Bias in Ground-Water Data Caused by Well-Bore Flow in Long-Screen Wells

by Peter E. Church and Gregory E. Granato[a]

Abstract

The results of a field experiment comparing water-quality constituents, specific conductance, geophysical measurements, and well-bore hydraulics in two long-screen wells and adjacent vertical clusters of short-screen wells show bias in ground-water data caused by well-bore flow in long-screen wells. The well screen acts as a conduit for vertical flow because it connects zones of different head and transmissivity, even in a relatively homogeneous, unconfined, sand and gravel aquifer where such zones are almost indistinguishable. Flow in the well bore redistributes water and solutes in the aquifer adjacent to the well, increasing the risk of bias in water-quality samples, failure of plume detection, and cross-contamination of the aquifer. At one site, downward flow from a contaminated zone redistributes solutes over the entire length of the long-screen well. At another site, upward flow from an uncontaminated zone masks the presence of a road salt plume.

Borehole induction logs, conducted in a fully penetrating short-screen well, can provide a profile of solutes in the aquifer that is not attainable in long-screen wells. In this study, the induction-log profiles show close correlation with data from analyses of water-quality samples from the short-screen wells; however, both of these data sets differ markedly from the biased water-quality samples from the long-screen wells. Therefore, use of induction logs in fully cased wells for plume detection and accurate placement of short-screen wells is a viable alternative to use of long screen wells for water-quality sampling.

Introduction

Selection of proper well design is critical for collection of water-quality samples that are representative of the quality of water in the aquifer. Historically, long-screen wells, herein defined as wells that screen more than one discrete zone of head or transmissivity, have been used to collect water-quality samples at specific depths and to provide a vertical profile of solutes in the aquifer. Long-screen wells were installed in the site evaluation phase of an investigation to determine the relative effectiveness of four highway-drainage designs in preventing road-salt contamination of ground water. Wells were installed with 10- to 20-m-long screens in a relatively homogeneous, unconfined sand and gravel aquifer, because preliminary inspections of aquifer sediments sampled during drilling indicated only small variations with depth (Pollock, 1984). However, sample bias became a concern because analyses of water-quality samples from long-screen wells were inconsistent with data from borehole induction logs. A literature search indicated an emerging debate concerning the possiblity that water-quality samples collected from long-screen wells may not be representatitive of the quality of water in the aquifer because of vertical flow of water in the well. Results from theoretical and experimental studies have indicated the potential for bias in water-quality samples collected from long-screen wells. However, definitive results from field experiments in a relatively homogeneous, unconfined sand and gravel aquifer are not available, creating the need for this experiment.

This paper presents the results of a field experiment that shows bias in ground-water data caused by well-bore flow in long-screen wells. Analyses of water-quality samples collected in long-screen wells were compared to analyses of water-quality samples collected in adjacent vertical clusters of short-screen wells, and geophysical logs were used to determine which set of data were most representative of water quality in the aquifer. Results of field experimentation indicate that (a) use of long-screen wells does not provide representative water-quality samples, even in a relatively homogeneous, unconfined sand and gravel aquifer, (b) vertical flow in long-screen wells can contaminate zones of the aquifer that are not contaminated upgradient of the well, and (c) use of induction logs in fully cased wells for plume detection and accurate placement of short-screen or multi-level sampling wells is a viable alternative to the use of long-screen wells for water-quality sampling. A road-salt plume was studied from January through June 1990 using water-quality data, borehole-fluid conductance logs, borehole-fluid velocity logs, and borehole-induction logs collected from two test sites each having one long-screen well and an adjacent cluster of short-screen wells.

This experiment was done by the U.S. Geological Survey in cooperation with the Massachusetts Highway Department, Research and Materials Division, and the U.S.

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Department of Transportation, Federal Highway Administration. The experiment is part of a larger investigation to determine the relative effectiveness of four highway-drainage designs in preventing road salt contamination of ground water along State Route 25 in southeastern Massachussetts (Figure 1) (Church and Friesz, 1993a).

Background

The available literature in the late 1980s indicates that unbiased detection and analysis of ground-water contaminants can be made only with proper design and installation of monitoring wells, coupled with appropriate sampling methods. Studies by Bennett and Patten (1962) and Papadopulos (1966) show that well-bore flow may be significant in unstressed long-screen wells that are screened in more than one aquifer because of differences in head and transmissivity between the aquifers. However, the possiblity that well-bore flow, caused by small vertical gradients present within relatively homogeneous, unconfined aquifers, was not explored.

In a well-referenced debate on the topic (Giddings, 1987), the proponents of long-screen wells claimed that long-screen wells are more sensitive to the presence of contaminants than short-screen wells because of better hydrologic connection with the aquifer. The ability to use discrete interval sampling, based on borehole electrical conductivity profiling to locate high concentration zones, is used as a defense against the argument that contaminants may be diluted to concentrations below detection limits in a composite sample from a long-screen well. Finally, the additional cost of more short-screen wells, and samples therefrom, support the argument for the use of long-screen wells (Giddings, 1987).

Confidence in the use of long-screen wells for water-quality sampling is countered many times in the literature by several hydrologists. Shosky (1987) criticized the use of long-screen wells for water-quality sampling because they commonly were installed without site-specific knowledge of hydrogeologic conditions. A long screen could penetrate many units of different thicknesses and hydraulic conductivities. If only one unit is contaminated, concentrations of contaminants in water samples from the long-screen well will be diluted by inflow from other contributing units. Shosky also concluded, using basic hydrologic principles, that water-quality samples from short-screen wells will provide a better definition of plume concentration and location in a homogeneous, sand aquifer than would be obtained from a long-screen well. Cohen and Rabold (1988) modeled a series of pumping scenarios and concluded that the quantity of water actually pumped from each layer to make up a composite sample from a long-screen well was proportional to the vertical distribution of formation permeability. Robbins (1989) demonstrated by use of an analytical model that sample concentrations from long-screen wells may underestimate concentration of the contaminant in the aquifer by dilution. Kaleris (1989) showed by use of a numerical model that contaminant concentrations of samples pumped from long-screen wells can vary depending on pumping rate, position of the pump intake, well diameter, and vertical distributions of contaminants and hydraulic conductivities in the aquifer. A lab experiment by Barczewski and Marschall (1989) showed that tracer sample concentrations in a well screened through two layers of different hydraulic conductivity can be influenced by the pumping rate, volume pumped, and position of the pump intake.

McIlvride and Rector (1988) compared hydrogeology, hydraulic data, and analyses of water-quality samples collected from long- and short-screen wells at several alluvial sites. The results indicate that where significant contrasts exist in transmissivity between two or more aquifers screened by one well, hydraulic head and water quality in the well primarily reflect the aquifer with the highest transmissivity. Also, where transmissivities are similar, water quality and measured water level in the well tend to reflect the aquifer with the highest hydraulic head. McIlvride and Rector (1988) concluded that in areas with a measurable vertical hydraulic gradient, a long-screen well causes either cross-contamination of other aquifers or dilution of the contaminant. This well-bore flow and redistribution of contaminants between aquifers may cause either an overestimation of contaminant concentrations or an underestimation as severe as the failure to detect the contaminants. These relations were apparent in an area with a 1.25-m-thick confining bed between zones, so this information suggests, but does not prove, that the same biases could occur in a long-screen well in a homogeneous, unconfined aquifer if vertical head gradients exist.

Reilly et al. (1989) demonstrated numerically that vertical borehole head differences in a homogeneous, unconfined aquifer that are equal to or less than current detection limits are sufficient to produce significant borehole flows. They concluded that these flows render long-screen wells "almost useless" in measuring and detecting contaminated ground water. Where contaminated water preferentially flows into the borehole, composite samples are artificially highly concentrated. In contrast, where uncontaminated water dominates well-bore flow, composite samples are diluted to artificially low concentrations. Reilly et al. (1989) also concluded that borehole flow and transport of contaminants in long-screen wells may contaminate parts of the aquifer that would not otherwise become contaminated in the absence of a long-screen well.

Technology described in current literature is trending toward the use of shorter screen lengths for better vertical definition of contaminant plumes than previously obtained with long-screen wells. For example, Smith et al. (1991) found that vertical gradients in chemical concentrations and microbiological populations vary greatly, in some cases several orders of magnitude, over a vertical distance of 2 m in what would be classically considered a homogeneous, unconfined sand and gravel aquifer. Therefore, Smith et al. (1991) suggested a 1-m vertical sampling density to adequately characterize the contaminant plume. Hydraulic data also support the use of short-screen lengths. Gibs et al. (1993) indicated that pumping rates from adjacent 0.15 to 0.20-m-thick sediment layers ranged from 77 to 120 percent of the median pumping rate in a zone of a sandy aquifer where layering of sediment was not detected by dry-sieve analyses, inspection of cores, and borehole natural-gamma logs. Therefore, chemical stratification coupled with small variations in hydraulic conductivity in a homogeneous aquifer will obscure the nature and extent of contaminant zones in direct proportion to the thickness of the sampled zone, even in the absence of vertical head gradients.

The available literature does provide information from which one may infer that long-screen wells may not be appropriate for sampling, even in homogeneous, unconfined, isotropic, sand and gravel aquifers. However, all the necessary information to reach a definitive conclusion about the use of long-screen wells for water-quality sampling in a relatively homogeneous, unconfined aquifer is not compiled in a single study. Reilly et al. (1989) used a numerical model to predict the vertical flow, cross-contamination of the aquifer, and sample bias that may occur with use of a long-screen well in a homogeneous aquifer. However, these results have not been verified by field experimentation.

Site Description

The well sites, designated B23 in test site B and C13 in test site C (Figure 1), are downgradient from State Route 25, a six-lane highway in the town of Wareham, Massachusetts. The undulating topography in this area is part of a large glacial outwash plain. Split-spoon samples of aquifer material collected during well construction show that the test sites generally are underlain by a 20- to 30-m thick layer of fine to coarse sand and some gravel. Fine sand and some silt typically are present below this sand and gravel unit. Water-table depths below land surface are about 9 m at well site B23 and 12 m at well site C13. The water table, which is within the fine to coarse sand and gravel unit, forms a relatively planar surface compared to the undulating topography of the land surface. Water-table gradients fluctuate annually from about 0.004 to 0.006, and ground-water flow is southeasterly, nearly perpendicular to the highway. Hydraulic conductivities determined from split-spoon samples by use of the Hazen approximation method range from about 1.3 m/d in fine sand and silt to 50 m/d sand and gravel. Based on results from a nearby experimental site where geologic materials are similar (LeBlanc et al., 1987), estimates of effective porosity of the aquifer range from about 0.35 to 0.40.

Background concentrations of chloride, sodium, and calcium in ground water, the primary constituents of road salt, range from 5 to 20 mg/l, 5 to 10 mg/l, and 1 to 10 mg/l, respectively. Background specific conductance of the ground water ranges from 40 to 70 μS/cm. Annual road salting of Route 25 causes a cyclic plume of chloride, sodium, and calcium, which varies as much as about seven times background concentrations in the late spring or early summer at these sites.

Design and Construction of Test Wells

Short-screen wells were installed at two well sites, B23 and C13, in order to collect water-quality samples for comparison with samples collected in long-screen wells. At site B23 (Figure 2), the long-screened well, B2303, was constructed with 21 m of screen extending from just above the

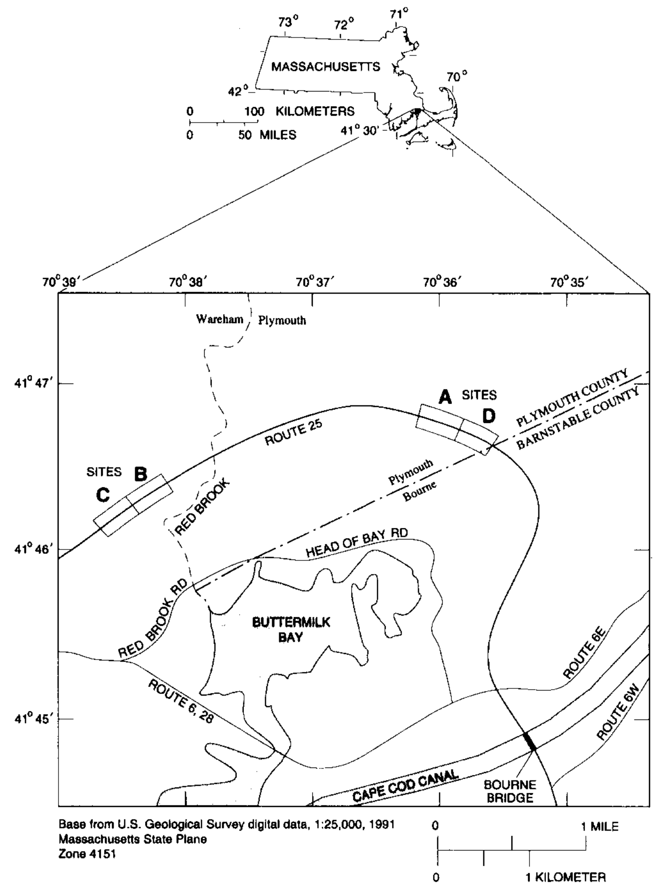

Fig. 1. Location of study area and test sites A, B, C, and D along Route 25 in southeastern Massachusetts.

water table at about 9 to 30 m below land surface. Seven additional short-screen wells with 3-m-length screens, designated B2304 through B2310, were installed just upgradient of the long-screen well. Well screens were placed at 3-m-vertical increments from the water table to the depth of the bottom of the long-screen well. Well construction at site C13 (Figure 3) is similar to that at site B23. At this site, the long-screen well, C1303, was constructed with 18 m of screen extending from just above the water table at about 12 to 30 m below land surface. Six wells, C1304 through C1309, were installed just upgradient of the long-screen well each with 3-m-length screens, and placed at 3-m-vertial increments from the water table to the depth of the bottom of the long-screen well. The placement of screens was staggered vertically and horizontally to minimize the possiblity of cross flow from the screened intervals in adjacent wells. The vertical staggering is shown in Figures 2 and 3. The horizontal staggering, although not visible in Figures 2 and 3, follows a zig-zag pattern where alternating wells are placed along two parallel lines that are 1-m apart. All wells are constructed of 5-cm inside diameter, schedule 40, polyvinylchloride casing (PVC) and slotted screen. Slot widths are 0.25 mm. Boreholes were drilled with 20-cm outside diameter hollow-stem augers. The PVC screen and casing were lowered in the augers when the appropriate depths were reached. Augers flights were then pulled from the borehole allowing the native sandy material to collapse against the

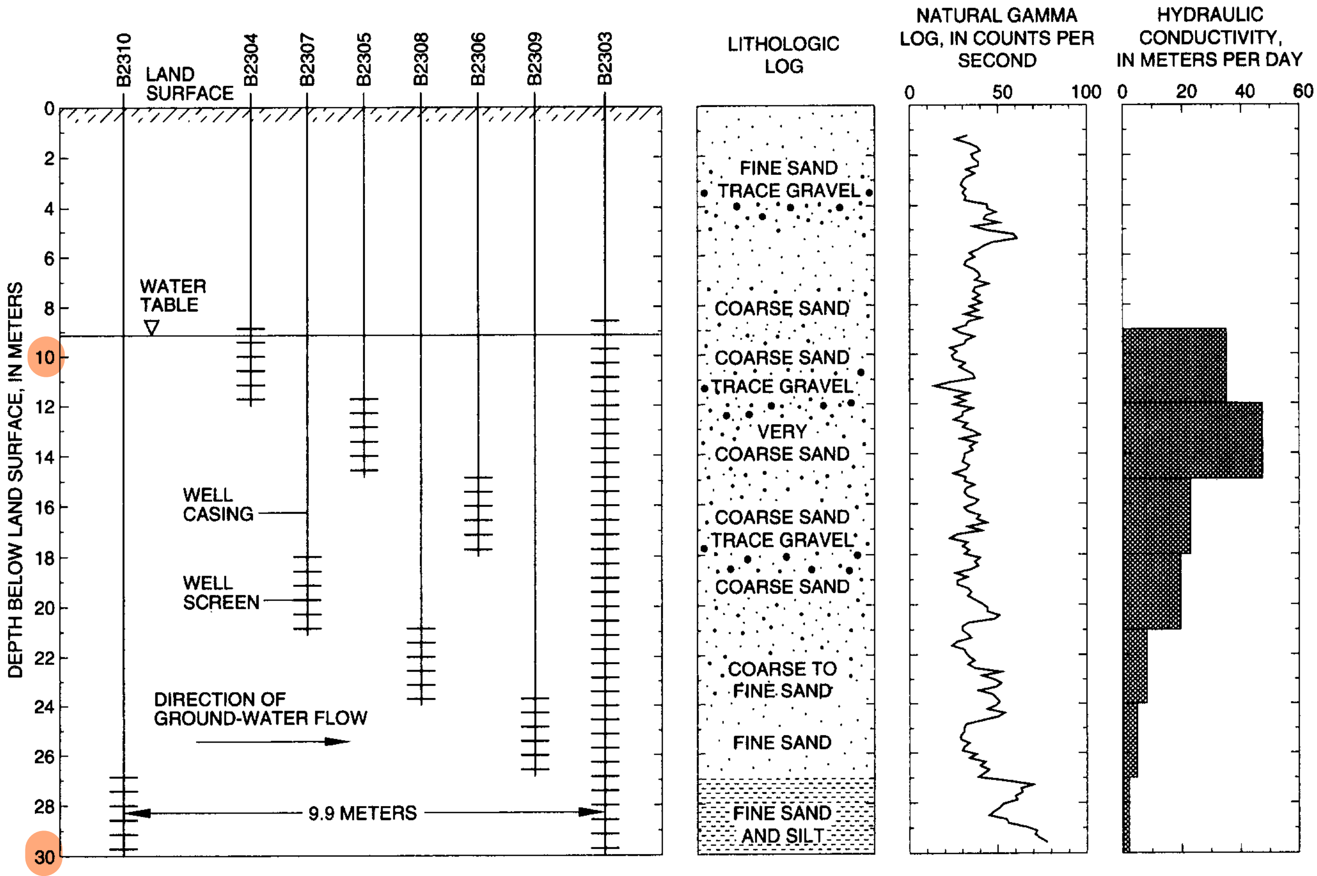

Fig. 2. Well construction, lithologic, natural-gamma, and hydraulic conductivity data for site B23 downgradient of Route 25 in southeastern Massachusetts.

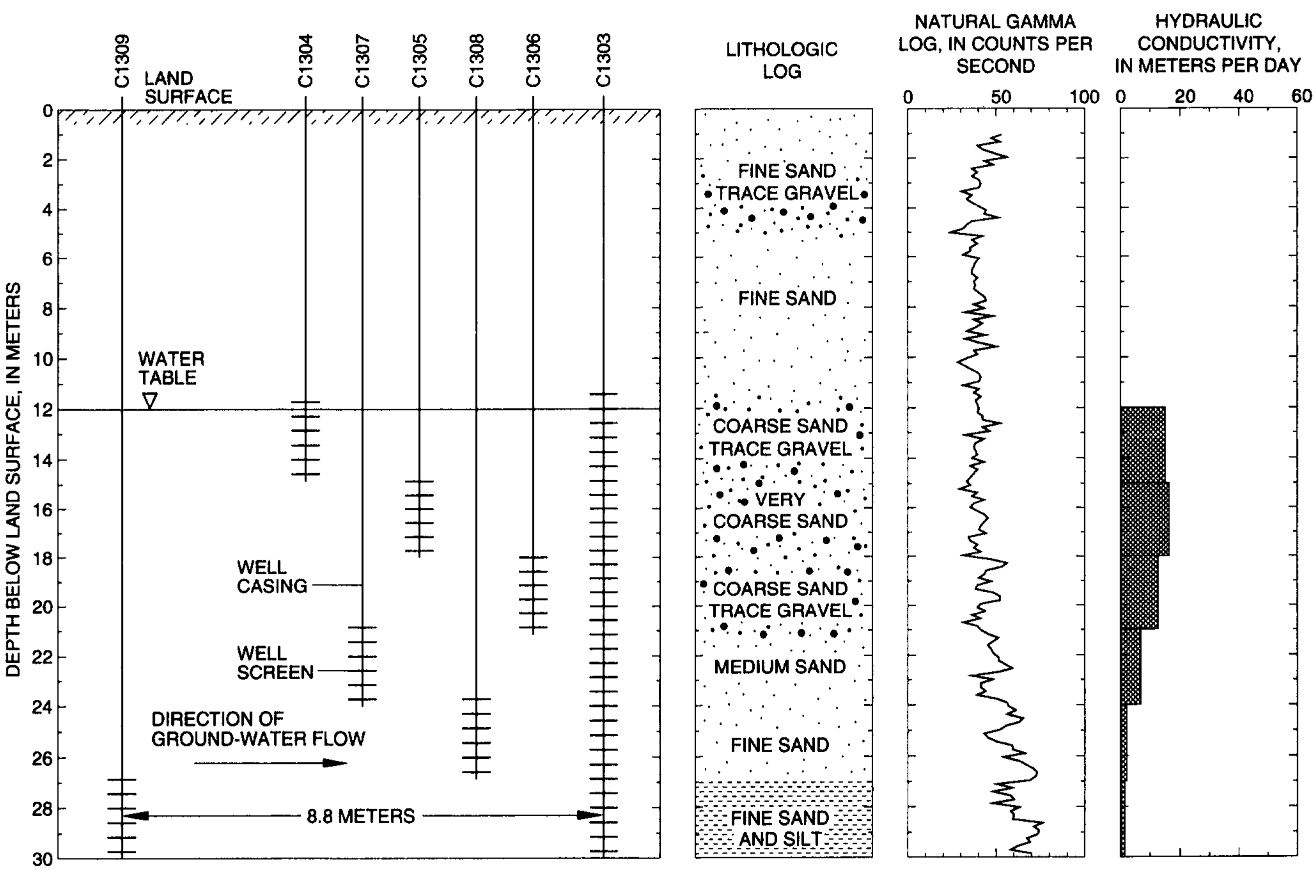

Fig. 3. Well construction, lithologic, natural-gamma, and hydraulic conductivity data for site C13 downgradient of Route 25 in southeastern Massachusetts.

well screen and casing. Filter packs and grout were not used in the installation of these wells.

Lithologic and natural gamma logs, coupled with grain-size distributions, indicate a general fining of sediments with depth, small scale stratification, and a decrease in hydraulic conductivity with depth as estimated by the Hazen approximation method (Figures 2 and 3). The natural gamma logs completed in the long-screen wells and in the deep short-screen wells are similar at each site. Although the natural gamma logs do not provide explicit evidence for discrete layers in the sand, the trend of higher radiation with depth corresponds with the lithologic and hydraulic conductivity data (Keys, 1990). The differences in the aquifer-test data were minor (horizontal hydraulic conductivities of 4-50 m/d) when compared to the range of values expected in a sand and gravel aquifer (horizontal hydraulic conductivities of 0.1 to 1,000 m/d). Therefore, the test sites are considered to be in a relatively homogeneous sand and gravel aquifer.

Methodology

A sampling program was designed to determine whether or not water samples collected from long-screen wells represent the actual quality of water in a relatively homogeneous, unconfined sand and gravel aquifer. Specific conductance and chloride, sodium, and calcium concentrations from water samples collected by two methods from long-screen wells are compared with water samples collected from adjacent vertical clusters of short-screen wells. By virtue of a borehole-induction logger's ability to measure changes in electrical conductivity in the aquifer caused by changes in chloride, sodium, and calcium concentrations (Church and Friesz, 1993b), induction log data were used as a standard to compare the analyses of water-quality samples collected from short-screen wells with those from long-screen wells.

Sampling Program

Water samples were collected monthly from January through June 1990 from the two well sites, each well site having one long-screen well and a cluster of short-screen wells installed to span the full vertical extent of the adjacent long-screen well. In addition, borehole induction logs and borehole-fluid conductance logs were obtained at each test site for salt-plume delineation. The sequence for collection of data from the well sites is as follows:

1. Measure static water level in long-screen and short-screen wells.
2. Perform fluid conductance log in long-screen well by use of a specific conductance meter.
3. Perform induction log in deepest short-screen well by use of an induction logger.
4. Collect water samples from discrete zones identified by the induction log in the long-screen well by use of a bladder pump and packers.
5. Collect a water sample from the long-screen well by use of a bulk-sampling method.
6. Repeat item #2.
7. Collect water samples from the short-screen wells with an electric submersible pump.

The borehole fluid velocities in the long-screen wells were measured once at the end of the study period in August 1991. Static water levels were measured in the long-screen and short-screen wells after the fluid velocities were measured.

Sampling Methods

Thermal-Pulse Flowmeter

The thermal-pulse flowmeter is used for borehole-fluid velocity logging. The instrument is designed to measure vertical velocities of water in a borehole. Developed by the U.S. Geological Survey (Hess, 1986), this electronic instrument measures borehole velocities by thermal-tag/trace-time techniques. A horizontal metal screen is heated by a pulse of electric current, which slightly heats a sheet or pulse of water that is moving in the borehole velocity field. Temperature sensors, one above and one below the heating element, register the direction and traveltime of the heated pulse of moving water. The thermal-pulse flowmeter is capable of detecting flows as low as 0.04 l/min and can detect velocities from 3 to 610 cm/min with a resolution of 0.9 cm/min (Hess and Paillet, 1990).

Borehole Induction Logging

An induction logger is used to measure electrical conductivity of the aquifer surrounding a well by inducing an electromagnetic field and measuring the aquifer response (Hearst and Nelson, 1985). Electrical conductivity in the aquifer is a function of soil type, porosity, and the concentration of dissolved solids in the pore fluid (Biella et al., 1983). Variations in electrical conductivity, either vertically or spatially in a well network, can be detected with induction logs. Variations in electrical conductivity of the pore fluid with time, such as variations caused by changes in concentration of chloride, sodium, and calcium (Church and Friesz, 1993b) or other dissolved constituents (DeSimone and Barlow, 1993), also can be detected by periodic logging of a single well or a well network.

The borehole-induction probe is insensitive to the conductivity of the borehole fluid and the formation immediately surrounding small diameter wells; the peak response of the instrument used is about 28 cm from the borehole axis (McNeill, 1986). This geophysical tool can be used in the test site wells that are constructed of polyvinyl chloride casing and screen. For the remainder of this report, the electrical conductivity of the formation measured by the induction logger is referred to as "induction conductivity."

Fluid Conductance Logging

Vertical profiles of the specific conductance of the borehole fluid are referred to as fluid conductance logs (Keys, 1990). A specific-conductance meter was used to manually record data as the probe was held at various depths in the well. Specific conductance typically was measured at 0.6-m intervals, and at shorter intervals when significant changes were detected over the 0.6-m interval.

Although the units of fluid-conductance logs are the same as induction logs (microsiemens per centimeter), the measurements are not equivalent in method or meaning.

The induction probe measures electrical conductivities of the aquifer outside the borehole; however, the specific-conductance meter measures electrical conductivities of the fluid between two electrodes on the probe in the well. For the remainder of this report, the electrical conductivity of borehole fluid or water-quality samples measured by a specific-conductance meter is referred to as "specific conductance."

Pumping Methods

The different hydraulic characteristics of the long- and short-screen wells created a need for three different pumping methods. The constituents studied, chloride, sodium, and calcium, are not so reactive that differences in pumping method would bias water-quality samples.

Bulk-water samples were collected from long-screen wells by use of a modified air-lift pump. An air compressor was used to force air into the annular space between two hoses secured at the bottom with a check valve. Water was forced up the inner hose in cycles as air pressure was applied and released. This method will be referred to as "bulk sampling." Water samples from discrete zones in long-screen wells were collected by use of a bladder pump with inflatable packers, referred to as "packer sampling." Inflatable packers were placed above and below the pump inlet to isolate zones in long-screen wells for selective sampling. These zones were selected by use of induction logs to locate and sample the contaminated zones as suggested by Giddings (1987). Compressed air was used to inflate the packers and actuate the bladder pump. Bulk-water samples were collected from the short-screen wells by use of an electrical submersible pump with a single inflatable packer. The inflatable packer was located just above the sample inlet in the well casing and was used to restrict the volume of water required to pump for collection of a water sample. This method of sampling was referred to as "cluster sampling."

Water-Sample Processing

The procedure for water-sample processing was independent of the type of well or pumping method. A minimum of three well bore volumes of water were evacuated while specific conductance of the evacuated water was monitored to assure that water samples pumped from wells came from the aquifer and not from stagnant water in the well bore. The sample was collected when the minimum volume was pumped and the specific conductance was stable. A minimum of five specific-conductance measurements were collected before physiochemical stability was assumed. One specific-conductance measurement was collected after each well volume was evacuated. After three full well volumes were evacuated, specific conductance was measured as each subsequent sampling churn was filled. Physiochemical stability was assumed once three consecutive specific-conductance measurements were consistent with an error of 2 percent. Water samples were prepared for laboratory analysis of specific conductance and dissolved chloride, sodium, and calcium concentrations at the U.S. Geological Survey, National Water-Quality Laboratory. The sodium, calcium, and chloride samples were filtered in the field using 0.45-μm pore-sized filters. The sodium and calcium samples were preserved with nitric acid. Samples for laboratory analysis of specific conductance were shipped unfiltered and unacidified.

Results and Discussion

Water-Level Measurements and Borehole Fluid Flow

Temporally averaged water levels from short-screen wells were compared to flow data from borehole flowmeter tests to define the hydrologic regime controlling fluid flow in long-screen wells. This is important to the visualization of the water-quality data because solutes are transported with the fluid. Water levels presented in Table 1 represent the

Table 1. Mean Water Levels and Mean Deviations from Water-Table Level, January Through June 1990, and Water Levels and Deviations from Water-Table Level, August 1991, from Short-Screen Wells at Test Sites B23 and C13 in Southeastern Massachusetts

Well	Screen Intervals: Below land surface	Screen Intervals: Altitudes	January-June, 1990: Mean water level altitude	January-June, 1990: Mean deviation from water table	August 1991: Water level altitude	August 1991: Deviation from water table
Site B23 Well Cluster						
B2304	9.0 to 12.0	3.2 to 6.2	5.574	0.000	5.249	0.000
B2305	12.2 to 15.2	0.0 to 3.0	5.561	-.013	5.243	-.006
B2306	15.2 to 18.2	-3.0 to 0.0	5.561	-.013	5.228	-.021
B2307	18.3 to 21.3	-6.1 to -3.1	5.564	-.010	5.237	-.012
B2308	21.2 to 24.2	-9.0 to -6.0	5.554	-.020	5.231	-.018
B2309	24.3 to 27.3	-12.1 to -9.1	5.541	-.033	5.219	-.030
B2310	27.4 to 30.4	-15.2 to -12.2	5.578	+.004	5.252	+.003
Site C13 Well Cluster						
C1304	12.0 to 15.0	3.0 to 6.0	4.671	0.000	4.301	0.000
C1305	15.6 to 18.6	-0.6 to 2.4	4.664	-.007	4.295	-.006
C1306	18.6 to 21.6	-3.6 to -0.6	4.677	+.006	4.313	+.012
C1307	21.5 to 24.5	-6.5 to -3.5	4.745	+.074	4.380	+.079
C1308	24.8 to 27.8	-9.8 to -6.8	4.763	+.092	4.402	+.101
C1309	26.5 to 29.5	-11.5 to -8.5	4.811	+.140	4.453	+.152

[All data are given in meters. Altitudes in meters above sea level.]

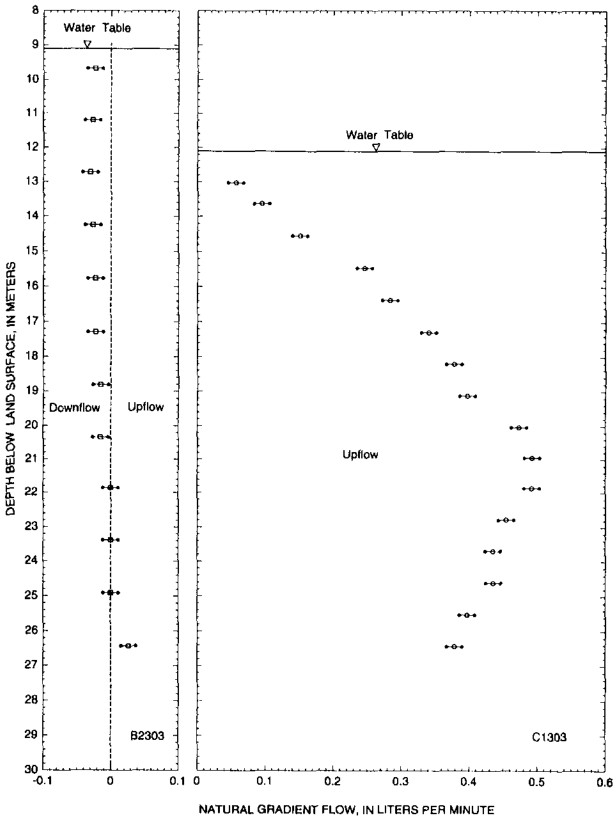

Fig. 4. Vertical well-bore flow obtained from heat pulse flow-meter measurements in long-screen wells B2303 and C1303 downgradient of Route 25 in southeastern Massachusetts, August 1991.

equilibration of flow losses as water moves into and out of the well bore. Thus, the volume of water gained or lost in each section of the screen is a function of the relative head and transmissivity in each layer, which induces a flow in the borehole and brings the system to equilibrium with atmospheric pressure. If there are no vertical head gradients, and therefore, no vertical flow in the aquifer, each head measurement, regardless of the depth, would equal the water-table level, as potential and pressure heads would be balanced. Water-level measurements from the vertical clusters of short-screen wells provide a vertical-head distribution that is not attainable in a single long-screen well.

The measured borehole flows (Figure 4) are consistent with the vertical head distributions in the short-screen wells (Table 1). At the B23 well cluster, decreasing head with depth causes a net downward flow in the zone from 9 to 18 m below land surface. The trends in head levels and flows, as measured, reverse in the zone from 18 to 27 m below land surface. A zone of slightly higher head causes an upward maximum flow in the zone from 27 to 30 m. The heads and borehole-fluid-velocity profiles indicate that the zones from 9 to 18 and 27 to 30 m deep are sources of water, and possibly contaminants into the borehole, whereas the zone from 18 to 27 m acts as a sink where water and contaminant could flow into the aquifer. The profiles in the C13 well cluster are less complex (Figure 4). The heads increase with depth and there is a strong upward flow in the entire length of the long-screen well bore. In this case, the lower zone from 21 to 30 m acts as a source, and the upper zone from 12 to 21 m below land surface acts as a sink.

Small vertical differences in head measurements, ranging from undetectable values to 0.15 m are sufficient to cause significant flows (as much as 0.5 l/min) in the well bore. This flow, occurring in what is considered a typical homogeneous, unconfined sand and gravel aquifer, is enough to exchange borehole fluid, and redistribute solutes in the aquifer downgradient from the long-screen well about once a day in well B2303 and about 20 times a day in well C1303.

Borehole Induction Logs and Water-Quality Data

Induction logs can be used to detect the arrival of contaminant plume (Desimone and Barlow, 1993) or detect and monitor changes with time in a contaminant plume if its electrical conductivity differs from that of the surrounding uncontaminated ground water (Church and Friesz, 1993b). The induction logger is sensitive to the electrical conductivity of the aquifer surrounding a well with the first response at about 0.15 m radial from the center of the borehole, a distance well beyond the zone disturbed by drilling and installation of a typical 5-cm-diameter monitoring well. The maximum response occurs at about 0.28 m. One-half of the induced response occurs beyond 0.58 m from the vertical axis of the well (McNeill, 1980). Therefore, the induction profile represents the conductivity of a coaxial cylinder of the aquifer outside of the borehole. At site B23, the volume of aquifer monitored by the induction log is a minimum of about 65 m^3 as compared to the borehole volume, which is 0.043 m^3.

Changes in the quality of water in the aquifer caused by winter highway salting have been measured monthly from 14 well sites using induction logs for two years prior to this study. These measurements have established the capability of this tool to accurately detect and delineate ground-water contaminant plumes containing dissolved concentrations of the constituents of road salt (chloride, sodium, and calcium). Therefore, the induction log, by virtue of its ability to measure conditions in the aquifer outside the borehole, is the standard to which each of the ground-water sampling methods were compared. The relation between the quality of water in the aquifer, as indicated by induction logs and specific conductance of water-quality samples collected by use of three pumping methods, are shown in Figures 5 and 6. The horizontal bars on these figures represent specific conductance, a function of total dissolved solids in the well bore, from the cluster-, packer-, and bulk-sampling methods. The thickness of each bar represents the vertical interval represented by each sample.

Specific conductance in the short-screen wells at site B23 closely resembles the profile of the induction log (Figure 5). Specific conductance was relatively high in the shallow well and was significantly lower in all the deeper wells. In contrast, the packer samples show a reverse profile where the highest specific conductance is detected at depth. The packer profile may be explained by the downward flow in the long-screen well (Figure 4), which redistributes the plume into the lower part of the aquifer. The relatively high

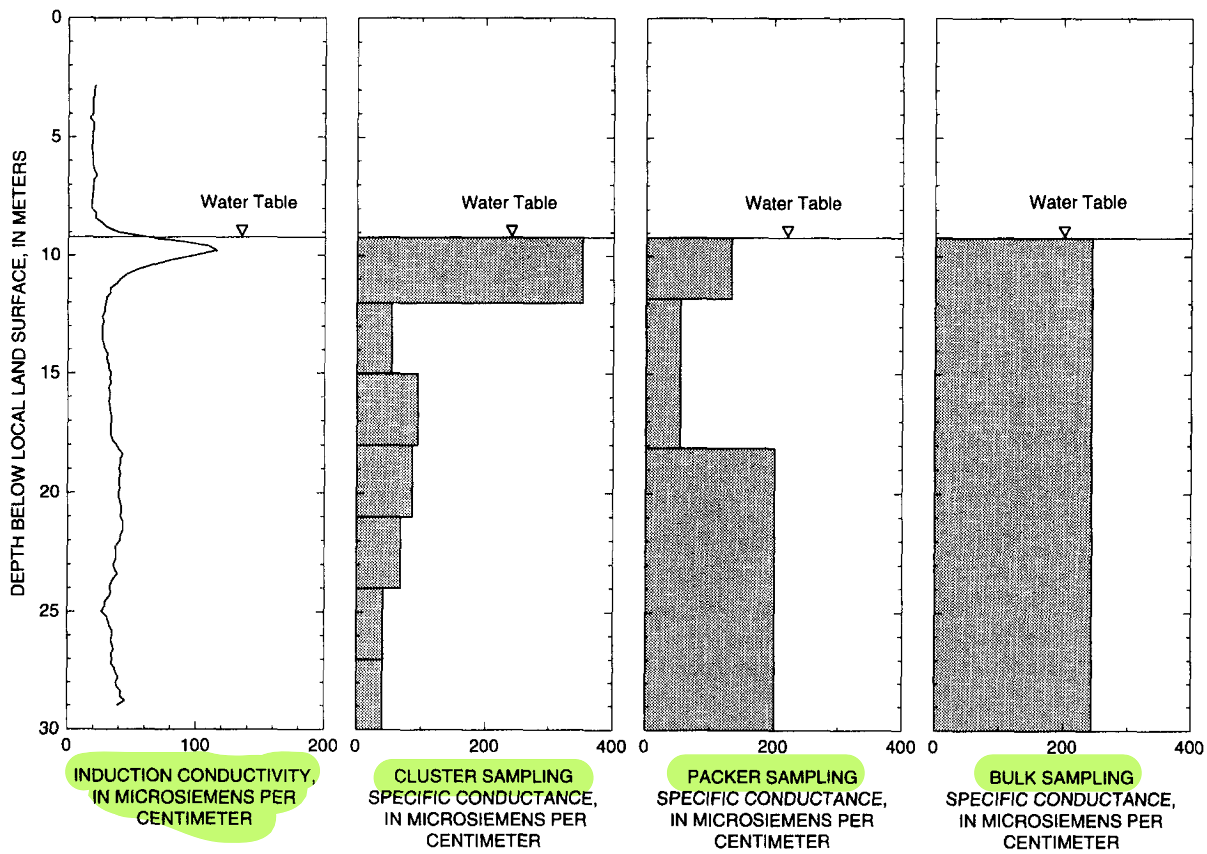

Fig. 5. Induction log and water-quality data from samples collected at site B23 downgradient of Route 25 in southeastern Massachusetts, April 1990.

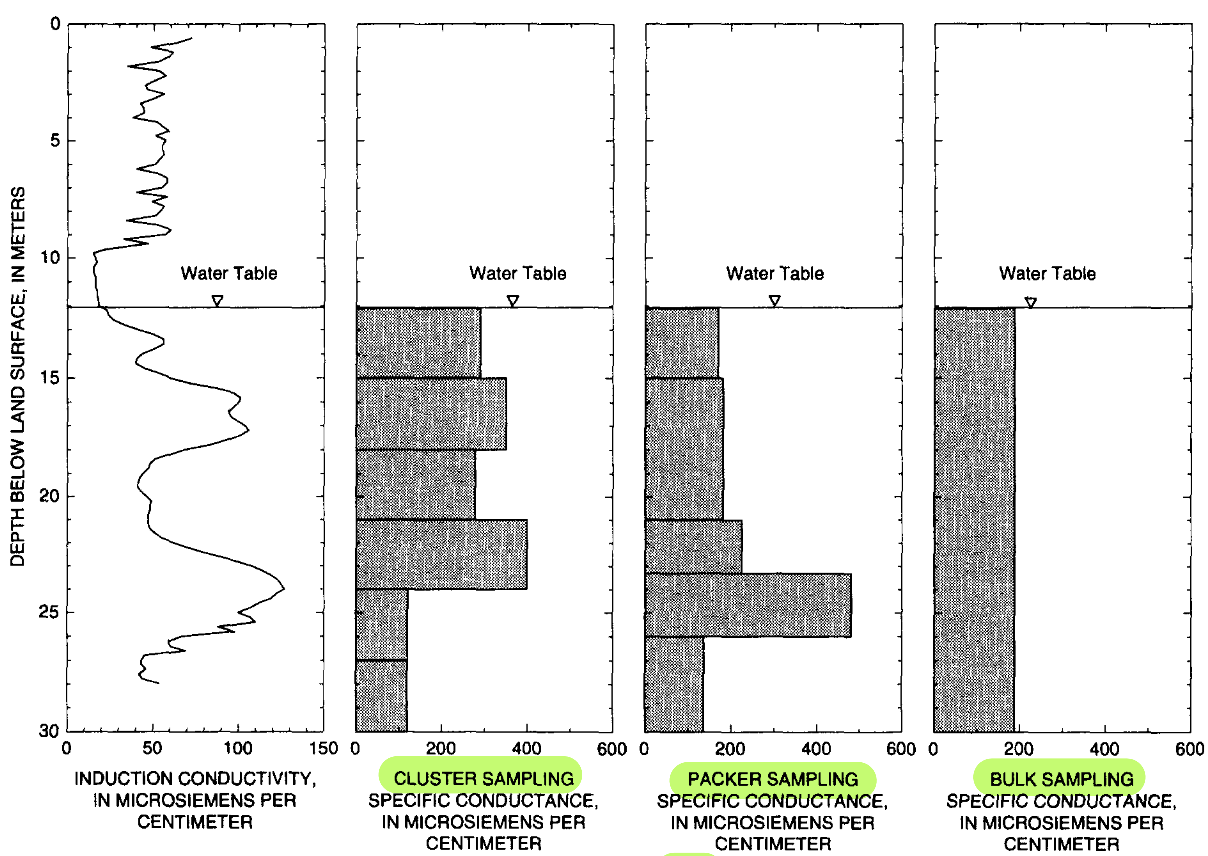

Fig. 6. Induction log and water-quality data from samples collected at site C13 downgradient of Route 25 in southeastern Massachusetts, April 1990.

specific conductance of the bulk-water sample is due to the redistribution of contaminated water from the upper part of the aquifer, down the well bore, and into the aquifer adjacent to the well. Water-quality sample data integrated over the length of the monitoring zone yields a screen-length weighted average (depth-integrated) value for each sampling method. Depth-integrated specific conductance in wells at the B23 site (Figure 5) during April 1990 was 98, 152, and 244 μS/cm from the cluster-, packer-, and bulk-sampling methods, respectively. Therefore, the depth-integrated specific conductances from packer and bulk samples are about 50 and 150 percent greater than that from the cluster sample.

Specific conductance in samples from the short-screen wells and at site C13 also closely resemble the profile of the induction log (Figure 6). Data from short-screen wells depict the bimodal distribution of specific conductance in the aquifer that is not evident in the results from the long-screen wells. In this case, the packer- and bulk-sampling methods used in the long-screen well fail to define the plume locations. Upward well-bore flow of water with low specific conductance from the lower part of the aquifer dilutes solutes in the upper part of the aquifer adjacent to the well. Depth-integrated specific conductance in wells at the C13 site (Figure 6) during April 1990 was 258, 216, and 188 μS/cm from the cluster-, packer-, and bulk-sampling methods, respectively. Therefore, the depth-integrated specific conductances from the packer and bulk samples are about 16 and 27 percent less than that from the cluster sample.

Monthly chloride concentrations from well clusters from January to June 1990 are shown in Figures 7 and 8 for sites B23 and C13, respectively. Chloride concentrations change significantly with time in the water-table well at site B23; however, little change is seen in the lower six wells (Figure 7). At site C, chloride concentrations in the first, second, and fourth wells from the water table show significant changes with time compared to the concentrations from the third, fifth, and sixth wells (Figure 8). The chloride concentration profiles at sites B23 and C13 are consistent with induction logs and show that the contaminant plumes are moving in discrete zones of the aquifer, and that, in this case, 3-m-long screens are the minimum resolution required to adequately characterize the quality of water in the aquifer.

Depth-integrated specific conductance and concentrations of chloride, sodium, and calcium (Figures 9 and 10) show that bulk and packer samples from the long-screen wells differ substantially from the water-quality samples from the cluster of short-screen wells. In well B2303, the small downward flow (Figure 4) redistributes solutes in the well bore and surrounding aquifer. Therefore, the screen-weighted average water-quality data obtained from the packer- and bulk-sampling methods in this long-screen well are consistently higher than the screen-weighted average

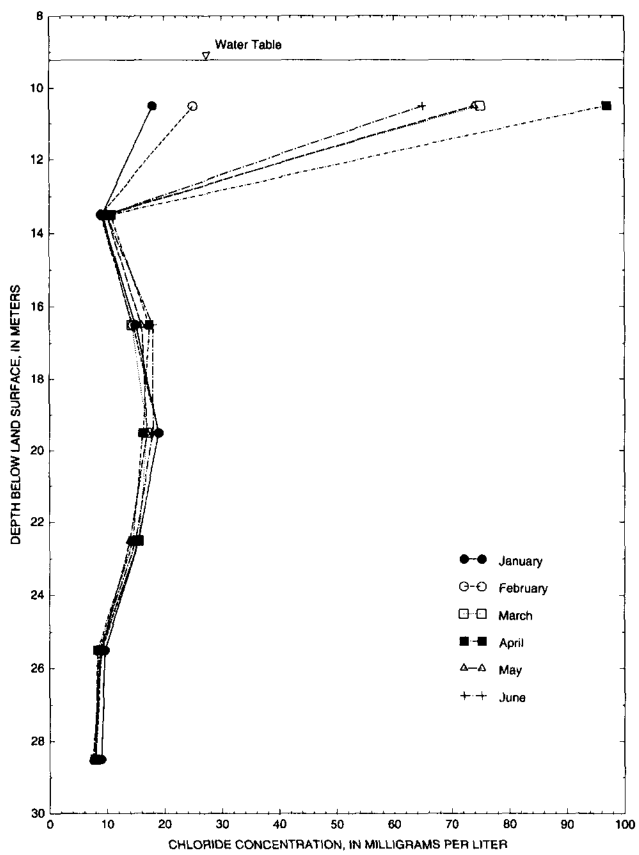

Fig. 7. Monthly chloride concentrations in the vertical cluster of short-screen wells at site B23 downgradient of Route 25 in southeastern Massachusetts, January through June 1990.

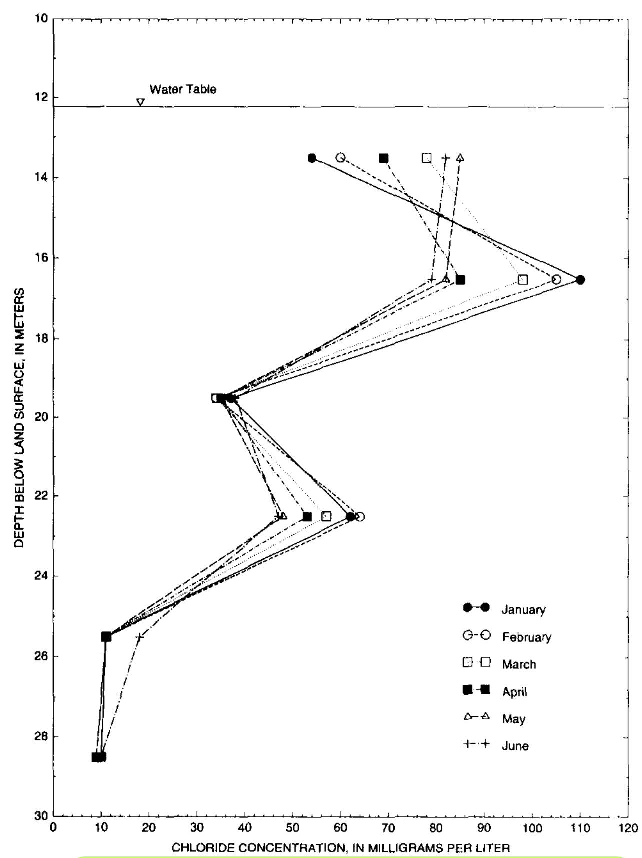

Fig. 8. Monthly chloride conentrations in the vertical cluster of short-screen wells at site C13 downgradient of Route 25 in southeastern Massachusetts, January through June 1990.

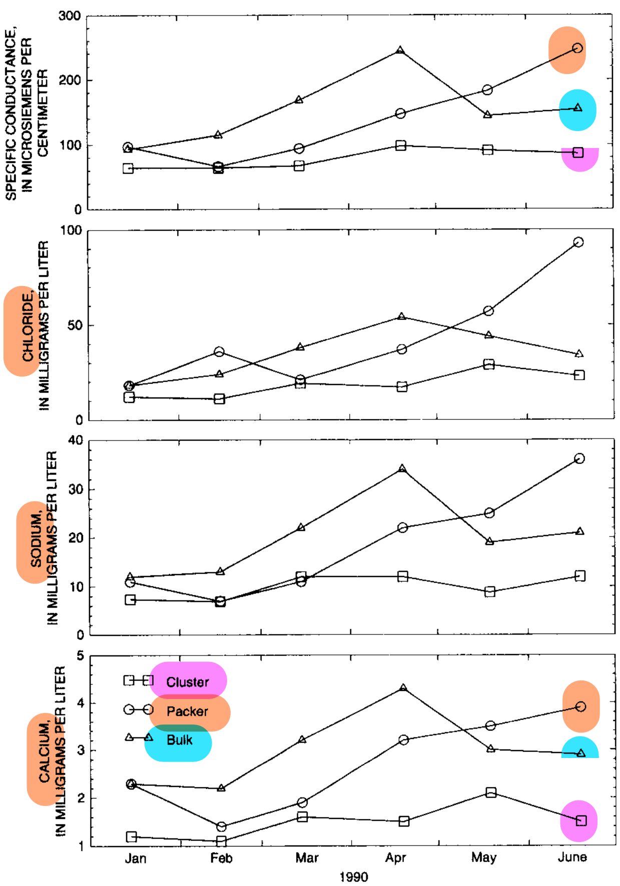

Fig. 9. Differences in measured water-quality constituents using cluster sampling in short-screen wells, and packer and bulk sampling in long-screen wells at site B23 downgradient of Route 25 in southeastern Massachusetts.

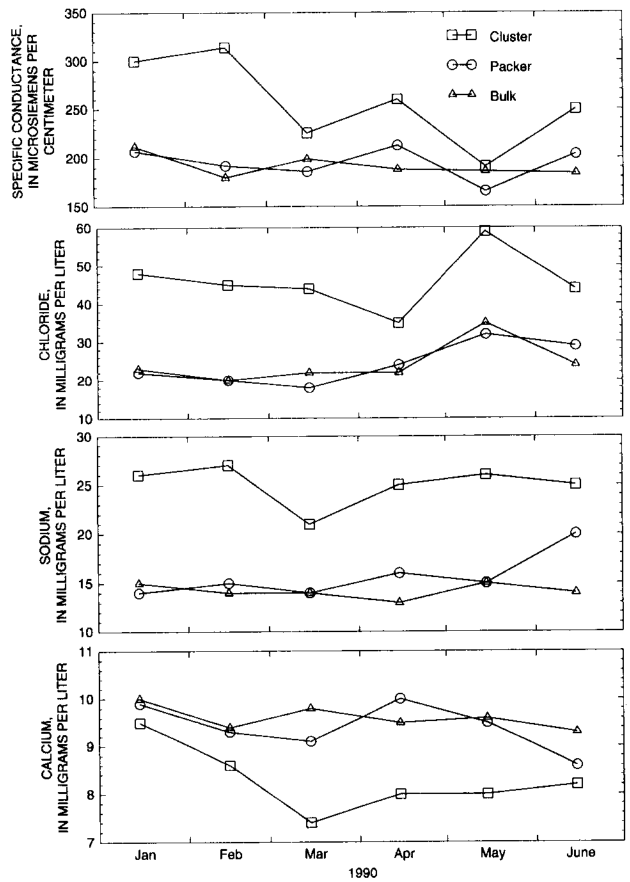

Fig. 10. Differences in measured water-quality constituents using cluster sampling in short-screen wells, and packer and bulk sampling in long-screen wells from site C13 downgradient of Route 25 in southeastern Massachusetts.

values for the adjacent cluster of short-screen wells (Figure 9). The upward flow (Figure 4) and consequent dilution of contaminants in well C1303 bias the water-quality samples collected from the long-screen well. Screen-weighted average specific conductance and concentrations of chloride and sodium (Figure 10) are underestimated as a result of this bias. In this case, the calcium data seem anomalous. Calcium concentrations at this site are not significantly different from background concentrations; therefore, the redistribution of calcium cannot be measured.

Borehole-Induction and Fluid-Conductance Logs

Borehole-fluid-conductance logs are more commonly used for plume detection than induction logs because the cost of the equipment required is two orders of magnitude less. It was economics that prompted Giddings (1987) to say, "Dilution below a detection threshold is often raised as an objection to construction of detection monitoring wells with long-screen lengths. Electrical conductance profiling of the monitoring well can be utilized to delineate zones of contaminant concentration, and then discrete interval sampling devices can be utilized to sample those zones of interest." In preparation for this experiment, induction logs were compared with fluid-conductance logs to plan the implementation of discrete zone sampling in the long-screen wells. Results show that fluid-conductance logs may not represent surrounding aquifer conditions in the long-screen wells. Pumping of about 20 well volumes of aquifer water in the long-screen well at site B23 was required before the specific conductance approached steady state. At this point, the fluid-conductance profile approximated that of the induction log (Figure 11).

The small vertical head gradient in the aquifer at this site, about 0.002, is sufficient to transport one well volume of water per day from the upper to lower part of the aquifer, about 43 l/d. The almost constant fluid-conductance profile evident before pumping (Figure 11) reflects this continuous transport of contaminated water in the well bore. The effects of flow in the long-screen well were temporarily reversed by pumping a large volume, about 870 l, of water from the well. The fluid-conductance log, done immediately after pumping, approximates the induction log, which monitors the conductance of the fluid outside the well.

Advection, caused by measurable head differences in the aquifer, dominates solute transport in the long-screen well bores at the study sites used in this experiment. However, Reilly et al. (1989) showed numerically that well-bore flow is significant in long-screen wells due to head gradients that are not measurable because they are smaller than can be detected with current instrumentation. In addition, mixing of water in the well bore may result from molecular diffusion of solutes or flow due to density or temperature gradients, factors not measured in this experiment. Therefore, chemical concentration differences in the borehole fluid might be obscured even where net borehole flow due to vertical head gradients is not significant and specific conductance logging in a long-screen well may be as misleading as the different methods for sampling such installations.

Conclusions

The results of this study, demonstrated by field experimentation, are summarized as follows: (1) Use of long-screen wells will not provide representative water-quality samples, even in a relatively homogeneous, unconfined sand and gravel aquifer. The quality of water samples collected from the long-screen well at site B23 overestimates the extent of contamination in the aquifer caused by downward flow of solute from a contaminated zone near the water table. Upward flow in the long-screen well at site C13 masks a bimodal distribution of solutes as a result of the redistribution of relatively pure waters in the zone of contamination. (2) Vertical flow in long-screen wells may contaminate zones of the aquifer that would not otherwise become contaminated in the absence of the long-screen wells. For example, at site B23, the long-screen well is a conduit for the downward flow and redistribution of contaminant-laden water. (3) Use of borehole-induction logs from fully cased wells for plume detection, and accurate placement of short-screen or multilevel sampling wells as needed for water-quality sampling, are viable alternatives to the use of the long-screen well in contaminant-transport studies. The ability of the induction logger to measure changes in water quality in a large volume of the aquifer adjacent to a monitoring well, without the need of a continuous screen, is ideal for detecting the arrival of a contaminant plume. As the plume develops, periodic induction logs will define the zone(s) of contamination. This information may be used for accurate placement of short-screen monitoring wells or multilevel samplers for detailed water-quality sampling. Therefore, if the electrical conductivity of the contaminant plume differs from that of the surrounding uncontaminated ground water, these methods will provide the data required to define the plume with a much lower risk of sample bias, failure of detection, and cross contamination of the aquifer inherent in long-screen well design.

The results of this study agree with predictions of well bore flow caused by small vertical head gradients in unstressed long-screen wells in a relatively homogeneous, unconfined sand and gravel aquifer on the basis of work using a numerical model (Reilly et al., 1989). Results of the field experiment and the numerical model study suggest that long-screen wells will fail even in a relatively ideal setting, and therefore, cannot be relied upon for accurate measurements of water-table levels, collection of water-quality samples, or fluid-conductance logging.

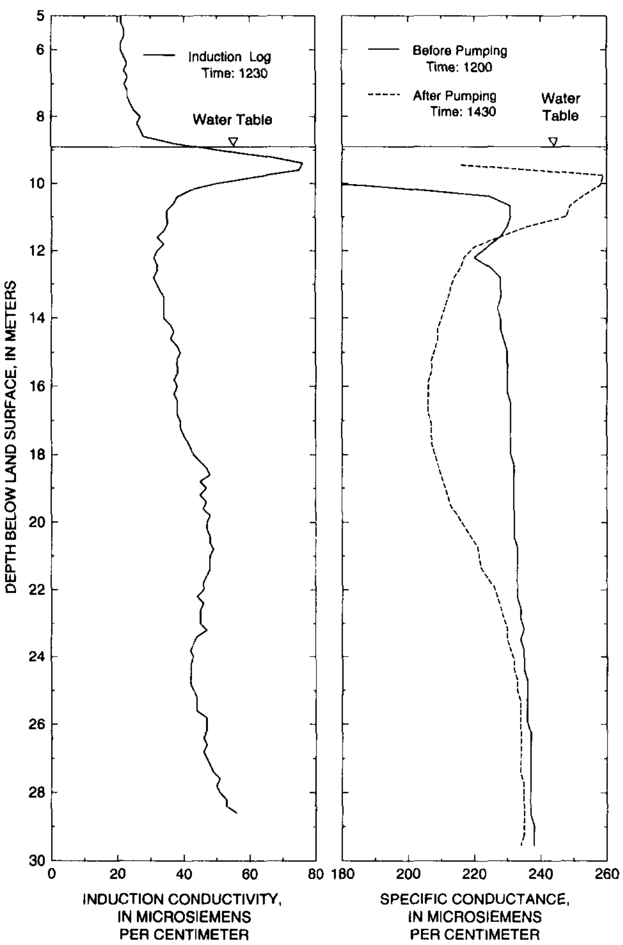

Fig. 11. Induction conductivity from borehole induction logs and specific conductance logs of borehole fluid done before and after pumping long-screen well B2303 downgradient of Route 25 in southeastern Massachusetts, May 5, 1989.

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