Black precipitate with sulfur odor.

Results of Water-Quality Sampling

The quality of water in a well is a function of the location of the well screen in the ground-water system and the geochemical conditions around the well screen. Results of chemical analyses of water samples collected during the study are presented in table 9 (at end of report). Dissolved-oxygen concentration ranged from 10.8 mg/L in recharge areas of the upper glacial

aquifer to 0.2 mg/L in the downgradient parts of the Magothy aquifer. The median dissolved-oxygen concentration was 4.6 mg/L. Field-measured alkalinity ranged from 2 to 35 mg/L as $CaCO_3$, and pH ranged from 4.9 to 7.5. The median field-measured alkalinity and pH values were 12.0 mg/L and 5.4, respectively. Dissolved-iron concentration ranged from <0.01 mg/L to 1.4 mg/L, and dissolved-manganese concentration ranged from <0.01 to 0.51 mg/L. The median concentrations of dissolved iron and manganese were 0.21 and 0.09 mg/L, respectively. Results of chemical analyses of lubrication water from several affected wells are presented in table 10 (at end of report). Field-measured alkalinity ranged from 18 mg/L to 133 mg/L, pH ranged from 6.5 to 8.4, and dissolved-oxygen concentration ranged from 0.4 mg/L to 11.2 mg/L. Dissolved-iron concentration ranged from 0.04 mg/L to 0.58 mg/L.

Mineral-Saturation Indices

The aqueous speciation model WATEQF (Plummer and others, 1984) was used to calculate the saturation indices of several iron and manganese oxyhydroxides in water samples from 33 pumping wells. The model uses water-quality data to calculate the state of equilibrium with respect to various mineral phases in the water from thermodynamic data. The saturation index (SI) is defined as:

$$SI = \log(IAP)/K_{eq} \qquad (1)$$

where *IAP* is the ion activity product, and K_{eq} is the equilibrium constant (Snoeyink and Jenkins, 1980). An *SI* greater than 0 indicates that a mineral phase would be expected to precipitate from the ground water, and an *SI* less than 0 indicates that the mineral phase will dissolve in the ground water; an *SI* of 0 indicates that the ground water is in equilibrium with the mineral phase.

Saturation indices for all manganese oxyhydroxide mineral phases were less than zero, indicating that the well waters were undersaturated with respect to those mineral phases and that manganese does not play an important role in the encrustation process. This agrees with solid-phase chemistry data that show manganese to be only a trace component of the encrusting material.

The saturation indices for iron oxyhydroxide mineral phases (fig. 5A, 5B) indicate that the well water is supersaturated with respect to goethite (FeOOH), hematite (Fe_2O_3), and magnetite (Fe_3O_4); therefore, those mineral phases are thermodynamically favored to precipitate from the ground water. X-ray-diffraction data confirm that these mineral phases generally are present in the encrusting material. Maghemite (Fe_2O_3), a polymorph of hematite, was favored to precipitate in eight water samples.

The ground water was undersaturated with respect to ferric hydroxide ($Fe(OH)_3$) and supersaturated with respect to quartz (fig. 5C, 5D). The low saturation indices of ferric hydroxide are due to the low pH of the ground water; if pH values were close to neutral (pH 7), ferric hydroxide would be thermodynamically favored to precipitate from the well waters. Although ferric hydroxide is not favored to precipitate from ground water, mineralogical and chemical data indicate that it is the primary inorganic component of the encrusting material. This suggests that much of the encrusting material observed in the wells may not precipitate while the wells are being pumped.

Oxidation-Reduction Conditions

Iron-related well-screen encrustation is an oxidation-reduction process that involves the oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) and the subsequent precipitation of ferric hydroxide. The principal reactions between ferrous iron and oxygen are given below:

$$4Fe^{2+} \leftrightarrow 4Fe^{3+} + 4e^- \qquad (2)$$

$$4e^- + 4H^+ + O_2 \leftrightarrow 2H_2O \qquad (3)$$

$$4Fe^{3+} + 12H_2O \leftrightarrow 4Fe(OH)_3 + 12H^+ \qquad (4)$$

$$4Fe^{2+} + 10H_2O + O_2 \leftrightarrow 4Fe(OH)_3 + 8H^+ \qquad (5)$$

Equations 2 and 3 are the two half reactions in which ferrous iron is oxidized to ferric iron (eq 2) and oxygen is reduced to form water (eq 3). Equation 4 is the reaction between ferric iron and water to form ferric hydroxide. Equation 5 is the overall reaction that describes the primary geochemical process involved in iron-related well-screen encrustation.

Oxidation-reduction conditions in 18 wells were estimated from ferrous- and ferric-iron concentrations and from dissolved-oxygen concentrations. Ferrous-

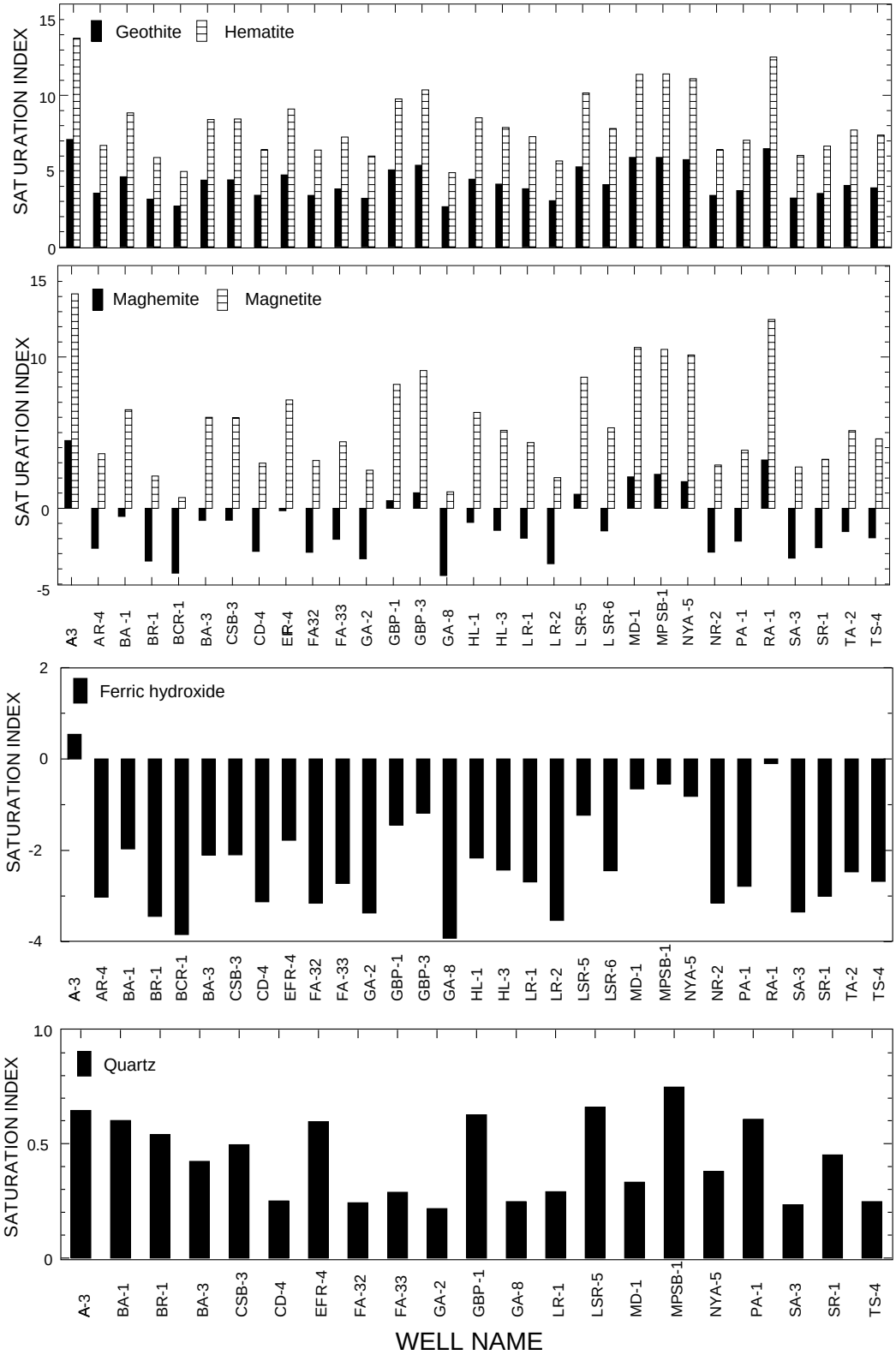

Figure 5. Saturation indices for selected mineral phases in water samples from 31 wells in Suffolk County, N.Y., as indicated by chemical equilibrium model WATEQF. (Well locations shown in fig. 4.) A. Goethite, hematite. B. Maghemite, magnetite. C. Ferric hydroxide. D. Quartz.

and ferric-iron data presented in table 5 indicate that ferrous iron is the predominant iron species in the well; it constituted between 81.6 and 99.4 percent of the total iron in almost all of the ground-water samples. The only well at which ferrous iron constituted less than 81.6 percent of the total iron is well FA-33, at which ferrous iron was 49.4 percent of the total iron.

The ferrous- and ferric-iron data indicate that conditions in the wells are reducing with respect to iron, whereas the presence of oxygen at concentrations equal to or greater than 0.1 mg/L suggest that conditions are more oxidizing. For example, water samples from well HL-3 (data not shown in table 9) contained 2.15 mg/L of ferrous iron and 4.7 mg/L of dissolved oxygen. Ferrous iron does not exist in equilibrium with detectable amounts of dissolved oxygen; therefore, the water was in a state of redox disequilibrium. Similar conditions were observed in all wells sampled. Redox disequilibrium is common in natural systems in which ground-water mixing is occurring.

The Nernst equation was used to calculate the redox potential (Eh) of ground water from the ferrous/ferric iron redox couple (eq 7) and the dissolved oxygen/water redox couple (eq 8). The general form of the Nernst equation (eq 6) and the corresponding equations for iron and oxygen (eq 7, 8) are as follows:

$$Eh = E_0 - (2.303RT/nF)\log Q, \quad (6)$$

$$Eh = +0.77 - (0.0159/1)\log([Fe^{2+}]/[Fe^{3+}]), \text{ and} \quad (7)$$

$$Eh = +1.23 - (0.0159/4)\log(1/pO_2[H^+]^4), \quad (8)$$

where

E_0 is the standard potential of the reaction,
R is the gas constant,
T is temperature, in degrees Kelvin,
n is the number of moles of electrons exchanged in the reaction,
F is Faraday's Constant, and
Q is the reaction quotient (Snoeyink and Jenkins, 1980).

The Eh values computed from the two redox couples would be equal if the water were in a state of equilibrium. Eh values computed from the ferrous/ferric iron couple, assuming equilibrium with aqueous iron species only, ranged from 246 mv to 505 mv and averaged 390 mv; Eh values computed from dissolved oxygen ranged from 838 mv to 940 mv and averaged 810 mv. This disparity indicates that the waters were in a state of redox disequilibrium with respect to iron and oxygen.

The Eh values for iron and oxygen are plotted on Eh-pH diagrams in figure 6. Figure 6A represents water in equilibrium with dissolved $Fe(OH)_3$, and figure 6B represents water in equilibrium with solid $Fe(OH)_3$, assuming no other important iron compounds in the system. The boundary of the $Fe(OH)_{3\ (solid)}$ field was calculated from total iron concentrations of 0.15 and 2.0 mg/L, the approximate range of total iron concentrations in water samples. The theoretical Eh values for iron fall within the ferrous iron predominance field in figure 6A and along the boundary between the Fe^{2+} and $Fe(OH)_{3\ (s)}$-predominance fields in figure 6B. Eh values for dissolved oxygen are within the $Fe(OH)_2^+$-predominance field in figure 6A and within the $Fe(OH)_{3\ (s)}$-predominance field in figure 6B.

The disequilibrium between iron and oxygen indicates that oxygenated water is mixing with water that contains ferrous iron near the well screen. The mixing could be occurring within the well casing in response to the input of waters with differing geochemical characteristics along the length of the screen (screen lengths range from 30 to 90 ft), or it could be occurring within the aquifer in response to the local drawdown of oxygenated water around the pumping well. The ferrous- and ferric-iron data indicate that no significant ferrous-iron oxidation had occurred before sampling and that significant ferrous-iron oxidation probably does not occur in the aquifer or well casing while the wells are in operation.

Iron-Oxidation Kinetics

The rate of oxidation of ferrous iron in ground water is strongly dependent on pH and dissolved-oxygen concentration. The rate can be estimated from the following equation:

$$t_{1/2} = 0.693/(k\ pO_2\ [OH^-]^2), \quad (9)$$

where

$t_{1/2}$ is the half-time of iron oxidation,
pO_2 is the partial pressure of oxygen,
$[OH^-]$ is the concentration of hydroxyl ions, and
k is the rate constant (Davidson and Seed, 1983).

Table 5. Ferrous- and ferric-iron concentrations, pH, dissolved-oxygen concentrations, and calculated redox potentials for water samples from selected wells in Suffolk County, N.Y.
[Iron concentrations in micrograms per liter; redox potentials (Eh) in millivolts. Fe(T), total iron; Fe_s, iron in solid state; Fe_{aq}, iron in aqueous state; standard electrode potentials (E_0) from Snoeyink and Jenkins (1980). Well locations are shown in fig. 4]

Well	Fe(T)	Fe(II)	Fe(III)	Fe(II)/Fe(T) ratio (percent)	pH	Dissolved oxygen	Eh[1]		
							(O_2)	(Fe_s)	(Fe_{aq})
AR-4	479	459	20	95.8	5.16	0.5	896	438	498
EFR-4	389	355	34	91.3	5.79	1.1	863	333	444
FA-32	152	139	13	91.4	5.19	.8	897	463	514
FA-33	346	171	175	49.4	5.36	1.2	890	428	556
GA-2	682	644	38	94.4	5.98	.8	850	284	409
GA-8	381	311	70	81.6	4.84	.1	904	505	519
GBP-1	461	431	30	93.5	5.94	.8	853	301	418
HL-1	1,484	1,447	37	97.5	5.25	.5	890	392	474
HL-3	2,162	2,150	12	99.4	4.66	4.7	940	487	505
LR-1	173	155	18	89.6	5.38	1.5	890	427	509
LR-2	280	254	26	90.7	4.93	.8	912	494	547
LSR-5	280	264	16	94.3	6.07	2.6	852	290	399
MD-1	749	703	46	93.9	6.18	.8	838	246	388
MPSB-1	689	657	32	95.4	6.15	4.2	851	253	384
NYA-5	1,107	1,071	36	96.7	5.98	1.2	853	270	394
SA-3	357	326	31	91.3	5.08	.2	895	461	528
TA-2	399	371	28	93.0	5.41	.6	882	399	481
TS-4	843	809	34	96.0	5.13	.8	901	429	501

[1] Eh(O_2) is Eh calculated from oxygen/water couple.

Eh (Fe_s) is Eh calculated from ferrous/ferric iron couple in equilibrium with $Fe(OH)_{3\ (solid)}$.

Eh (Fe_{aq}) is Eh calculated from ferrous/ferric iron couple.

The equation shows that the rate of iron oxidation is first order with respect to dissolved-oxygen concentration and second order with respect to pH. Thus, iron oxidation would be expected to occur fastest at high pH values and high dissolved-oxygen concentrations, but pH has the greater effect. The presence of ferric hydroxide has an autocatalytic effect on ferrous iron oxidation (Tamura and others, 1976b). The rate of ferrous-iron oxidation also increases significantly in the presence of trace amounts of copper and anions that form stable complexes with ferric iron (Stumm and Morgan, 1981) and increases in the presence of fluoride and phosphate (Tamura and others, 1976a). The oxidation rate decreases in the presence of high concentrations of sulfate, chloride, nitrate, bromide, and iodide, presumably through the complexation of free ferrous iron. Finally, Sung and Morgan (1980) found that the rate constant decreased with increasing ionic strength. The rate constant is independent of temperature in the range of 5˚C to 35˚C (Stumm and Lee, 1961). Natural organic matter, such as fulvic acid, also could increase the rate of ferrous-iron oxidation (Stumm and Morgan, 1981). Iron bacteria also catalyze the oxidation of ferrous iron, and biological oxidation of ferrous iron can be significant in low-pH waters (Ghiorse, 1984).

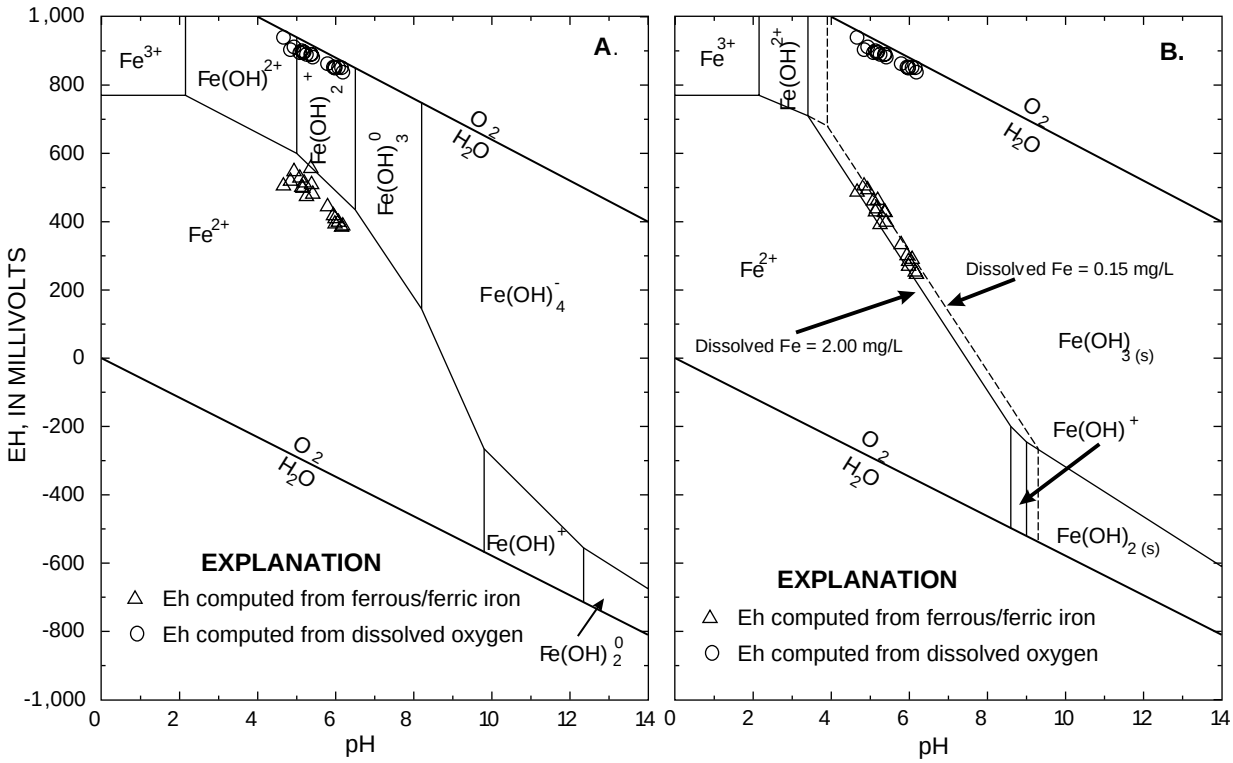

Figure 6. Distribution of Eh values computed from concentrations of ferrous and ferric iron and dissolved oxygen: A. In water containing only aqueous iron species. B. In water in equilibrium with solid ferric hydroxide ($Fe(OH)_{3(s)}$).

Ferrous-iron oxidation rates in well water and treated water were estimated through equation 9; results are presented in table 6. A rate constant of 8.0×10^{13} l^2/min•atm•mol^2 was used in the calculations (Stumm and Morgan, 1981). The half-time of oxidation in well water ranged from 2 min (0.001 days) to 2,700 days and had a median of 4.2 days. Oxidation of ferrous iron in the wells is extremely slow because the rate is strongly dependent on the ground-water pH, which is naturally low; if the pH of well water were increased to 7, the median half-time of oxidation would decrease to 146 min (0.1 days) (table 6). The low ambient pH of the ground water explains the origin of the redox disequilibrium observed in the ground water and the lack of significant iron oxidation in ground water sampled at the well head despite the presence of dissolved oxygen. The rate of ferrous-iron oxidation is too slow for precipitation to occur while the wells are in operation. The residence time of water in a well with a depth of 55 to 830 ft, a diameter of 24 in., and a pumping rate of 1,000 gal/min—1.3 to 19.5 min—would be much too brief for significant encrustation to develop.

A study of iron-related well-screen encrustation in the Missouri River alluvial aquifer found that the half-time of oxidation averaged about 40 min and that about 1 half-time of oxidation had occurred before the time of sampling at the well head (Walter, 1989). The ground-water pH, which was buffered by carbonate bedrock, averaged about 7.5. This relatively high pH resulted in rapid oxidation of ferrous iron while the wells were operating and allowed ferric iron to precipitate around the well screen. In contrast, the low pH and slow oxidation rate in the aquifer system of Suffolk County suggest that most of the well-screen encrustation occurs while the wells are not operating.

Effects of Lubrication Water during Well Shutdowns

During periods of low water demand, wells either run on a timed cycle or are controlled by pressure transducers and are frequently shut down. During these idle periods, water from the distribution system is discharged into the well to keep pump turbines wet; this water, which is referred to as “lubrication water,” is a mixture of ambient ground water and treated water. During periods of frequent well shutdowns, the water

Table 6. Iron-oxidation rates in well water and treated water at selected wells, Suffolk County, N.Y.

[Half-time of ferrous-iron oxidation calculated from pH and dissolved-oxygen concentration; --, no data available. Well locations are shown in fig. 4]

Well	Half-time of oxidation				
	Well water (days)	Well water (pH=7)		Treated water	
		Minutes	Days	Minutes	Days
AA-3	.001	150	0.10	--	--
AR-4	120	300	.21	--	--
BA-1	.14	16	.01	--	--
BA-3	3.3	50	.04	24	.16
BP-1	18	15	.10	--	--
CD-4	3.9	15	.01	--	--
CSB-1	.18	25	.02	.95	.007
EFR-4	3.3	140	.10	--	--
FA-32	68	190	.13	190	.13
FA-33	21	120	.09	.5	>.001
GA-2	1.9	200	.14	3,000	2.1
GA-8	2,700	1,500	1.0	4.8	.003
GBP-1	2.1	190	.13	.4	>.001
GBP-3	2.5	250	.17	--	--
HL-1	82	300	.21	7.4	.005
HL-3	130	32	.02	16	.011
LR-1	16	100	.07	24	.017
LR-2	220	190	.13	1.1	.001
LSR-5	.37	59	.04	14.	.01
LSR-6	5.4	390	.27	--	--
MD-1	.74	190	.14	--	--
MPSB-1	.16	36	.02	--	--
NYA-5	1.2	120	.09	--	--
NR-2	1.6	17	.01	--	--
PA-1	.60	16	.01	--	--
RRA-1	4.5	750	.52	--	--
SA-3	2,700	1,500	1.0	4.8	.003
SR-1	8.4	15	.01	27	.02
TA-2	33	250	.17	390	.27
TS-4	89	190	.13	1.9	.001

used for pump lubrication consists mainly of ground water that is treated with lime ($CaCO_3$) to increase the pH and minimize corrosion in the distribution system. Treated water has a higher pH and contains more dissolved oxygen than the untreated ground water remaining in the static water column in the wells. The introduction of treated water into an idle well could raise the pH and allow ferrous iron to oxidize and ferric hydroxide to precipitate within the well. A discharge rate of 5 gal/min would displace the entire static water column in an 800-ft-deep well in less than 2 days.

Water samples collected from the local distribution systems of 16 iron-encrusted wells indicate that treated water had a higher pH and higher dissolved-oxygen concentration than untreated water; pH values ranged from 6.5 to 8.5 and had a median of 7.5, and dissolved-oxygen concentrations ranged from 0.4 mg/L to 11.2 mg/L and had a median of 6.0 mg/L. Saturation indices for manganese oxyhydroxide mineral phases were less than zero, indicating that those mineral phases were not thermodynamically favored to precipitate. Saturation indices for selected iron oxyhydroxides in the treated water were significantly higher than those for the ambient ground water and indicate that the treated lubrication water was supersaturated with respect to ferric hydroxide, goethite, hematite, maghemite, and magnetite (fig. 7A). Comparison of saturation indices for ferric hydroxide in well water and treated water shows that the treated water was supersaturated with respect to ferric hydroxide at 13 of 16 wells (fig. 7A), whereas ground water was undersaturated with respect to ferric hydroxide in all wells (fig. 7B). The difference in saturation indices is due to the higher pH of the treated water. Chemical and mineralogical data show ferric hydroxide to be the predominant mineral phase in the encrusting material; this indicates that a significant amount of the encrusting material precipitates while the wells are idle, and that the use of treated water to keep pump turbines wet during idle periods can significantly accelerate the encrustation process.

The elevated pH of treated water increases the rate at which ferrous iron oxidizes and ferric hydroxide precipitates in the well water because the iron-oxidation rate is strongly pH-dependent. The half-time of oxidation was computed for treated-water samples through equation 9; the results (table 6) ranged from 0.4 min to 3,000 min (2.1 days), and the median half-time was 12 min (0.008 days). In contrast, the median half-time of oxidation in ground-water samples was 4.2 days (table 6). These data corroborate the previously discussed evidence that the discharge of treated water into the static water column accelerates the rate at which ferrous iron is oxidized and ferric iron precipitates in the well water. The diffusion of oxygen from the head space above the water

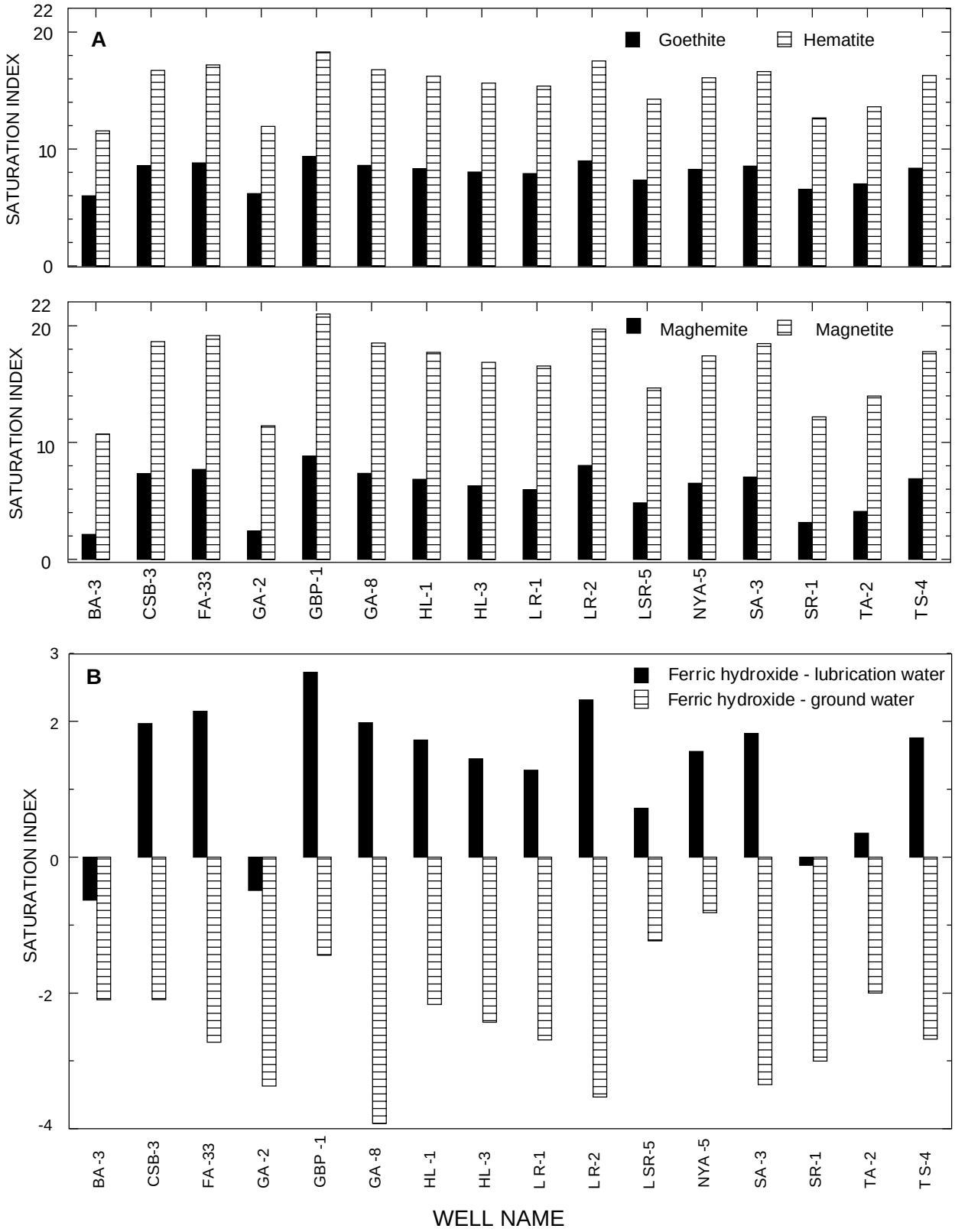

Figure 7. Saturation indices calculated by the chemical equilibrium model WATEQF for 16 well sites in Suffolk County, N.Y.: A. Iron compounds in treated lubrication water (goethite, hematite, maghemite, and magnetite). B. Ferric hydroxide in well water and treated lubrication water.

column, and the introduction of treated water, which has higher dissolved-oxygen concentrations than the ground water, would also accelerate the rate of iron oxidation.

The use of treated water from nearby wells that contain high concentrations of dissolved iron also could cause well-screen encrustation in wells that are screened in areas with little or no dissolved iron. Many wells within the SCWA well network have undergone large declines in specific capacity even though iron concentrations were below detection limits while the wells were in operation. For example, well CSB-1 (fig. 4) has had a history of biofouling even though the iron concentration in the well is below 0.01 mg/L. The use of treated water from nearby well CSB-3, which contained 0.66 mg/L dissolved iron, could have introduced iron into the well during idle periods and would explain the observed specific-capacity declines. The specific-capacity declines in wells screened in low-iron, oxygenated environments, particularly in shallow parts of the upper glacial aquifer, could be due to the introduction of dissolved iron into the well by treated lubrication water.

MICROBIOLOGY

The specific-capacity declines seen in some SCWA wells are caused partly by the growth of iron-oxidizing bacteria on or near the well screens. Detailed descriptions of the taxonomy and ecology of iron-oxidizing bacteria are given in Pringsheim (1949) and Ghiorse (1984). Iron-oxidizing bacteria are aerobic bacteria that either catalyze the oxidation of ferrous iron or alter the water chemistry such that conditions are favorable for iron oxidation (Ghiorse, 1984).

Well-screen encrustation and aquifer biofouling occur when filamentous iron bacteria produce large extracellular filaments during growth that become coated with iron oxyhydroxides; this combination of ferric oxyhydroxides and extracellular material can clog not only well screens, but pore spaces in the surrounding formation (Ghiorse, 1986). The growth of iron-oxidizing bacteria on well screens and in the surrounding aquifer begins when nonfilamentous iron-related bacteria, such as *Pseudomonas*, colonize the surfaces of well screens and sediment grains; this organic substrate allows filamentous iron bacteria to form extensive biofilms (George Alford, oral commun., 1991). Iron-oxidizing bacteria, which require ferrous iron and dissolved oxygen, are found naturally in the vicinity of Eh gradients, such as ground-water seeps and wetlands (Pringsheim, 1949). Conditions favorable for their growth also are produced where drawdowns induce the mixing of oxygenated water and high-iron water near a pumping well screened in reduced, high-iron environments.

Iron-oxidizing bacteria include both heterotrophic and chemoautotrophic organisms. Heterotrophic iron bacteria require organic carbon as a carbon source; instead of deriving metabolic energy from the oxidation of iron, they may enzymatically catalyze the oxidation of ferrous iron or otherwise cause the reaction by creating high-pH and high-Eh microenvironments (Ghiorse, 1984). Heterotrophic iron bacteria that are important biofouling agents include the filamentous bacteria genera *Leptothrix*, *Crenothrix*, *Clonothrix*, and *Spaerotilus* and the nonfilamentous genus *Siderocapsa*. Iron bacteria of the genus *Leptothrix* can be facultative autotrophs that use iron oxidation as an energy source under certain conditions (Pringsheim, 1949). Chemoautotrophic iron bacteria derive metabolic energy from the oxidation of ferrous iron; they catalyze the oxidation of ferrous iron and can use CO_2 as a carbon source. A species of chemoautotrophic bacteria that is commonly involved in the well-screen encrustation and biofouling process is *Gallionella ferruginea* (Halbeck and Pederson, 1986). The presence of *Gallionella*, an oligotrophic organism that can grow in low carbon environments but requires an organic substrate, is indicative of a mature biofilm. *Gallionella* is microaerophilic (requires oxygen but prefers environments with low dissolved-oxygen concentrations). The species produces long extracellular filaments that are nonliving and typically coated with inorganic iron oxyhydroxides (Ghiorse, 1986). *Gallionella* prefers low dissolved-oxygen concentrations (0.1 to 1.0 mg/L), near-neutral pH (6.0 to 7.6), and redox potentials between 200 and 320 mv; it also prefers temperatures between 4 and 25.6˚C and iron concentrations between 1 and 10 mg/L (Driscoll, 1986). A typical biofilm consists of a consortium of organisms that represent several species of iron bacteria. Iron bacteria can also catalyze the oxidation of manganese; manganese-related biofouling can occur in high-pH and high-manganese environments, although this seems to be insignificant in Suffolk County.

Occurrence of Iron-Related Bacteria

Biofilm samples from 43 SCWA wells were collected and assayed by light microscopy to identify the species of filamentous iron bacteria involved in the well-screen-encrustation process; results are shown in table 7. Filamentous iron bacteria were observed in samples from 31 of the 43 wells. *Leptothrix ocracea* was observed in 8 samples, and *Leptothrix discophora* was observed in 9 samples. *Leptothrix* are highly aerobic and produce long bacterial colonies encased in a sheath of extracellular material. *Crenothrix polyspora* was observed in 6 biofilm samples; this species forms large, branching colonies. The common nonfilamentous iron bacteria, *Siderocapsa,* is difficult to identify with a light microscope and could have been present in the biofilm samples.

Gallionella ferruginea was observed in 29 of the 43 biofilm samples and was the predominant filamentous organism in all 29 samples; this indicates that *Gallionella f.* is the principal species in most of the biofilm samples and is a major biofouling agent in Suffolk County. *Gallionella f.* was observed in wells representing a wide range of water conditions; dissolved-iron concentration in wells containing *Gallionella* ranged from <0.01 to 2.15 mg/L; field pH ranged from 4.7 to 6.4, and dissolved-oxygen concentrations ranged from 0.1 to 10.0 mg/L.

Dissolved-iron- and oxygen-concentration data from wells containing *Gallionella f.* were compared with those from wells containing no *Gallionella f.* (fig. 8). The data indicate that water from wells containing *Gallionella* was statistically lower in dissolved oxygen and higher in dissolved iron than water elsewhere, although the iron concentrations in these wells were generally near the lower limit of the species' preferred range; the p-values for both iron and oxygen were less than 0.001. The median concentrations of dissolved iron and dissolved oxygen in wells that contained *Gallionella f.* were 0.35 mg/L and 0.8 mg/L, respectively, in contrast to median concentrations of <0.03 mg/L and 9.6 mg/L, respectively, in wells without *Gallionella f.* The data also indicate that *Gallionella f.* is more common in the Magothy aquifer than in the upper glacial aquifer; *Gallionella f.* was observed in 23 of the 25 biofilm samples collected from the Magothy aquifer and 5 of the 19 biofilm samples collected from the upper glacial aquifer. The affinity of *Gallionella f.* for waters with low dissolved-oxygen concentration and elevated dissolved-iron

Table 7. Detection of filamentous iron bacteria in selected Suffolk County Water Authority wells, Suffolk County, N.Y.

[X indicates detection. Well locations are shown in fig. 4]

	Genus and species				
Well	*Gallionella ferruginea*	*Crenothrix polyspora*	*Leptothrix ocracea*	*Leptothrix discophora*	**No bacteria**
AA-3	X		X	X	
AR-4	X				
BA-1	X			X	
BA-3	X				
BCR-1					X
BHR-2					X
BHR-5					X
BPR-1					X
BPR-4					X
BR-1					X
BR-2					X
CD-4	X		X		
CSB-1	X		X		
CSH-1	X	X	X	X	
DS-2					X
EFR-4	X				
FA-32	X				
FA-33	X				
GA-2	X			X	
GA-8	X		X	X	
GBP-3	X		X		
HL-1	X				
HL-3	X				
LA-1		X		X	
LR-1	X				
LR-2	X				
LSR-3		X			
LSR-5	X				
LSR-6	X				
MD-1	X	X		X	
MPSB-1	X				
NR-2					X
NYA-5	X				
PA-1	X				
PA-2	X		X	X	
RA-1					X
RRA-1					X
SA-3	X	X			
SL-2					X
SR-1	X				
TA-2	X				
TS-4	X				
WR-4	X	X	X	X	
Total	29	6	8	9	12

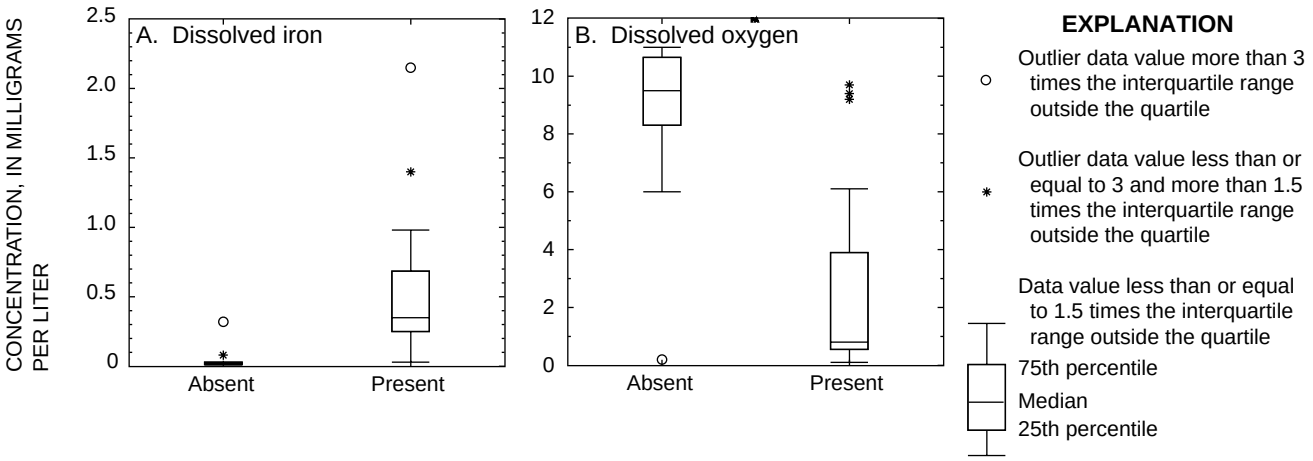

Figure 8. (A) Dissolved iron, and (B) dissolved oxygen in wells in Suffolk County, N.Y., that contained the iron bacteria *Gallionella ferruginea* and in wells that did not.

concentration explains the predominance of this species in biofilm samples from the Magothy aquifer. No statistical difference in pH was found between wells that contained *Gallionella f.* and wells that did not.

The presence of *Gallionella* in wells completed in the upper glacial aquifer, which has high dissolved-oxygen concentrations and low dissolved-iron concentrations, is consistent with the use of high-iron ground water as lubrication water. For example, the biofilm sample from well CSB-1 (fig. 4) consisted almost entirely of iron-encrusted *Gallionella* filaments even though the well water contained less than 0.01 mg/L dissolved iron. The growth of *Gallionella* in this well could be attributed to the use of lubrication water from well CSB-3 at the same site, which is screened in the Magothy aquifer; water from this well contained 0.66 mg/L dissolved iron. In addition, *Gallionella* prefers water with pH of 6.0 to 7.6 and, therefore, may not proliferate when the wells are in operation and the water pH is below 6.0. Although water from well CSB-1 contained 0.26 mg/L dissolved manganese, energy-dispersion analysis revealed no manganese in the encrusting material; this is further evidence that iron bacteria do not use appreciable amounts of manganese from the ground water and that manganese does not play an important role in the biofouling process in Suffolk County, even where the water contains dissolved manganese and little dissolved iron.

The Starkey (1945) method for estimating the percentage (by weight) of organic carbon in iron bacteria biofilms from the thermodynamics of ferrous iron oxidation and cell synthesis was modified to fit the mineralogy of the encrusting material and used to estimate the percentage of cellular carbon associated with *Gallionella* in a typical biofilm. The mass ratio of iron oxyhydroxides to cellular carbon ranged from 350:1 for pure $Fe(OH)_3$ to 890:1 for pure FeOOH; this shows that *Gallionella* produces between 350 and 890 grams of iron for every gram of cellular material that it produces. The low amount of energy yielded from the oxidation of ferrous iron requires the production of large quantities of ferric oxyhydroxide during the growth of the biofilm; thus, the ability of *Gallionella* to catalyze the oxidation of large amounts of ferric hydroxide makes the species an effective biofouling agent.

Source of Iron-Related Bacteria

Core samples from several locations within the aquifer system were tested to determine whether iron bacteria associated with well-screen encrustation had

been introduced during drilling of the wells or were naturally present in the aquifer sediments. Recent evidence suggests that iron bacteria can shed extracellular material and can be transported for great distances in a ground-water system (Cullimore, 1986). Iron bacteria are ubiquitous and occur in a wide range of environments (Ghiorse, 1986); iron bacteria in the soil zone could be introduced into the ground-water system at the water table through areal recharge and could be transported deeper into the system through advection. Iron bacteria are transported through the ground-water system as small (0.1-µm to 0.3-µm diameter), inactive cells that attach themselves to surfaces and begin to grow when conditions become favorable, such as in the vicinity of a well screen (Cullimore, 1986).

A total of 35 sediment composites, each consisting of material from 1 to 14 cores from the same borehole, were tested for the presence of iron bacteria; iron-related reactions were observed in all 35 composites (table 8). The type of reaction observed in the bacteria test in all composites was consistent with iron-related bacteria (George Alford, oral commun., 1991). Analysis of the biofilms grown in the reaction test revealed numerous rods and filaments associated with iron-oxidizing bacteria and iron oxyhydroxide precipitates, suggesting the presence of iron-oxidizing bacteria in all composites. Positive identification of *Gallionella* was difficult, however, and the presence of the species could not be confirmed in the cultured biofilm samples (table 8) because large *Gallionella* filaments were lacking in the cultures; this could be due to the depletion of the iron medium before the filaments could grow to observable size (George Alford, oral commun., 1991).

Iron-related bacteria also were detected in all nine samples of circulation water collected during the drilling of the replacement wells and test borings. The use of water from iron-encrusted wells for this purpose would explain the presence of iron bacteria in the circulation water, which comes from the local distribution system. *Gallionella f.* was observed in biofilm samples collected from within the SCWA distribution system at 12 locations, indicating that *Gallionella* is likely to be present in all distribution and circulation water.

The reaction time of biofilm growth observed in the tests is an inverse function of the number of iron bacteria in the sample; reaction times decrease as the number of bacteria increases. Comparison of reaction times for the composite sediment samples with those for the circulation water showed that reaction times in 26 of the 35 composites were shorter than in circulation water. The shorter reaction time in the cores than in circulation water indicates that the cores were not contaminated by the drilling fluid and that iron-related bacteria occurred naturally in the aquifer sediments; results of tests in which reaction times in the cores were equal to or larger than reaction times in the water were inconclusive (table 8).

Role of Lignite in Well-Screen Encrustation

The Magothy aquifer contains lignite, which can be disseminated within the aquifer sediments and also can be present as discontinuous beds and lenses (Smolensky and others, 1989). Lignite can act as a carbon source for heterotrophic iron bacteria. Lignin forms the base of microbial food chains in ground-water systems, and the oxidation of organic carbon, coupled with the reduction of iron and sulfate, can be the source of CO_2 in deep ground-water systems (Chapelle, 1993).

The presence of lignite near a well screen can accelerate the formation of iron-bacteria biofilms. Video logging of replacement well RW3 (Prospect Ave. 3) (fig. 4) in October 1991 provided direct observation of a biofilm. The well was installed in June 1991 and had not been operated prior to the video log. The well was screened in the Magothy aquifer from 213 to 249 ft below land surface. The well screen was heavily encrusted with an iron biofilm from 230 to 236 ft below land surface; suspended material was observed in the water column throughout the interval from 226 to 236 ft below land surface. The 17 ft above and the 13 ft below the biofouled zone showed little or no iron-biofilm growth, and the water was not turbid.

A gamma log and geologic log from the well are shown in figure 9. The zone of biofouling is adjacent to a bed of lignite that lies between 225 and 228 ft below land surface. A core collected at 234 ft below land surface, which is near the biofouled zone, consisted of coarse, gray sand. The growth of the biofilm near the lignite is further evidence that lignite is a carbon source for heterotrophic iron bacteria such as *Pseudomonas*. The growth of *Pseudomonas* forms the organic substrate necessary for the growth of filamentous iron bacteria such as *Gallionella* and allows the bacteria to colonize the surface of the well screen.

The iron biofilm was removed when the well was reconditioned in December 1991. Bail samples of

Table 8. Detection of iron-related bacteria in aquifer sediments and in drilling water, Suffolk County, N.Y.
[<, less than; >, greater than. Well locations are shown in fig. 4]

Location	Type of well	Abbreviation	Depth interval (feet below land surface)	Number of cores	Reaction time (days)		Detection
					Cores	Water	
Brook Avenue	replacement	RW5	305-325	4	10	7	inconclusive
			335-355	3	>21		inconclusive
			365-375	2	15		inconclusive
			385-405	3	15		inconclusive
			415-435	3	>21		inconclusive
			445	1	15		inconclusive
			455-465	1	7		inconclusive
East Hampton Airport	test boring	TB1	160-240	5	4	7	present
			260-340	5	4		present
			360-440	5	4		present
			460-540	5	4		present
Jennings Road	test boring	TB2	260-280	2	<3	5	present
			300-320	2	<3		present
			340-360	2	<3		present
Montauk Point	replacement	RW2	240-254	3	3	5	present
Oakview Highway	replacement	RW4	120-140	5	<3	5	present
			150-165	4	<3		present
			170-190	4	<3		present
Pleasant Avenue	replacement	RW7	645-665	3	10	7	inconclusive
			675-695	3	7		inconclusive
			705-725	3	<3		present
			735-755	3	<3		present
			765-795	3	4		present
Sy Court	replacement	RW6	280-320	3	<3	7	present
			340-380	3	<3		present
			400-440	3	6		present
			460-500	3	5		present
			580-620	3	<3		present
			640-660	2	5		present
West Prospect Street	replacement	RW3	184-202	3	<3	7	present
			212-234	3	<3		present
			242-262	3	<3		present
			272-282	2	<3		present
			292-302	2	<3		present
Wyandanch Avenue	replacement	RW1	507-637	14	4	5	present

ground water and biofilm were collected in March 1992 from three depth intervals within the screen zone—the biofouled zone and the zones directly above and below it (biofilm samples were filtered from bail samples). The water samples were analyzed for gross aerobic bacteria populations and dissolved organic carbon; the biofilm sample was assayed by light microscopy. Results of the analyses (fig. 9) indicate that the dissolved-organic-carbon concentration was slightly higher in the biofouled zone (1.2 mg/L) than in the zones above or below (1.0 and 1.1 mg/L, respectively); all three values were within experimental error.

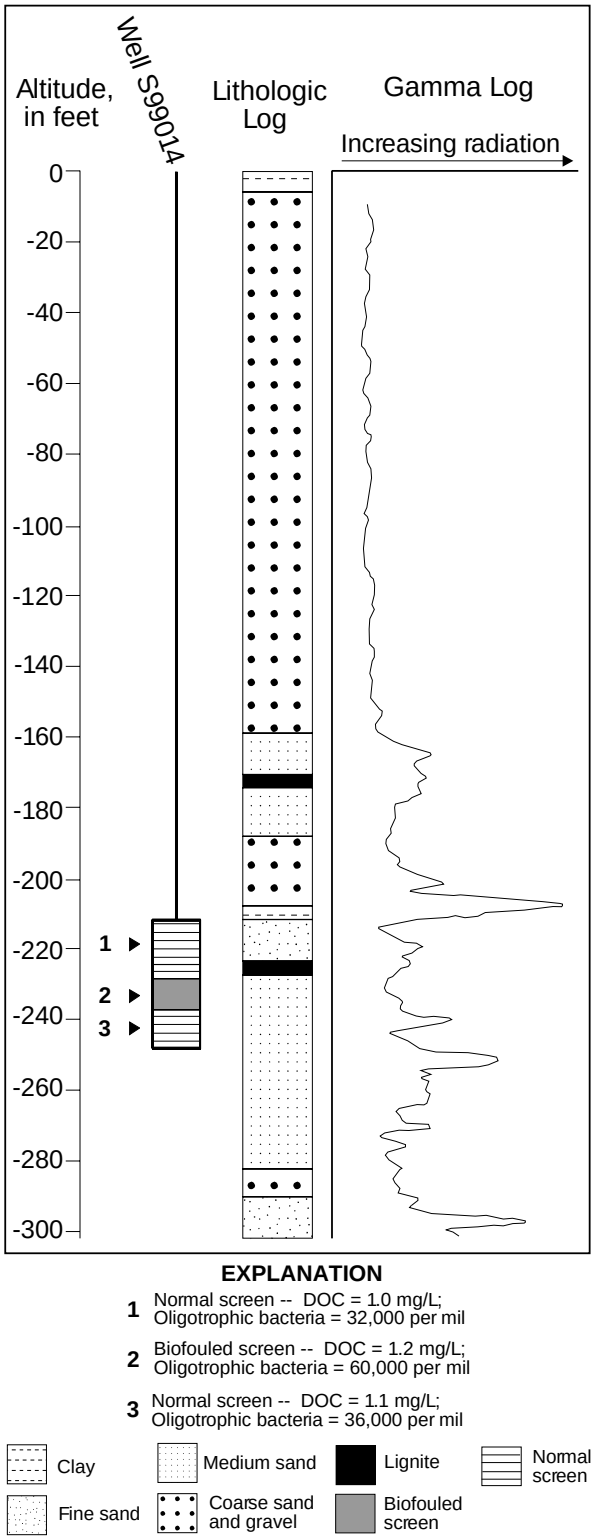

Figure 9. Lithologic and gamma logs showing concentrations of dissolved organic carbon (DOC) and oligotrophic bacteria, and location of iron-bacteria biofilm growth at well S99014, Suffolk County, N.Y. (Location is shown in fig. 4.)

Differences among the three zones could have been diminished through vertical mixing within the open water column. Iron-reducing bacteria and nonspecific heterotroph counts in water from all three zones exceeded 10^5 and 10^6 per mil, respectively. Oligotrophic bacteria counts, which included *Gallionella*, were 32,000 per mil above the biofouled zone and 36,000 per mil below the biofouled zone; those within the biofouled zone were 60,000 per mil (Alfred Mikell, University of Mississippi-Oxford, written commun., 1992). The data indicate a larger population of oligotrophic bacteria near the lignite bed than in the intervals above or below it. Analysis of the biofilm sample showed that the sample consisted mainly of *Gallionell*a filaments; this indicates that a mature *Gallionella* biofilm was actively growing on the idle well screen 3 months after the well was reconditioned. These data are further evidence that the presence of lignite can accelerate the growth of iron biofilms on the screens of idle wells and that highly developed iron-bacteria biofilms can develop in a well within a few months.

Role of Sulfur-Reducing Bacteria

Sulfate reduction and the precipitation of iron sulfides could play a role in the encrustation process in some localities. Pyrite forms naturally within the Magothy aquifer where conditions are sufficiently reducing, and the bacterially mediated reduction of sulfate to sulfide and the precipitation of iron sulfide on well screens could contribute to the encrustation process. Four cores composited from a test boring at East Hampton Airport (TB1, fig. 4) were tested for the presence of sulfur-reducing bacteria by sulfur-specific Biological Activity Reaction Tests; results indicated that sulfate-reducing bacteria were present in all four composite samples.

Black material containing sulfide was collected from six biofouled wells; coatings of oxidized iron typically formed on the black material within 1 hour of sample collection. X-ray-diffraction analysis of black material from well HL-1 (fig. 4) indicated the material to be amorphous, and energy-dispersion analysis found it to consist primarily of iron and sulfur with minor amounts of copper and phosphorus, indicating that sulfate reduction could contribute to well-screen encrustation in some locations. The iron-sulfide phase was associated with material consisting of ferric oxyhydroxides in most samples; this suggests that redox

conditions within the biofilms can vary significantly on a small scale. For example, iron sulfide formed in biofilm material from well GA-2 (fig. 4) after part of the biofilm was allowed to become anaerobic. The biofilm material, which initially consisted of ferric oxyhydroxide, was placed in a sterilized clear tube with triple-distilled water and then air sealed. After about 2 days, a black precipitate consisting of iron sulfide had formed in the bottom of the tube. This is attributed to the Eh gradient that resulted from dissolved oxygen in the head space—the water near the top of the tube was aerobic, whereas the sediment biofilm at the bottom became anaerobic. A film of *Leptothrix discophora* was observed at the air/water interface, and a *Gallionella* biofilm formed at the sediment/water interface. Iron sulfide precipitated at the bottom of the tube, where conditions were favorable for anaerobic-iron and sulfate-reducing bacteria. Eh gradients that develop by oxygen diffusion within a biofilm can lead to the formation of microredox environments that would allow ferric oxyhydroxides and iron sulfide to be present in the same biofilm (Roy Cullimore, University of Saskatchewan, oral commun., 1991). Microaerophilic iron-oxidizing bacteria, such as *Gallionella*, might depend on sulfate-reducing bacteria to maintain low-oxygen environments within a biofilm (Ghiorse, 1984).

Effects of Iron-Oxidizing Bacteria on Water Quality

The ability of ferric hydroxide to effectively sorb metals, anions, and organic substances (Hem, 1985) indicates that biofilms in a well could significantly alter the water quality. Water samples were collected before and after the reconditioning of 26 iron-encrusted wells to assess the effect of biofilms on well-water quality. Water samples collected before well reconditioning were assumed to represent water affected by the iron biofilm, whereas samples collected after reconditioning were assumed to be more representative of the quality of water not exposed to a biofilm. Sufficient volumes of water were removed after reconditioning to ensure that no residual effects of the reconditioning procedure remained. Differences between the two samples from each well were attributed to the effects of the biofilm. Statistical differences between the two data sets were determined by the Wilcox Signed Rank Test (Helsel and Hirsch, 1992); this technique tests whether the difference between two paired data sets is normally distributed around zero. A p-value of less than 0.05 indicates a statistical difference in the data at the 95-percent confidence level. Where no statistical trends were observed, differences in the two data sets may be relatively small, or trends could be obscured by natural fluctuations in background water quality.

Iron and Manganese

The differences between concentrations of total iron and dissolved iron in water samples collected before and after reconditioning are plotted in figure 10A and 10B. The differences represent the initial concentration minus the final concentration; thus, points that plot above zero indicate an increase in concentration after reconditioning. The plots indicate that dissolved-iron concentrations increased after reconditioning in 13 of the 22 wells, decreased in 2, and were unaltered in 7, and that total iron concentration increased after reconditioning in 18 of 24 wells, decreased in 4, and were unaltered in 3. The p-values of 0.004 and 0.013, for dissolved and total iron, respectively, indicate that iron concentrations were statistically higher after reconditioning and removal of the biofilm. The increases in dissolved-iron concentration ranged from <0.03 to 1.5 mg/L and averaged 0.15 mg/L, and the increases in total iron ranged from <0.03 to 0.7 mg/L and averaged 0.2 mg/L. The results suggest that an existing iron biofilm can effectively remove iron from solution to form iron oxyhydroxides and would thereby decrease iron concentrations in the well water. The data also suggest that an iron biofilm could increase the rate of iron oxidation and precipitation; if the rates were the same after reconditioning as before, iron concentrations would not be expected to have changed significantly. The presence of ferric hydroxide around the well screen also could increase iron-oxidation rates because ferric hydroxide has an autocatalytic effect on ferrous-iron oxidation (Tamura and others, 1976b). The removal of dissolved iron by iron-bacteria biofilms suggests that some iron oxyhydroxide mineral phases could precipitate while the wells are in operation; equilibrium modeling showed well waters to be supersaturated with respect to goethite and hematite. Water-chemistry and solid-phase-chemistry data indicate that most encrusting material is deposited when the wells are shut down.

Total and dissolved manganese concentrations before and after well reconditioning are plotted in figure 10C and 10D. Total manganese concentration increased in 11 of 25 wells, decreased in 4 wells, and was unchanged in 10 wells. Dissolved-manganese concentration increased in 8 of 22 wells, decreased in 6, and was unchanged in 8. Dissolved-manganese increases ranged from <0.01 to 0.04 mg/L, and total manganese increases ranged from <0.01 to 0.05 mg/L. The p-values for dissolved and total manganese were 0.028 and 0.025, respectively. These data indicate that dissolved and total manganese concentrations were statistically higher after reconditioning than before, and that manganese, therefore, like iron, is removed by an iron biofilm. This is consistent with the presence of manganese in the encrusting material.

Dissolved Oxygen, pH, and Dissolved Carbon

Differences between dissolved-oxygen concentrations before and after reconditioning were not statistically significant (p-value 0.100) and indicate that biofilms may not significantly affect the dissolved-oxygen concentration of the ground water or that the differences are obscured by natural fluctuations.

pH values before and after reconditioning are plotted in figure 10E. pH decreased in samples from 22 of 24 wells and increased (negligibly) in samples from 2 wells. pH decreases ranged from 0.07 to 1.04 pH units. The p-value of less than 0.0001 indicates that pH was statistically lower after reconditioning than before and that iron biofilms increase the pH of well water. The pH increases could be due to the consumption of organic acids by the consortia of biofilm organisms (Driscoll, 1986); also, the presence of chemoautotrophic iron bacteria, such as *Gallionella*, could increase the pH by fixing inorganic carbon as a carbon source and removing CO_2 from solution.

Field-measured alkalinity (as $CaCO_3$) showed no significant change after reconditioning, and the p-value of 0.25 indicates that the biofilm does not significantly affect the alkalinity of the well water. The partial pressure of CO_2 was calculated from field alkalinity and pH data, and the resulting p-value of 0.30 indicates no statistical change in pCO_2 after reconditioning. Thus, the removal of CO_2 by chemoautotrophic bacteria may not adequately explain the observed pH decreases that followed reconditioning. Dissolved-organic-carbon concentration increased in 5 of 25 wells and decreased in 6 wells after reconditioning; the p-value of 0.200 indicates these changes were not statistically significant and suggests that the iron biofilms, therefore, did not affect the dissolved-organic-carbon concentration of the well water. Changes in organic- or inorganic-carbon concentrations of water as it moves through the biofilm could be obscured by natural fluctuations in background concentrations.

Sulfate, Nitrate, and Phosphate

Sulfate concentrations increased at 15 of 25 wells after reconditioning, decreased at 5 wells, and remained unchanged at 5. The p-value of 0.014 indicates that the changes were statistically significant and that the biofilm removes sulfate from solution. This process could be due to the bacterially mediated reduction of sulfate to sulfide within the well; iron sulfide was observed in several biofilm samples, and sulfate-reducing bacteria are known to be present in the aquifer.

Nitrate concentrations increased at 14 of the 24 wells after reconditioning, but the p-value of 0.30 indicates that the change was not statistically significant and suggests that the iron biofilms did not affect the nitrate concentrations. Phosphate concentrations increased after reconditioning in 4 of 14 wells, decreased in 4 wells, and remained unchanged in the 6 remaining wells; the p-value of 0.500 indicates that the changes were not statistically significant and suggests that the biofilms did not greatly affect the concentration of phosphorus in well water.

The growth of iron bacteria requires nitrogen and phosphorus; therefore, a growing biofilm would remove those elements from solution. The resulting decrease in the concentration of those constituents, reported above, probably is masked by natural fluctuations in background concentrations. The use of metaphosphate as a chelating agent for iron in several SCWA wells caused rapid specific-capacity decreases when phosphate was allowed to backwash into the wells (Steven Colabufo, Suffolk County Water Authority, oral commun., 1991); this suggests that phosphate is an important nutrient in the growth of iron bacteria and is commonly a limiting nutrient in ground-water systems (Chapelle, 1993). Phosphate concentrations in wells with iron-related specific-capacity decreases were statistically higher than in unaffected wells (Walter, 1997). Despite the widely reported importance of phosphate in the biofouling process, the biofilms in wells examined did not appear to alter phosphate concentrations of the well water significantly.

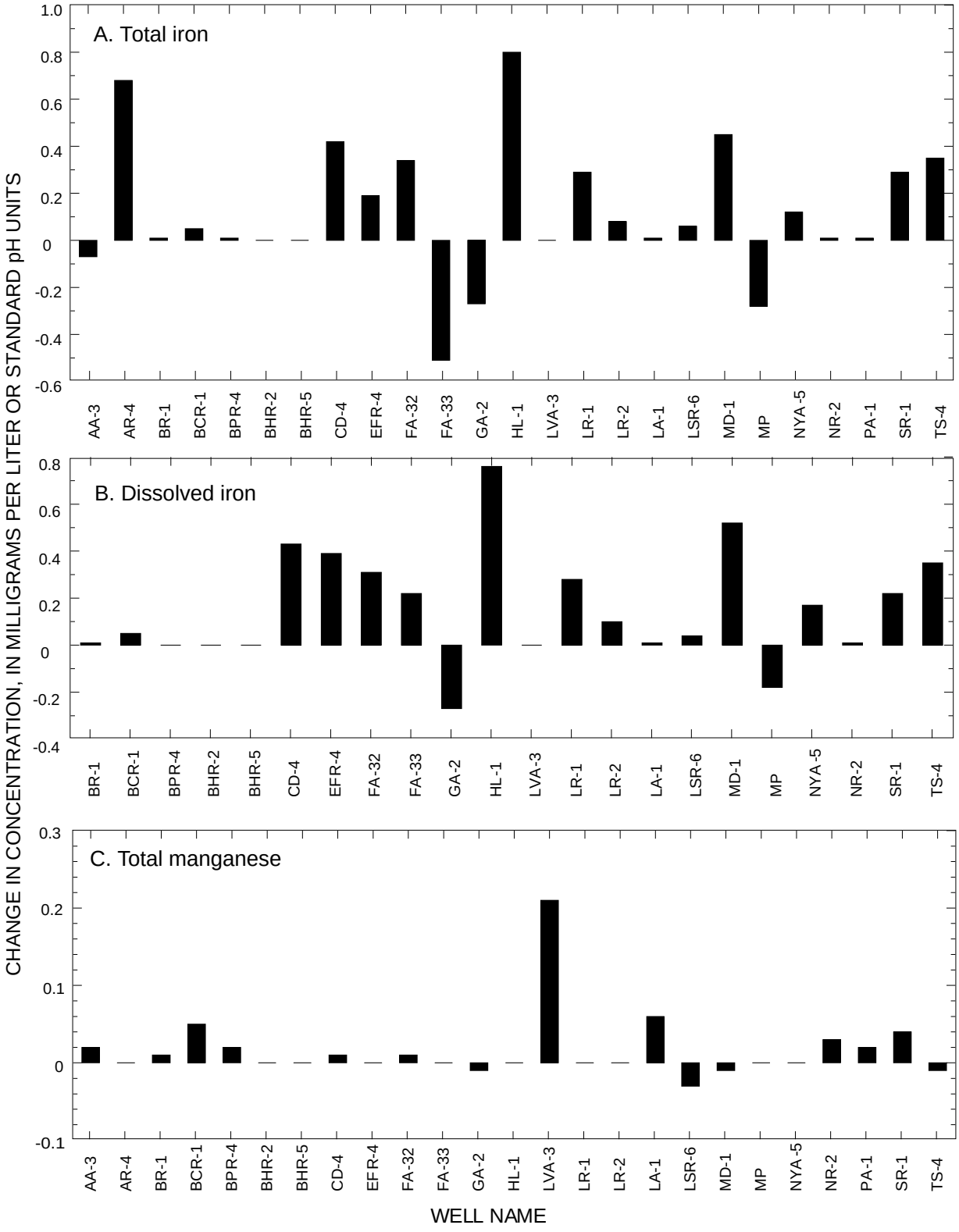

Figure 10. Changes in concentrations of selected constituents in well water and treated lubrication water samples from Suffolk County Water Authority wells, Suffolk County, N.Y., after removal of encrusting material and iron-bacteria biofilms from well screens. (Well locations are shown in fig. 4.) A. Total iron. B. Dissolved iron. C. Total manganese. D. Dissolved manganese. E. pH. F. Sulfate.

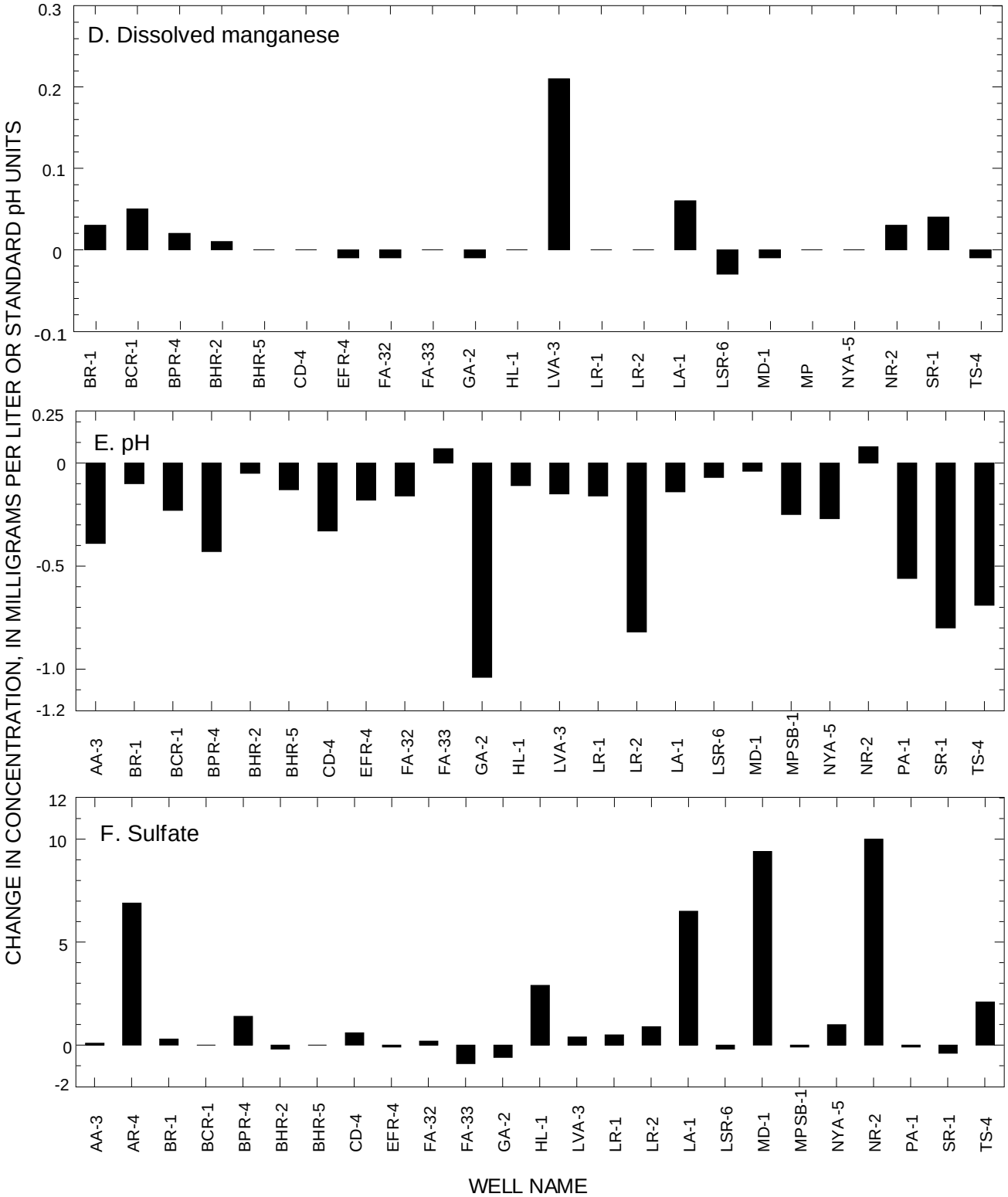

Figure 10 (continued). Changes in concentrations of selected constituents in well water and treated lubrication water samples from Suffolk County Water Authority wells, Suffolk County, N.Y., after removal of encrusting material and iron-bacteria biofilms from well screens. (Well locations are shown in fig. 4.) A. Total iron. B. Dissolved iron. C. Total manganese. D. Dissolved manganese. E. pH. F. Sulfate.

SUMMARY

Iron oxidation and precipitation, and the growth of iron-related bacteria on the screens of production wells and in the pore spaces of the surrounding aquifer, can decrease the specific capacity and yield of wells. These effects are serious in parts of Suffolk County, N.Y., and have compelled the Suffolk County Water Authority, the primary water supplier for the County, to adopt a costly well-reconditioning and replacement program to ensure adequate supplies. The specific-capacity declines, which are most common where production wells are screened in reducing environments, are attributed to the oxidation of ferrous iron and the precipitation of iron oxyhydroxides as well as to the growth of iron-related bacteria on and around the well screens.

Chemical and mineralogic analyses of encrusting material from several affected wells show the inorganic part of the material to be primarily amorphous ferric hydroxide with lesser amounts of goethite, hematite, maghemite or magnetite, and quartz and to consist primarily of iron with some silica, manganese, and trace elements. Microscopic analysis of the material showed that much of the encrusting material was organic and consisted of extracellular bacteria filaments.

Chemical-equilibria modeling showed that water samples from pumping wells were supersaturated with respect to goethite, hematite, magnetite, quartz, and, in a few samples, maghemite; these mineral phases would be thermodynamically favored to precipitate from the water. The samples were undersaturated with respect to ferric hydroxide, the primary inorganic component of the encrusting material. Thermodynamic data indicated that ferric hydroxide would not be expected to precipitate while the wells were in operation because the pumping would keep ground water of naturally low pH in the well and thereby maintain undersaturation.

Ferrous/ferric-iron data and dissolved-oxygen concentrations indicate that the well waters were in a state of redox disequilibrium. Theoretical Eh values computed from the ferrous/ferric iron couple and the oxygen/water couple averaged 390 mv and 810 mv, respectively. This disequilibrium indicates that ground waters of differing chemistry are mixing near the well as a result of either (1) the drawdown of oxygenated water into reducing environments with high dissolved-iron concentration, or (2) the inflow of differing waters along the length of the well screen. Ferrous- and ferric-iron data show that most of the total dissolved iron was ferrous iron, indicating that little ferrous-iron oxidation had occurred prior to sampling at the well head.

The rate of ferrous-iron oxidation, which is strongly pH-dependent, was extremely slow in the water samples. Half-times of oxidation, as estimated from dissolved-oxygen concentration and pH, ranged from 0.001 days to 2,700 days and averaged 4.2 days; the slow rate of oxidation is attributed to the low pH of the ground water. Mineralogical analysis, iron-oxide saturation indices, redox conditions, and iron-oxidation kinetics indicate that significant iron encrustation does not occur while the wells are in operation and that most of the encrusting material is deposited when the wells are shut down. The data also show that manganese, which also can cause well-screen encrustation, does not play an important role in the encrustation process in Suffolk County.

The use of treated water, which has a high pH, as lubrication water when wells are shut down could significantly increase the pH in the static-water column. Saturation indices for iron oxide mineral phases were much higher in the treated lubrication water than in ambient water. Supersaturation of the treated water with respect to ferric hydroxide, the primary inorganic component of the encrusting material, indicates that much of the encrusting material is deposited when the wells are not in operation. The rate of iron oxidation in the treated water was much faster than in ambient water; the half-times of oxidation in treated water had a median of 12 min (0.008 days) as opposed to a median of 4.2 days for ambient water. The use of high-pH and high-iron well water for pump lubrication could also explain the encrustation in wells in areas of low iron concentrations.

The second contributor to encrustation in and around production wells is the growth of iron-related bacteria, which form long extracellular filaments that become coated with iron oxyhydroxides. Iron bacteria were identified by light microscopy in 31 of the 43 samples of encrusting material collected from affected wells. The heterotrophic iron bacteria *Leptothrix ocracea* and *Leptothrix discophora* were found in 8 and 9 of the 43 samples, respectively, and the heterotrophic species *Crenothrix polyspora* was observed in 6. The most common species observed in the biofilm samples was *Gallionella ferruginea* (29 of 43 samples). *Gallionella* was observed in samples from wells differing widely in water chemistry; statistical analysis

of water-quality data indicated, however, that the species occurs primarily in wells with low, but detectable, dissolved-oxygen concentration and high dissolved-iron concentration. *Gallionella* was more common in samples from the Magothy aquifer than in samples from the upper glacial aquifer.

Analysis of core samples revealed iron-related bacteria in the aquifer sediments; these bacteria can shed extracellular material and can be transported for long distances within a ground-water system. Iron bacteria also were detected in circulation water used during the drilling of replacement wells and test borings. Field data indicate that lignite, which is common in the Magothy aquifer, can accelerate the formation of iron-bacteria biofilms in new wells.

The reduction of sulfate to sulfide and the precipitation of iron-sulfide mineral phases could be locally important in the iron-encrustation process. Black material was observed in samples from six affected wells; analysis of one sample indicated it was composed primarily of iron and sulfur. The material, which was associated with iron oxyhydroxides, could have developed microredox environments within the biofilms; microaerophilic bacteria, such as *Gallionella*, often rely on sulfate-reducing bacteria to maintain low-oxygen environments.

The presence of a biofilm can affect well-water quality. Statistical comparison of water samples collected before and after reconditioning suggest that biofilms remove iron and manganese from solution; they also remove trace elements from solution, as evidenced by the presence of several trace elements in the biofilms examined. The biofilms also increase the pH of the well water, possibly by consuming organic acids or removing CO_2 from solution. The iron-bacteria biofilms did not appear to change the concentrations of carbon dioxide, organic carbon, nitrate, or phosphate; the effects of biofilms on these constituents could be obscured, however, by natural fluctuations in background concentrations.

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Table 9. Inorganic chemical analyses of ground-water samples collected at Suffolk County Water Authority wells, Suffolk County, N.Y., January 1991 through August 1992

[Complete well names and abbreviations are shown in table 2. Analyses by Suffolk County Water Authority Central Laboratory, Oakdale, N.Y. NYSDEC, New York State Department of Environmental Conservation; µS/cm, microsiemens per centimeter at 25 degrees Celsius; ˚C, degrees Celsius; mg/L, milligrams per liter; --, no data available; <, less than]

				Constituent or property													
Well name	NYSDEC number	Date	Time	Specific conductance (µS/cm)	pH (standard units)	Temperature (˚C)	Dissolved oxygen (mg/L)	Total calcium (mg/L)	Dissolved calcium (mg/L)	Total magnesium (mg/L)	Dissolved magnesium (mg/L)	Total sodium (mg/L)	Dissolved sodium (mg/L)	Total potassium (mg/L)	Dissolved potassium (mg/L)	Field alkalinity (mg/L as $CaCO_3$)	Total sulfate (mg/L)
AA-3	S48014	2-13-91	1200	109	7.3	8.0	1.0	12	--	4.1	--	4.8	--	1	--	--	9.9
AA-3	S48014	3-7-91	0930	105	6.9	9.0	1.0	8.4	--	3.6	--	3.8	--	1	--	--	10
AA-3	S48014	8-28-91	0245	106	7.5	11.5	1.0	8.9	9.3	3.8	3.8	4.8	4.9	1	1	35	10
AR-4	S37861	1-22-91	1100	31	--	--	--	1.4	--	.5	--	2.7	--	.4	--	--	5.9
AR-4	S37861	1-31-91	1315	182	--	--	--	4.0	--	1.6	--	13	--	.4	--	--	5.8
AR-4	S37861	2-22-91	1400	38	--	--	--	1.6	--	.4	--	2.2	--	.3	--	--	13
AR-4	S37861	7-7-92	1400	31	5.2	12.0	0.5	0.7	0.7	0.4	0.4	2.8	2.8	0.4	0.5	2	5.0
BA-1	S23440	7-17-92	1200	171	6.0	12.5	9.2	13	13	4.0	4.0	11	11	2	2	24	15
BA-3	S36714	8-29-91	1230	40	6.1	10.5	3.0	3.9	2.4	.9	.9	3.4	3.4	.5	.5	40	1.7
BCR-1	S32326	4-9-92	0930	162	4.9	11.0	10.2	6.1	5.6	4.0	.9	15	16	2	2	12	12
BHR-2	S83707	10-7-91	0230	104	5.6	11.0	10.2	5.5	--	3.0	--	8.5	8.6	0.5	0.5	--	12
BHR-2	S83707	10-21-91	1445	124	5.5	11.0	9.6	4.8	6.3	2.8	2.9	12	12	.75	.8	--	13
BHR-2	S83707	10-21-91	1446	--	--	--	--	--	6.2	--	2.9	--	11	--	1	--	--
BHR-5	S73332	11-6-91	1200	277	5.6	11.0	9.2	16	17	9.4	9.5	18	13	2	2	--	34
BHR-5	S73332	11-21-91	1215	267	5.6	11.0	9.4	15	15	8.0	8.1	19	19	2	3	--	34
BPR-1	S38320	8-11-91	1400	169	5.8	11.0	10.8	6.7	18	2.8	2.8	18	18	2	3	8	12
BPR-1	S38320	8-11-91	1405	170	--	--	--	6.6	6.7	2.7	2.7	18	18	2	2	--	12
BR-2	S20479	12-13-91	1000	164	5.5	12.0	6.3	11	11	2.6	2.6	2.2	12	3	3	--	25
BR-2	S20479	1-14-92	1400	166	5.4	12.5	6.0	4.0	10	2.6	2.6	15	15	3	3	9	4.9
CD-4	S48193	10-9-91	1200	21	5.3	11.0	10.1	1.7	--	.2	--	2.0	1.9	.1	.2	10	1.3
CD-4	S48193	11-1-91	1315	27	4.9	11.0	3.6	.9	.6	.3	.3	2.7	2.6	.3	.3	1	1.3
CSB-1	S52126	9-3-91	1745	156	6.0	10.8	6.1	8.2	8.3	3.3	3.4	13.	13	2	2	13	12
FA-32	S20566	1-29-92	1200	25	5.5	--	.9	.7	.8	.3	.4	2.6	4.1	.3	.3	25	3.5
FA-32	S20566	3-5-92	1100	27	5.0	11.5	.5	.8	.6	.3	.3	2.0	2.8	.4	.4	27	3.4
FA-33	S26535	11-22-91	0900	24	5.2	12.0	1.0	.4	.5	.4	.4	3.1	3.1	.4	.4	24	3.2
FA-33	S26535	12-18-91	1115	26	5.3	11.5	0.8	.4	.5	.4	.4	3.4	3.1	.4	.4	26	3.4
GA-2	S65505	4-21-92	1400	23	4.9	12.0	.8	1.6	1.5	.3	.3	2.1	2.2	.2	.3	4	2.3
GA-8	S47887	7-22-92	1100	21	4.8	12.0	6.0	.2	.3	.2	.1	2.2	2.3	.3	.3	21	2.5
GBP-1	S54730	7-17-92	1400	60	5.9	11.0	.8	3.5	3.6	1.3	1.3	4.4	4.5	.6	.5	60	9.9
GBP-3	S66183	8-10-92	1700	43	6.0	12.0	.6	3.5	3.0	.8	.8	3.2	3.5	.4	1	12	5.0
GBP-3	S66183	8-10-92	1705	43	--	--	--	3.3	2.9	.8	.8	3.2	3.3	.4	.8	.	5.0

Table 9. Inorganic chemical analyses of ground-water samples collected at Suffolk County Water Authority wells, Suffolk County, N.Y., January 1991 through August 1992--continued

				Constituent or property												
Well name	NYSDEC number	Date	Time	Total chloride (mg/L)	Total nitrate as NO_3 (mg/L)	Total nitrite as NO_2 (mg/L)	Total ammonium as NH_4 (mg/L)	Total phosphate as PO_4 (mg/L)	Dissolved phosphate as PO_4 (mg/L)	Total orthophosphate as PO_4 (mg/L)	Dissolved copper (mg/L)	Total iron (mg/L)	Dissolved iron (mg/L)	Total manganese (mg/L)	Dissolved manganese (mg/L)	Dissolved zinc (mg/L)
AA-3	S48014	2-13-91	1200	3.6	<.10	<.10	<.01	.43	--	.43	--	.10	--	.03	--	--
AA-3	S48014	3-7-91	0930	3.8	<.10	<.10	.02	.35	--	.35	-	.03	--	.05	--	--
AA-3	S48014	8-28-91	0245	3.5	<.10	<.10	<.01	<.10	--	<.01	<.02	.03	.03	.29	.29	.03
AR-4	S37861	1-22-91	1100	3.4	<.10	<.10	<.01	<.10	--	<.01	--	1.1	--	<.01	--	--
AR-4	S37861	1-31-91	1315	3.6	<.10	<.10	<.01	<.10	--	<.01	--	<.03	--	.01	--	--
AR-4	S37861	2-22-91	1400	13	.29	<.10	<.01	<.10	--	<.01	--	1.7	--	<.01	--	--
AR-4	S37861	7-7-92	1400	3.5	<0.10	<0.10	<0.01	<0.10	<0.10	<0.01	<0.02	0.34	0.32	<0.01	<0.01	0.03
BA-1	S23440	7-17-92	1200	16	5.2	<.10	<.01	<.10	<.10	<.01	<.02	.04	.04	.06	.06	<.02
BA-3	S36714	8-29-91	1230	3.5	<.10	<.10	<.01	<.10	--	<.01	<.02	<.03	.03	.01	.01	.10
BCR-1	S32326	4-9-92	0930	21	4.0	<.10	<.01	.18	--	--	.08	.18	.08	.06	.06	<.02
BHR-2	S02405	10-9-91	0230	11	1.8	<.10	<.01	<.10	--	<.01	<.02	<.03	<.03	.02	.01	.05
BHR-2	S02405	10-21-91	1445	15	2.5	<.10	<.01	.16	--	.09	<.02	<.03	<.03	.02	.02	.02
BHR-2	S02570	10-22-91	1230	--	--	--	--	--	--	--	<.02	--	<.03	--	.03	.18
BHR-5	S73332	11-6-91	1200	29	7.2	<.10	<0.01	.11	--	<0.01	<0.02	0.04	<0.03	0.02	0.02	0.02
BHR-5	S73332	11-21-91	1215	31	6.7	<.10	<.01	<.10	--	<.01	<.02	.14	<.03	.02	.02	<.02
BPR-1	S38320	8-11-91	1400	28	3.1	<.10	<.01	<.10	--	<.01	<.02	<.03	<.03	.10	.10	<.02
BPR-1	S38320	8-11-91	1405	28	3.1	<.10	<.01	<.10	--	<.01	.02	<.03	<.03	.10	.10	<.02
BR-2	S20479	12-13-91	1000	17	4.1	<.10	<.01	.13	--	<.01	<.02	<.03	<.03	.09	.08	<.02
BR-2	S20479	1-14-92	1400	16	4.2	.10	<.01	--	--	--	.02	.03	.03	.10	.11	.04
CD-4	S48193	10-9-91	1200	2.7	<.10	<.10	<.01	<.10	--	<.01	.02	.06	.04	<.01	<.01	<.02
CD-4	S48193	11-1-91	1315	3.3	<.10	<.10	<.01	<.10	--	<.01	<.02	.48	.47	.01	<.01	<.02
CSB-1	S52126	9-3-91	1745	19	1.8	<.10	<.01	.17	--	.12	<.02	<.03	<.03	.09	.09	<.02
FA-32	S20566	1-29-91	1200	3.2	<.10	<.10	<.01	<.10	--	--	<.02	.1	.09	<.01	<.01	<.02
FA-32	S20566	3-5-91	1100	3.3	<.10	<.10	<.01	<.1	<.10	<.01	<.02	.44	.40	.01	<.01	.03
FA-33	S26535	11-22-91	0900	3.4	<.10	<.10	<.01	<.10	<.10	<.01	<.02	.87	.15	<.02	<.01	.02
FA-33	S20566	12-17-91	1115	3.4	<.10	<.10	<.01	<.10	--	<.01	<.02	.36	.35	<.01	<.01	<.02
GA-2	S65505	4-21-92	1400	2.8	<.1	<.10	<.01	<.10	<.10	<.01	<.02	.05	.35	<.01	<.01	<.02
GA-8	S47887	7-22-92	1100	3.2	<.10	<.10	<.01	<.10	<.10	<.01	<.02	.27	.26	<.01	<.01	.03
GBP-1	S54730	7-17-92	1400	5.8	<.10	<.10	<.01	<.10	<.10	<.01	<.02	.35	.33	.01	.01	.02
GBP-3	S66183	8-10-92	1700	3.4	.12	<.10	<.01	.11	--	<.01	<.02	.53	.52	.03	.02	<.07
GBP-3	S66183	8-10-92	1705	3.4	<.10	<.10	<.01	.11	--	.08	<.02	.52	.49	.03	.02	.03
HL-1	S21366	5-6-92	1400	3.2	<.10	<.10	<.01	<.10	.12	--	<.02	1.4	1.4	.02	.03	.03
HL-3	S39024	6-30-92	1400	3.1	<.10	<.10	<.01	<.10	.22	--	<.02	.60	.58	.01	.01	.04
LA-1	S38701	4-8-91	1445	26	6.8	<.10	<.01	.17	--	.17	<.02	.05	.03	.23	.22	<.02
LKA-3	S09893	1-8-92	1220	11	.43	<.10	<.01	<.10	--	--	.15	<.03	<.03	.46	.51	.04
LKA-3	S09893	1-8-92	1219	9.8	.39	<.10	<.01	<.10	--	<.01	.02	<.03	<.03	.25	.25	.03

Table 9. Inorganic chemical analyses of ground-water samples collected at Suffolk County Water Authority wells, Suffolk County, N.Y., January 1991 through August 1992--continued

Well name	NYSDEC number	Date	Time	Constituent or property												
				Total chloride (mg/L)	Total nitrate as NO_3 (mg/L)	Total nitrite as NO_2 (mg/L)	Total ammonium as NH_4 (mg/L)	Total phosphate as PO_4 (mg/L)	Dissolved phosphate as PO_4 (mg/L)	Total orthophosphate as PO_4 (mg/L)	Dissolved copper (mg/L)	Total iron (mg/L)	Dissolved iron (mg/L)	Total manganese (mg/L)	Dissolved manganese (mg/L)	Dissolved zinc (mg/L)
LR-1	S28503	12-16-91	1230	2.7	<.10	<.10	<.01	.12	--	<.01	<.02	.10	.11	<.01	<.01	<.02
LR-1	S28503	2-3-92	1400	2.7	<.10	<.10	<.01	--	--	--	---	.41	--	<.01	--	--
LR-2	S33005	2-25-92	1400	2.9	<.10	<.10	<.01	.34	.35	--	<.02	.16	.15	<.01	<.01	<.02
LR-2	S33005	5-15-92	1300	2.8	<.10	<.10	<.01	<.10	--	--	<.02	.24	.25	<.01	.01	.02
LSR-5	S31037	9-10-91	1030	29	<.89	<.10	<.01	<.10	--	<.01	<.02	.25	.25	.01	.01	.02
LSR-5	S31037	9-17-91	1100	--	--	--	--	--	--	--	--	.27	.26	.01	.02	--
LSR-6	S67819	2-24-92	1200	16	<.10	<.10	<.01	<.10	--	<.01	<.02	.04	.01	.05	<.01	<.02
LSR-6	S67819	2-28-92	1310	17	<.10	<.10	<.01	--	--	--	<.02	.10	.05	.02	.01	<.02
MD-1	S18729	3-19-91	1500	4.8	<.10	<.10	<.01	.32	--	.32	.05	.20	.13	.02	.02	<.02
MD-1	S18729	9-10-91	1230	4.2	<.10	<.10	<.01	<.10	--	.12	<.02	.65	.65	.01	.01	.03
MPSB-1	S70155	12-4-91	1500	29	<.10	<.10	<.01	<.10	--	<.01	<.02	.97	.88	.03	.03	.02
MPSB-1	S70155	2-4-92	1300	31	<.10	.10	<.01	--	--	--	--	--	.70	--	--	.03
NR-2	S33308	4-9-92	1230	26	5.6	<.10	<.01	.22	.18	--	<.02	.04	.03	.15	.15	<.02
NYA-5	S23631	11-6-91	1430	3.6	<.10	<.10	<.01	<.1	--	<.01	<.02	.86	.86	<.01	.01	<.02
NYA-5	S23631	11-21-91	1415	6.2	<.10	<.10	<.01	<.10	--	<.01	.04	1.0	.98	.01	.01	.64
PA-1	S39347	2-5-91	1300	5.7	.24	<.10	<.01	1.5	--	1.5	--	.08	--	<.01	--	--
PA-1	S39347	2-22-91	1415	15	5.7	<.10	<.01	1.1	--	1.1	--	.09	--	.03	--	--
PA-1	S39347	9-3-91	1545	17	6.0		<.01	.12	--	.12	.02	.03	.03	.04	.04	<.02
RA-1	S17037	8-11-92	1530	15	5.7	<.10	<.01	<.10	--	<.01	<.02	<.03	<.03	.16	.16	.02
RA-1	S17037	8-11-92	1535	15	5.7	<.10	<.01	<.10	--	<.01	.02	<.03	<.03	.16	.16	.02
RRA-1	S33920	8-12-92	1130	5.2	<.10	<.10	<.01	<.10	--	<.01	<.02	.32	.32	.02	.02	.02
RRA-1	S33920	8-12-92	1135	5.2	<.10	<.10	<.01	<.10	--	<.01	<.02	.32	.32	.02	.02	.03
SA-3	S40498	7-22-92	1600	2.7	<.10	<.10	<.01	<.10	<.10	<.01	<.02	.15	.27	<.01	.01	.02
SL-2	S36459	8-29-91	1615	3.9	.30	<.1	<.01	<.10	--	<.01	.03	.03	<.03	.02	<.01	.05
SR-1	S33826	10-8-91	0200	29	1.5	<.10	<.01	<.10	--	<.01	.06	<.03	<.03	<.01	<.01	<.02
SR-1	S33826	10-29-91	1200	37	1.5	<.10	<.01	<.10	--	<.01	.21	.36	.25	.05	.04	.03
TA-2	S50546	9-25-91	0900	--	--	--	--	--	--	--	--	.34	.34	.01	.01	--
TS-4	S51457	11-13-91	1700	5.5	<.10	<.10	<.01	<0.10	--	.72	<0.02	.86	.37	.03	.02	.02
TS-4	S51457	12-3-91	1015	3.2	<.10	<.10	<.01	<.10	--	<.01	<.02	.74	.72	<.01	<.01	.03

Table 10. Inorganic chemical analyses of lubrication-water samples collected at Suffolk County Water Authority wells, Suffolk County, N.Y., July-August 1992

[Complete well names and abbreviations are shown in table 2. Analyses by Suffolk County Water Authority Central Laboratory, Oakdale, N.Y. NYSDEC, New York State Department of Environmental Conservation; µS/cm, microsiemens per centimeter at 25 degrees Celsius; ˚C, degrees Celsius; mg/L, milligrams per liter; --, no data available; <, less than]

				Constituent or property													
Well name	NYSDEC number	Date	Time	Specific conductance (µS/cm)	pH (standard units)	Temperature (˚C)	Dissolved oxygen (mg/L)	Total calcium (mg/L)	Dissolved calcium (mg/L)	Total magnesium (mg/L)	Dissolved magnesium (mg/L)	Total sodium (mg/L)	Dissolved sodium (mg/L)	Total potassium (mg/L)	Dissolved potassium (mg/L)	Field alkalinity (mg/L as $CaCO_3$)	Total sulfate (mg/L)
BA-3	S36714	8-10-92	1330	78	--	--	--	10	10	0.5	0.5	3.7	3.8	0.4	0.6	--	2.4
BA-3	S36714	8-10-92	1335	77	6.7	13.0	2.2	13	10	.5	.5	3.7	3.8	.4	.5	33	2.4
CSB-1	S52126	7-27-92	1730	107	7.8	12.5	.5	13	13	1.6	1.5	6.0	6.4	.9	.9	51	3.8
FA-33	S26535	7-27-92	1430	80	8.0	14.0	2.6	12	12	.4	.4	3.2	3.2	.4	.4	36	3.2
GA-2	S65505	7-22-92	1730	56	6.5	14.0	.6	6.7	6.6	.3	.3	3.2	3.3	.3	.3	19	3.4
GA-8	S47887	7-23-92	1800	97	7.5	13.0	3.6	8.0	8.2	.2	.2	8.9	8.9	.4	.4	43	2.3
GBP-1	S54730	8-10-92	1800	96	8.4	13.5	.5	12	12	.9	.8	4.8	4.8	.4	.4	37	5.0
GBP-1	S54730	8-10-92	1805	97	--	--	--	12	12	.9	.8	4.9	4.9	.4	.5	--	5.1
HL-1	S21366	7-22-92	1415	107	7.4	12.5	2.8	11	11	.9	.9	5.1	5.5	.5	.6	38	9.5
HL-3	S39024	7-22-92	1430	79	7.3	12.0	2.1	10	11	.7	.6	4.1	4.2	.4	.6	38	3.0
LR-1	S28503	7-22-92	1530	77	7.4	13.0	1.0	9.8	10	.3	.2	2.3	2.4	.3	.4	38	2.0
LR-2	S33005	7-22-92	1645	75	8.0	14.0	1.7	12	11	.3	.3	2.4	2.4	.3	.3	36	2.0
LSR-5	S31037	7-29-92	1500	293	7.1	13.0	6.3	22	21	4.7	4.6	15	15	1	2	37	21
NYA-5	S23631	7-29-92	1830	77	7.8	15.0	5.1	11	9.0	1.4	1.1	4.1	3.8	.5	.6	35	2.9
SA-3	S40498	7-27-92	1400	79	7.7	15.0	.5	10	8.7	.4	.3	3.2	3.3	.4	.7	32	3.8
SR-1	S33826	7-29-92	1130	178	6.9	14.0	11.2	7.1	6.5	2.3	2.1	21	20	1	1	18	9.7
TA-2	S50546	7-22-92	1230	80	7.0	14.0	0.4	8.0	7.8	.6	.6	5.3	5.4	.4	.5	26	4.8
TS-4	S51457	7-23-92	1700	349	7.7	12.5	3.5	28	29	2.4	2.3	36	36	5	5	130	6.2

Table 10. Inorganic chemical analyses of lubrication-water samples collected at Suffolk County Water Authority wells, Suffolk County, N.Y., July-August 1992--continued

				Constituent or property													
Well name	NYSDEC number	Date	Time	Total chloride (mg/L)	Total nitrate (mg/L)	Total nitrite (mg/L)	Total ammo-nium (mg/L)	Total phos-phate (mg/L)	Dis-solved phos-phate (mg/L)	Total ortho-phos-phate (mg/L)	Dis-solved copper (mg/L)	Total iron (mg/L)	Dis-solved iron (mg/L)	Total manga-nese (mg/L)	Dis-solved manga-nese (mg/L)	Dis-solved zinc (mg/L)
BA-3	S36714	8-10-92	1330	4.1	<0.10	<0.10	<0.01	<0.10	--	<0.01	0.04	0.17	0.04	<0.01	<0.01	0.04
BA-3	S36714	8-10-92	1335	4.2	<.10	<.10	<.01	<.10	--	<.01	.03	.17	.04	.01	<.01	.04
CSB-1	S52126	7-27-92	1730	5.7	<.10	<.10	<.01	1.1	.83	1.1	.03	.68	.29	.03	.01	.02
FA-33	S26535	7-27-92	1430	3.7	<.10	<.10	<.01	<.10	<.10	<.01	<.02	.24	.14	<.01	<.01	<.02
GA-2	S65505	7-22-92	1730	3.4	<.10	<.10	<.01	1.8	.78	.94	.04	.28	.25	<.01	<.01	<.02
GA-8	S47887	7-23-92	1800	3.1	<.10	<.10	<.01	1.9	1.8	1.8	<.02	.76	.58	.01	--	.02
GBP-1	S54730	8-10-92	1800	5.2	<.10	<.10	<.01	.68	--	.48	<.02	.76	.50	.02	.02	<.02
GBP-1	S54730	8-10-92	1805	5.3	<.10	<.10	<.01	.78	--	.49	<.02	.73	.49	.02	.02	<.02
HL-1	S21366	7-22-92	1415	5.1	<.10	<.10	<.01	1.8	1.5	1.6	.02	.67	.42	.04	.03	.03
HL-3	S39024	7-22-92	1430	4.2	<.10	<.10	<.01	2.2	2.2	1.8	<.02	.41	.37	.02	.02	<.02
LR-1	S28503	7-22-92	1530	2.9	<.10	<.10	<.01	.50	--	.50	.02	.26	.20	.01	.01	<.02
LR-2	S33005	7-22-92	1645	2.7	<.10	<.10	<.01	.22	.28	.22	.05	.13	.25	<.01	--	<.02
LSR-5	S31037	7-29-92	1500	29	1.9	<.10	<.01	<.10	.14	<.01	<.02	.23	.14	.01	<.01	.03
NYA-5	S23631	7-29-92	1830	4.4	<.10	<.10	<.01	.38	<.10	.16	<.02	.08	.06	<.01	<.01	<.02
SA-3	S40498	7-27-92	1400	3.6	<.10	<.10	<.01	.41	.44	.34	.05	.24	.21	.01	.01	.02
SR-1	S33826	7-29-92	1130	30	1.7	<.10	<.01	.04	--	.02	.12	.05	.05	.02	.02	.03
TA-2	S50546	7-22-92	1230	5.1	<.10	<.10	<.01	4.1	3.8	2.4	.05	.24	.18	<.01	<.01	.03
TS-4	S51457	7-23-92	1700	54	.15	<.10	1.9	1.1	.41	.72	.02	.86	.21	.03	.03	.03