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SINGLE-WELL TRACER METHODS FOR HYDROGEOLOGIC EVALUATION OF TARGET AQUIFERS

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ABSTRACT

Designing an efficient well field for an aquifer thermal energy storage (ATES) project requires measuring local ground-water flow parameters as well as estimating horizontal and vertical inhomogeneity. Effective porosity determines the volume of aquifer needed to store a given volume of heated or chilled water. Ground-water flow velocity governs the migration of the thermal plume, and dispersion and heat exchange along the flow path reduces the thermal intensity of the recovered plume. Stratigraphic variations in the aquifer will affect plume dispersion, may bias the apparent rate of migration of the plume, and can prevent efficient hydraulic communication between wells. Single-well tracer methods using a conservative flow tracer such as bromide, along with pumping tests and water-level measurements, provide a rapid and cost-effective means for estimating flow parameters. A drift-and-pumpback tracer test yields effective porosity and flow velocity. Point-dilution tracer testing, using new instrumentation for downhole tracer measurement and a new method for calibrating the point-dilution test itself, yields depth-discrete hydraulic conductivity as it is affected by stratigraphy, and can be used to estimate well transmissivity. Case study data from a Tuscaloosa, Alabama, ATES project site show that tracer methods can be used to detect vertical flow in well bores caused by hydraulic interaquifer communication. Finally, current research shows that single-well tracer methods may be useful in estimating longitudinal dispersivity as well as both thermal and solute-retardation coefficients.

INTRODUCTION

Hydrogeologic aquifer characterization is the estimation of ground-water flow patterns in three dimensions and time. Characterization requirements often include estimating aqueous mass transport parameters that govern movement of ground-water solutes, suspended matter, or thermal plumes. Such characterization is legally mandated for environmental monitoring and cleanup of waste disposal sites, is beneficial for water resources investigations, and is critical for engineering aquifer thermal energy storage (ATES) projects. Despite need or mandate, adequate characterization is seldom achieved, because conventional field methods for hydrologic investigation of flow parameters other than transmissivity and hydraulic gradient tend to be prohibitively expensive. For example, dual- or multiwell geochemical tracer tests are useful for direct measurement of ground-water flow rate. However, this method requires extensive sampling and analysis of ground water (often for weeks or months), and may require drilling observation wells otherwise unneeded. Further, the effects of dispersion, dilution, and sometimes adsorption or chemical reaction on the tracer itself can render the tracer tests very difficult to interpret.

Because of the expense of field methods, laboratory examination of rock or sediment samples collected during well drilling often is performed to estimate flow parameters such as effective porosity and solute retardation. Unfortunately, the laboratory data are often inaccurate because collecting geologic samples from the subsurface, especially poorly consolidated sediment samples, can irrevocably alter both physical and chemical characteristics or the material.

Single-well tracer methods and related instrumentation developed in recent years offer economic means for field-estimating ground-water velocity, effective porosity, and the vertical distribution of hydraulic conductivity. Hall et al. [1] showed that by combining the drift-and-pumpback, single-well tracer test presented by Leap and Kaplan [2] with conventional pumping tests and hydraulic gradient measurement, both effective porosity and ground-water flow velocity can be estimated. Hall and Raymond [3] described a new method for performing and interpreting point-dilution tests. This test yields an estimate of hydraulic conductivity as a function of depth, and has been useful in detecting and quantifying vertical currents in wells.

These single-well methods were developed as part of Pacific Northwest Laboratory's* Seasonal Energy Storage program, and were originally tested and demonstrated at three ATES project sites in Tuscaloosa, Alabama. The methods have been useful for providing field data for the design or modification of ATES well fields, and for predicting well field performance using computer modeling. Recent experience, new information, and improved instrument design have increased the efficacy of these tests.

In this paper, both the power and limitations of the drift-and-pumpback and point-dilution tests are re-examined with respect to recent experience, and within the context of field studies. The case study data are taken from two Tuscaloosa ATES project sites and from an experimental *in situ* bioremediation area at the Hanford Site in Washington State. The improved equipment and instrumentation used in conducting the tests are presented.

Even within the limited acreage represented by an ATES well field, the transmissivity of wells can vary significantly. The principal means for measuring transmissivity is the constant-discharge pumping test, but it is not necessarily economic or efficient to conduct such tests while each well is completed, developed, and proved. More often than not, economics dictate that formal pumping tests are conducted at only a fraction of the wells in a network. Thus, a method for rapidly estimating the transmissivity of completed wells that does not require a workover rig to set a pump could be useful. Such a method, based on point-dilution testing, is proposed and described here.

Ground-water velocity, effective porosity, and the vertical distribution of hydraulic conductivity do not completely define the movement of a thermal plume. Knowledge of two other flow parameters, dispersivity and thermal retardation, need to be quantified to predict the rate of plume movement and the loss of thermal intensity that is a necessary result of dispersion during movement. Evidence suggests that single-well, injection-withdrawal tracer methods can be used to estimate these parameters as well as the retardation of nonconservative chemical species in ground water. (This latter application may be important to ATES systems that require require water treatment.) This evidence is presented with respect to its potential for characterizing target aquifers.

IMPROVED APPARATUS

The program of tracer testing introduced by Hall and Raymond [3] includes both a drift-and-pumpback test and a point-dilution test. In that program, a single placement of a tracer into a well bore serves as the start of both tests. Both tests depend on quickly and evenly introducing a flow tracer into the screened interval of the test well; the point-dilution test further depends on downhole instrumentation suitable for monitoring tracer concentration as a function of both depth and time. In this section, recent improvements in equipment used for tracer introduction and measurement are described.

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Most target aquifers for ATES projects will be dominated by horizontal advective flow, and the need for evenly distributing a ground-water flow tracer in the water column of a test well is intuitively self-evident. For example, most of the ground-water flow at one of the Tuscaloosa ATES project sites occurs in the approximate lower half of the target aquifer [3]. At that site, if a disproportionately large amount of the tracer were initially confined to the upper half of the aquifer, the results of a drift-and-pumpback test (designed to yield an estimate of net flow within the saturated interval) would be biased toward flow velocity of the upper half.

The original injection apparatus described by Hall and Raymond [3] was a weighted and sealed plastic jug suspended from a flexible 1.6-cm inner-diameter (ID) flexible rubber hose with both ends open. To introduce tracer to the water column, the contained volume of the hose from the lower open end, as it would be situated at the bottom of the well, to the top of the water column was first calculated. The tracer salt (lithium bromide) was then dissolved in that volume of distilled water. The hose was lowered to the bottom of the well, and the tracer poured into the upper end of the hose. The tracer solution displaced the ground water from the hose. When the hose was retrieved from the well, the tracer drained from the hose and was left in place. The weighted jug mixed the tracer in the well bore.

Three problems with this apparatus were identified. First, stretching of the rubber hose, caused by the combined weight of jug and the hose itself, progressively reduced the inner diameter from the lower end to the surface. Thus, the hose contained somewhat more tracer near the bottom than near the top of the water column, and the volume of tracer solution represented by the difference between the calculated hose volume and the volume of the stretched hose was deposited in the well bore near the lower open hose end. This problem has since been eliminated by using nylon flexible pipe, which does not stretch significantly.

Second, field data indicated poor lateral mixing by the weighted jug within the well bore using this method. In recent field experiments, the jug has been replaced by a cylindrical, two-stage, in-line mixer having an outer diameter slightly less than the inner diameter of the well. The mixer is suspended from the nylon hose by a yoke. When the hose and mixer assembly is retrieved from the well, tracer solution flows from the lower end of the hose directly to the center of the upper opening of the mixer. The internal paddles of the mixer split and recombine the tracer and the well water, leaving the tracer sufficiently distributed laterally as well as vertically within the well bore.

Mixers of this type, having no moving parts, are commonly used in chemical processing and similar industries for mixing flowstreams, and are commercially available. Generally termed "static in-line mixers," they are installed and used directly in process piping, and usually have more than the two mixing stages employed in the previously described circumstance.

The third problem related to the difference in density between ground water and the tracer solution. The tracer introduced into the test well at one of the Tuscaloosa ATES sites [3] was 125 g lithium bromide mixed into 2.2 L of distilled water; the resulting 0.65 M solution had a density of approximately 1.04 g/mL as compared to the nominal 1.00 g/mL density of ground water. Because of the density difference, the top of the column of tracer solution within the injection hose settled somehat below the top of the ground-water column in the well bore–in this case by 4%. Thus, only the lower 96% of the well-water column received tracer. Excess tracer (like that resulting from hose stretch) was immediately expelled from the lower end of the injection hose.

The problem of density has been overcome since primarily by using a larger hose diameter; the nylon hose used in more recent experiments has an inner diameter of 2.5 cm. With the larger diameter, the volume of water intercepted by the hose for a given well depth is more than 2.5 times that intercepted by the 1.6-cm ID hose. With the larger diameter hose, the concentration of the aforementioned tracer charge would have been 0.26 M, with a density of approximately 1.01 g/mL, and a correspondingly smaller portion of the well water-column that cannot receive tracer.

Despite these better means for introducing tracer to the well, further improvement is warranted. The current method requires that, for reasonable convenience in the field, the tracer injection hose be cut to no more than a few feet longer than the total depth of the well. This stipulation is inconvenient when wells of different depths must be tested. Further, the author's experience in using such equipment in 90-m-deep test wells has yielded convincing evidence that the method is best suited to young and robust investigators. A prototype for a reel-deployed tracer injector that does not require cut-to-length hose, and that avoids problems caused by density differential, has been designed. Construction and testing of this apparatus is anticipated soon.

Bromide solutions often have been used as ground-water tracers because the bromide ion behaves conservatively and is foreign to freshwater aquifers and surface water. Multiple, simultaneous point-dilution tests in a single well have been made possible by the development of submersible ion-selective electrodes (ISEs) suitable for *in situ* measurement of bromide ion as a function of both depth and time [3, 4]. In this analytical method, a reference electrode and a bromide-sensing electrode are immersed in a bromide solution of unknown concentration. The voltage potential developed between the electrodes is a function of bromide concentration, and may be measured with a high-input impedance potentiometer (e.g., $10^{12}\ \Omega$).

However, prototype electrodes first used at one of the Tuscaloosa ATES sites were inconvenient to use, and because of difficulty in measuring the high-impedance signal along more than approximately 30 m of cable, collection of bromide data from deeper wells was difficult. Also, these first electrodes were of crude and cumbersome construction, and attempts to use them in 10-cm ID monitoring wells at the Hanford Site showed that moving the electrodes up and down the well bore induced vertical mixing [5] of the bromide tracer, thus invalidating point-dilution testing. (This was not a problem in the 25-cm ID wells at the Tuscaloosa ATES sites.)

The bromide-sensing apparatus has been improved in three ways. Both the reference and sensing electrodes have been combined into a single streamlined probe body 1.6 cm in diameter and 20 cm long. Recent field experiments have shown that this probe configuration does not induce significant vertical mixing in a 10 cm ID well bore [5]. Also, the probe body now includes an onboard unity gain signal amplifier to reduce signal impedance, thus eliminating depth limitations. Finally, the tendency for the capillary tube of the original electrode design to lose fluid and develop an open circuit has been eliminated.

FIELD EXPERIMENTS

Experience in conducting both drift-and-pumpback and point-dilution tests at three different test sites has yielded important information that highlights both the power and the limitations of the single-well tracer methods. These sites are the University of Alabama Student Recreation Center (UASRC) ATES well field and the VA Medical Center (VA) ATES well field, both located in Tuscaloosa, Alabama, and the Hanford bioremediation test site north of Richland, Washington.

Test results for these sites show distinctly different patterns. The UASRC test results are most satisfactory; the results have been previously reported [3] and are included here as a basis against which to compare results from the other test sites. The VA site illustrates anomalous and misleading results caused by the unanticipated presence of a perched aquifer immediately overlying the target aquifer. The Hanford test site illustrates the effects of local inhomogeneity on point-dilution test results, and of extremely low natural ground-water flow velocity on a drift-and-pumpback test. The target aquifer at each of these sites is unconfined (except for local confinement by the perched zone at the VA site), and consists of essentially similar, poorly sorted, unconsolidated sediments.

Of most importance here are the data patterns that emerged from the test programs, some of which can be considered diagnostic of the problems encountered at the VA and Hanford test sites. However, before describing the test results, it is appropriate to briefly review the test methods and interpretation.

Test Methods

In an aquifer dominated by horizontal advective ground-water flow, a drift-and-pumpback test can be combined with conventional hydrologic field methods to yield estimates of ground-water flow velocity and effective porosity [1]. The hydrologic methods usually include a pumping test to determine the transmissivity of a test well, and water-level measurements to determine local hydraulic gradient. A flow tracer (e.g., bromide) is introduced into the test well and allowed to drift for a period of time, often a few days. The well is then pumped to recover the tracer. Ground-water flow velocity, V, is determined as

$$V = Qt/\pi b T^2 KI \qquad (1)$$

where Q = pumping discharge rate
t = pumping time to recover center of mass of tracer
b = aquifer thickness
T = drift time plus t
K = hydraulic conductivity
I = hydraulic gradient

After flow velocity has been determined, the effective porosity, n, can be calculated from Darcy's law as

$$n = KI/V \qquad (2)$$

The tracer introduction for the drift-and-pumpback test also can be used to conduct the point-dilution test by using the ISEs to measure tracer concentration as a function of both depth and time. For a given depth, the slope of a plot of the electrode response in millivolts *versus* time is directly proportional to the mean rate of ground-water flow through, and normal to, the axis of the well bore [3, 4]. In a valid point-dilution test, the plot slope must be a straight line, although there is often some curvature, caused by initial inefficient mixing, in the early stage of testing.

Recent work has shown that in a minimally developed well, the flux through the well bore should be nearly proportional to hydraulic conductivity at each test depth [6]. It must be emphasized that "minimal development" should not be interpreted as "poor development." Development must be sufficient to ensure that the hydraulic conductivity of the well screen and artificial filter packing material is much greater than that of the surrounding aquifer, and that any skin effect caused during the drilling of the well (e.g., by mud cake) is negated. A minimally developed well is one that has not been pumped enough to remove a significant amount of fines from the surrounding aquifer; that is, the hydraulic conductivity of the aquifer immediately adjacent to the well bore has not been altered.

If conditions are met for a minimally developed well, the slopes of the plots of electrode response *versus* time for the test depths in a well will be proportional to hydraulic conductivities at those depths. Because the net hydraulic conductivity of the screened interval of the well is known from pumping tests, relative hydraulic conductivities (slopes) can be converted to absolute values by back-calculating using weighted averages.

It is important to note that the mean flow velocity through the well bore, V*, at any given depth only indirectly reflects the rate of ground-water flow through the aquifer at that depth. This velocity is influenced by effective porosity and a flow distortion factor [7] according to

$$V^* = Vna \qquad (3)$$

where

V = flow velocity within the aquifer
n = effective porosity
a = flow distortion factor

The flow distortion factor arises because the hydraulic conductivity of the well installation is much greater than that of the aquifer, therefore causing the flow net in the horizontal plane to converge towards the well on approach, and to again diverge downgradient from the well. It is generally accepted that the flow distortion factor is determined by well construction [6, 7]; wells having identical construction should have the same flow distortion factor.

University of Alabama Student Recreation Center

The UASRC ATES project has operated since 1985 [2], so it is unlikely that point-dilution results accurately reflect the vertical distribution of hydraulic conductivity, because the wells are past being "minimally developed." With that single caveat, however, field data from the tracer testing program illustrate near-ideal test results.

The saturated aquifer thickness at the site was 11.3 m. Hydraulic gradient was 0.0045, and the hydraulic conductivity of the test well was 25 to 28 m/d based on constant-discharge pumping tests. A tracer consisting of 125 g lithium bromide was introduced to the test well, allowed to drift for slightly more than 2 d, and recovered by pumping at 227 L/min. The center of mass of the tracer was recovered after 51 min; the tracer plume was virtually completely recovered after 4 h. Net ground-water flow velocity was calculated to be 0.6 to 0.7 m/d, and effective porosity was 16 to 21%. Figure 1 shows the results of recovery pumping.

Tracer introduction for the drift-and-pumpback test also was used as the beginning of a point-dilution test. The total test duration was 6 h. Figure 2 includes plots of electrode response *versus* time at all test depths; note that each plot tends toward a straight line.

After normalizing the plot slopes to that of the 6.9-m depth, which had the steepest slope observed, the relative rates of ground-water flow through the well bore were plotted as a function of depth. The resulting plot is compared in Figure 3 to sediment stratigraphy derived from samples collected during well drilling. This comparison shows that most ground-water flow occurs in a 4.6-m-thick sand layer approximately representing the lower half of the aquifer.

VA Medical Center

In February 1993, Pacific Northwest Laboratory staff conducted hydrologic testing at the VA site to provide design data for an ATES well field. At the principal test well, H3, static water-level measurements indicated that 24 m of saturated sediment overlay the Pottsville Formation, which consists of well-indurated shales and limestones and is the regional lower boundary of the unconfined aquifer. Total depth to the Pottsville Formation was 65 m. The 25-cm ID well fully penetrated the saturated interval, which was presumed at the time to represent the regional unconfined aquifer.

A constant-discharge pumping test indicated a transmissivity of 619 m^2/d for the entire saturated interval. Static water-level measurements at H3, one other 25-cm ID well, and two 5-cm ID observation wells were used to prepare a water-table map. The map showed that the hydraulic gradient in the vicinity of H3 was 0.0035. The measured well transmissivity and gradient are consistent with hydraulic properties found at other Tuscaloosa test sites [1, 3]. However, repeated measurements at three other observation wells, at distances from H3 of 46, 131, and 183 m, displayed anomalously high static-water levels. Compared to the prepared water-table map, the anomalies were 11.9, 4.9, and 13.4 m, respectively. Water levels from two remaining observation wells, 23 and 38 m distant from H3, were unstable, but consistent with the water-table map.

The 5-cm ID wells were constructed to fully penetrate the saturated sediments, but were developed only by brief air-lifting. It was assumed that poor development had caused insufficient hydraulic communication between the well bores and the sediments, and that the anomalous water levels may have been caused by inadvertent infiltration of surface water into the wells. (In other words, for lack of a better explanation, the anomalies were ignored.)

A drift-and-pumpback test was initiated in H3 by introducing 250 g of lithium bromide into the well using a rubber hose and weighted jug as described previously. The tracer was allowed to drift for 4.05 d, and recovery pumping was started at the rate of 303 L/min. A conventional bromide ISE was used for monitoring the bromide concentration of the effluent stream. The initial bromide concentration was 0.4 mg/L, and after 1 h of pumping the concentration decreased to 0.3 mg/L. The pumping rate was increased to 420 L/min, the maximum discharge of the pump. Recovery of the main tracer plume did not begin for another 90 min. The center of mass of the tracer plume was recovered after 358 min of pumping, and the pump was stopped 45 min later.

The tracer recovery curve is shown in Figure 4. Application of equations (1) and (2) yielded a ground-water movement rate equal to 1.2 m/d, and an effective porosity of 7.8%. These values are comparable to those found at another Tuscaloosa test site [1], where the estimated flow velocity was confirmed with a dual-well tracer test conducted under natural gradient. Nevertheless, the delayed tracer recovery from H3 was unique among results from Tuscaloosa-area test sites (e.g., compare Figure 4 to Figure 1).

A point-dilution test was conducted at H3 using 100 g of lithium bromide to prepare the tracer solution (it had not been convenient to do the point-dilution test coincident with the start of the drift-and-pumpback test). The tracer concentration was monitored over time at 16 depths from 42.7 to 64.6 m. Figure 5 includes plots of the electrode response *versus* time for four representative depths.

In Figure 5, the plots for the 45.7- and 63.4-m depths appear consistent with the requirement that suchs plots for a valid point-dilution experiment be a straight line. The former displays a steep positive slope, which is consistent with rapid flushing of the tracer from the well bore. The latter plot shows fluctuations of approximately 10 mV about the mean, but has a net slope of nearly zero, which indicates a depth interval with little or no ground-water flow.

The plot for the 45.7-m depth shows the greatest slope, +4.3 mV/min, for any of the depths monitored in well H3. However, the greatest slope shown in Figure 2 for the UASRC site is +0.12 mV/min. These slopes differ by a factor of 36, which from equation (3) would indicate that at the 45.7-m depth, well H3 had either a flow rate or a flow distortion factor that was unrealistically high.

Further, plots in Figure 5 from the 53.3- and 57.9-m depths show negative slopes for the first 23 and 42 min of the test, respectively, followed by positive slopes nearly as great as that for the

45.7-m depth. In other words, the negative slopes from the first part of the test at these depths show that tracer concentration at these depths clearly increased.

The cause of these perplexing test results was eventually deduced by plotting depth profiles of actual tracer concentration in the well bore at various times during the test. Representative plots are shown in Figure 6. Inspection of the plots shows that initial tracer distribution was uneven, and at 5 min into the test there was a concentration peak clearly visible at the 45.7-m depth. The peak descends through the well bore over time, with virtually no dilution, at the rate of 0.24 m/min, and disappears near the bottom of the well. That is, the tracer moved vertically through the well bore in virtual plug flow, and the rate of flow was sufficient to overwhelm any significant tracer dilution by horizontal advective ground-water flow.

Vertical movement in the well bores at the VA site was confirmed by placing a tracer slug at the top of the water column and monitoring its position over time. Well drillers have noted that a highly transmissive stratum, in which drilling circulation is easily lost, lies just above the Pottsville Formation. It is reasonable to conclude that virtually all of the tracer injected into these wells was lost to that stratum, and that the results of the earlier drift-and-pumpback test reflected recovery of tracer from that stratum and do not reflect properties of the whole saturated interval.

The hydraulic head-driving downward flow in the wells is best explained by postulating existence of a perched aquifer immediately overlying the regional sedimentary aquifer. A perched aquifer, and probable poor development of the 5-cm ID monitoring wells at this site, is consistent with observed water-level anomalies.

A clay layer approximately 18 m above the Pottsville Formation was subsequently identified as the likely boundary between the regional aquifer and the perched aquifer.

Hanford Bioremediation Site

Tracer test data from the Hanford bioremediation test site [5] are included here for two reasons. First, the results from the drift-and-pumpback test at this site illustrate the effect of insufficient tracer drift on the tracer recovery curve. Second, the results of point-dilution tests conducted at two wells 7.9 m apart highlight the power of this test to elucidate local aquifer inhomogeneity.

The program of tracer testing began at the bioremediation test site in June 1992. At that time, a drift-and-pumpback test using a lithium bromide tracer was attempted at well 299-W11-30. For the tracer solution, a volume of 4.6 L of water was used to dissolve 80 g of lithium bromide. The tracer was allowed to drift 10 d, because the best estimate of ground-water flow velocity available before tracer testing was 0.3 m/d or slightly less; in the author's experience, a net drift of 2 to 3 m seems to be generally optimal.

The well was pumped at a discharge rate of approximately 40 L/min to recover the tracer, and the bromide concentration of the discharge stream was monitored in the field using ISEs. The tracer recovery curve for this test is shown in Figure 7. Inspection of the figure shows that peak tracer concentration during recovery occurred as soon as pumping began, that is, the tracer had virtually not moved away from the well bore.

Insufficient drift can bias the results of a drift-and-pumpback test [1, 2]. Analysis of test data using equation (1) yielded an apparent flow rate of 0.03 m/d, but because of the experimental uncertainty, this value was taken as an upper bounding limit only. Confirmation of the actual flow velocity was done using a two-well tracer test conducted under natural gradient, where a second tracer slug was introduced into 299-W11-29, and well 299-W11-30, which is 7.9 m downgradient from 299-W11-29, was monitored for bromide. This test required several months of sampling and analysis of ground water from the downgradient well, and yielded a net flow velocity of approximately 0.027 m/d.

In January 1994, point-dilution tests were conducted at wells 299-W11-29 and 299-W11-30. Before the test, the standing water column of each well was found to be approximately 9.4 m. Both wells had a 10-cm ID, and the volume of water in each borehole was 74 L. A tracer charge of 25 g of potassium bromide per well was used, yielding an initial mean concentration of 230 mg/L as bromide in each well.

The results of the point-dilution tests are shown in Figure 8. Neither of the test wells has been extensively pumped; it is assumed that development has been adequate. Thus, the wells are considered to meet the criteria for a "minimally developed well." Data from both wells were normalized to the highest V* obtained during the tests, which was at the 76.5-m depth in well 299-W11-29. Therefore, assuming identical well construction specifications, the figure may be presented as a representation of the relative hydraulic conductivities of the two wells.

These tests show that the mean hydraulic conductivity of well 299-W11-30 is approximately 37% that of well 299-W11-29. Test data [5] obtained by monitoring well 299-W11-30 while conducting development pumping and a step-drawdown test at well 299-W11-29 yielded a transmissivity, and therefore mean conductivity, nearly equal to the transmissivity obtained from a constant-discharge pumping test at well 299-W11-30 where well 299-W11-29 was used for observation. This result occurred because the use of an observation well yields a transmissivity that reflects the mean characteristics of the sediments lying between the two wells. In both instances, the same body of sediments was interrogated. The point-dilution test, on the other hand, reflects characteristics of the sediments only in the immediate vicinity of a well.

DISCUSSION

Widespread acceptance of single-well tracer methods will depend in the future on the availability of practical equipment for performing the tests, and on the distribution of information suitable to aid in designing tests and interpreting results. The intent of this paper has been to move toward fulfillment of these two requirements by describing and comparing test experiences at three distinctly different sites, and by briefly presenting advances in the design of test apparatus. Equipment is not yet generally available, although servicable apparatus can be constructed from information in this and other published work [3, 4].

To date, the single-well methods discussed here have proved useful for estimating flow velocity, effective porosity, and the vertical distribution of flow at several field sites. Other single-well tracer methods have been conceived, but not yet practiced at field sites. Two such methods, briefly presented below, will serve to illustrate the direction of future research for field tracer methods.

Well transmissivity is best determined using conventional hydrologic field methods such as a constant-discharge pumping test. However, such testing is equipment- and time-intensive, so the number of tests performed at a field site–regardless of the number of available wells–is often limited. A simple and less expensive method for estimating well transmissivity is needed to determine which wells should be chosen for more extensive testing (e.g., in an ATES well field or in a network of monitoring wells at a waste disposal site). Under certain circumstances, point-dilution testing as described here can serve this purpose. First, all wells must be constructed to the same specifications (e.g., size of open hole, packing material, screen type, and slot size) so the flow distortion factor from equation (3) is the same for each well. Second, the transmissivity of at least one of the wells must be known (e.g., from a constant-discharge test). Third, each well must meet the criteria for a minimally developed well. Fourth, the local hydraulic gradient in the vicinity of each well must be known.

Equations (2) and (3) can be combined and rearranged as

$$V^* = KI/a \qquad (4)$$

Equation (4) relates the V* and K for each tested depth. However, the equation also relates the mean V* for the well to the mean hydraulic conductivity. Further, the factor V* is proportional to the slope of a plot of millivolts *versus* time, derived from point-dilution tests. Like the flow distortion factor, a, the proportionality constant, p, that relates V* to plot slope, m, depends on well construction specifications and will be the same for all wells. Substituting the product of plot slope and the proportionality constant for V* yields

$$pm = KI/a \qquad (5)$$

Two wells may then be compared using equation (5) as

$$(pm_1/pm_2) = (K_1I_1/a)/(K_2I_2/a)$$

and

$$K_2 = K_1I_1m_2/m_1I_2 \qquad (6)$$

Thus, if the hydraulic conductivity of one well is known, the hydraulic conductivity of another may be calculated from the respective local hydraulic gradients and point-dilution data, where the factors m are the mean slopes of the plots of electrode response in millivolts *versus* time for the two wells. The transmissivity of the second well is then calculated by multiplying K_2 by the aquifer thickness.

Figure 8 suggests a second application of this type of analysis. Transmissivities (and mean hydraulic conductivities) assigned to the wells represented in Figure 8, based on pumping tests at each well, were nearly the same because each pumping test interrogated nearly the same body of sediments. That is, the design of the pumping tests masked small-scale, local inhomogeneities that may be important in the analysis of bioremediation experiments planned for this test site. Point-dilution data showed that the mean hydraulic conductivity of well 299-W11-30 is 37% of that of well 299-W11-29. Pumping tests yielded a mean conductivity of approximately 7.6 m/d for the body of sediments between the wells. For modeling and analyzing the results of the bioremediation demonstration, a somewhat better representation of local hydraulic conditions might be obtained by assigning hydraulic conductivities of 4.3 and 11.0 m/d to these wells, respectively (yielding a mean of 7.6 m/d).

Finally, a body of evidence shows that single-well, multiple tracer, injection-withdrawal experiments may be useful for estimating flow parameters such as dispersivity and solute retardation. Consider, for example, a series of tracer experiments conducted in 1986 in deep basalt aquifers at the Hanford Site [8]. (Many of the confined aquifers within the regional basalt flows are highly transmissive and are sources of irrigation water, although natural ground-water flow velocities are negligibly low.) In one of the experiments, 38,000 L of water labeled with a lithium bromide tracer was injected into the Sentinel Gap flow top aquifer at borehole DC-18. The injection was followed immediately by a chaser of 30,000 L of fresh water. One week later, pumping was begun to recover the tracer and continued for 6 d. Figure 9 shows the results of tracer recovery, where bromide and lithium concentrations are plotted against pumped volume normalized to the total injected volume (68,000 L).

Three facets of this figure are striking. First, the recovery plots of both lithium and bromide tend toward a straight line. Second, the slopes of the individual plots are distinctly different. Third, the recovery plots are unaffected by the rate of pumping. Analysis of these results suggests that the slope of the plot of the conservative bromide ion depends on aquifer dispersivity. The difference in slope between the lithium and bromide plots must reflect the tendency for lithium to reversibly adsorb to mineral phases in the aquifer. Such adsorption is responsible for retardation of certain solutes (e.g., lithium) in ground-water plumes, and is in many ways analogous to the heat exchange between ground water and mineral phases that causes retardation of thermal plumes.

Tracer behavior similar to that shown in Figure 9 has been observed in other aquifers (e.g., sedimentary aquifers), and for other tracers [8]. However, there is currently no method for quantitatively translating the shape of recovery plots from single-well, injection-withdrawal tests into numerical factors for modeling dispersion or retardation. Development of such a method is a main goal of continuing research.

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FIGURE CAPTIONS

Figure 1. Bromide concentration *versus* time during the tracer recovery stage of the drift-and-pumpback test at the University of Alabama Student Recreation Center.

Figure 2. Bromide ion-selective electrode response *versus* time at 12 depth intervals during the point-dilution test at the University of Alabama Student Recreation Center.

Figure 3. Relative horizontal ground-water flow *versus* depth within the bore of the test well at the University of Alabama Student Recreation Center. The flow profile is compared to stratigraphy based on the examination of drill cuttings.

Figure 4. Bromide concentration *versus* time during the tracer recovery stage of the drift-and-pumpback test at the VA Medical Center.

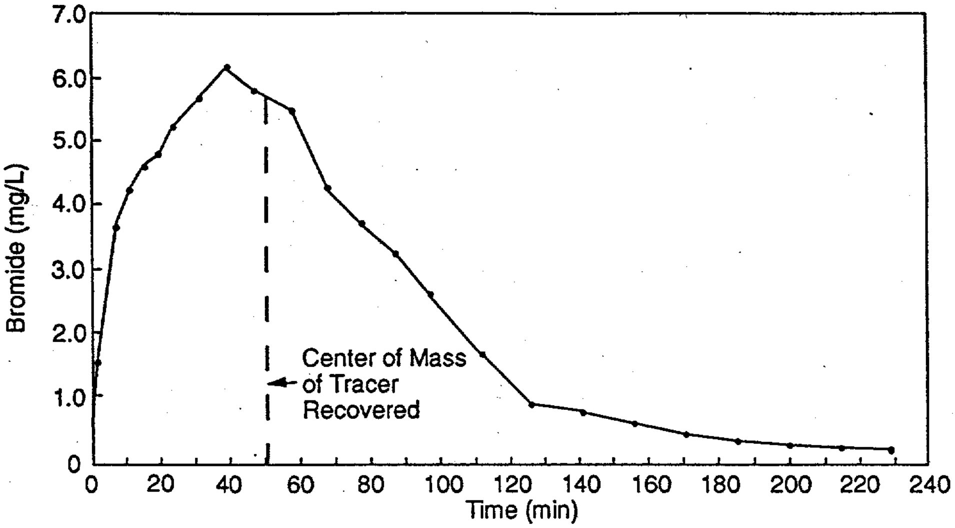
Figure 5. Bromide ion-selective electrode response *versus* time at four selected depth intervals during the point-dilution test at the VA Medical Center.

Figure 6. Bromide concentration *versus* depth at various times during the point-dilution test at the VA Medical Center.

Figure 7. Bromide concentration *versus* time during the tracer recovery stage of the drift-and-pumpback test in well 299-W11-30 at the Hanford bioremediation test site.

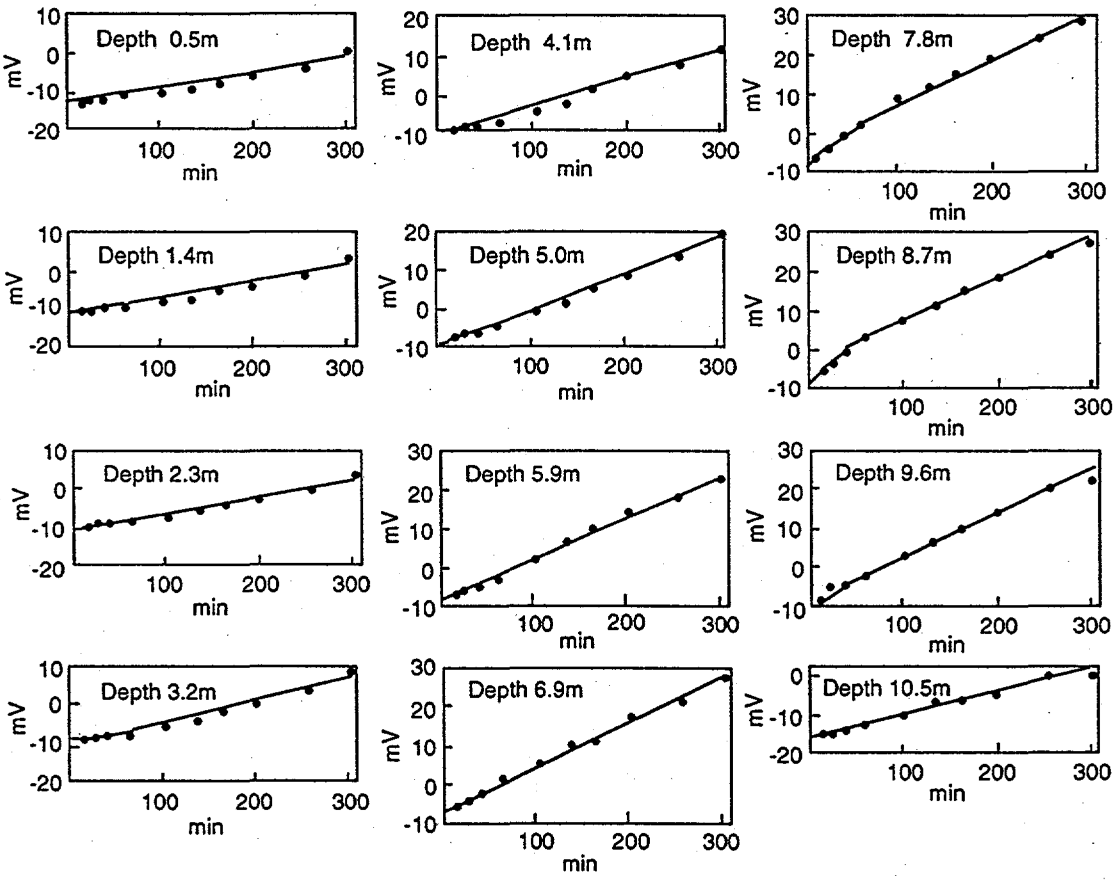
Figure 8. Relative horizontal ground-water flow *versus* depth within the bores of two test wells at the Hanford bioremediation test site.

Figure 9. Concentrations of lithium and bromide tracers *versus* pumped volume during the tracer recovery stage of an injection-withdrawal test in a confined basalt flow top aquifer at the Hanford Site. Pumped volume is normalized to the 68,000 L tracer volume injected for the test.



7.0
6.0
5.0
4.0
3.0
2.0
1.0
0
Bromide (mg/L)
0
20
40
60
80
100
120
140
160
180
200
220
240
Time (min)
Center of Mass
of Tracer
Recovered

FIG 2



Depth 0.5m
Depth 4.1m
Depth 7.8m
Depth 1.4m
Depth 5.0m
Depth 8.7m
Depth 2.3m
Depth 5.9m
Depth 9.6m
Depth 3.2m
Depth 6.9m
Depth 10.5m
mV
min
10
0
-10
-20
20
30
100
200
300

FIG 3

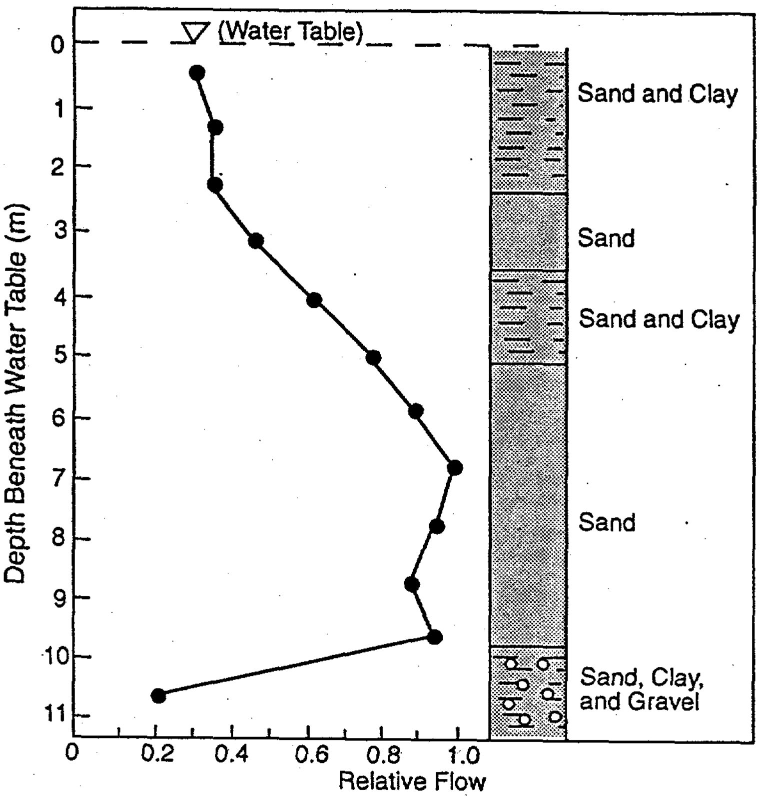

(Water Table)
0
1
2
3
4
5
6
7
8
9
10
11
Depth Beneath Water Table (m)
0
0.2
0.4
0.6
0.8
1.0
Relative Flow
Sand and Clay
Sand
Sand and Clay
Sand
Sand, Clay, and Gravel

FIG 4

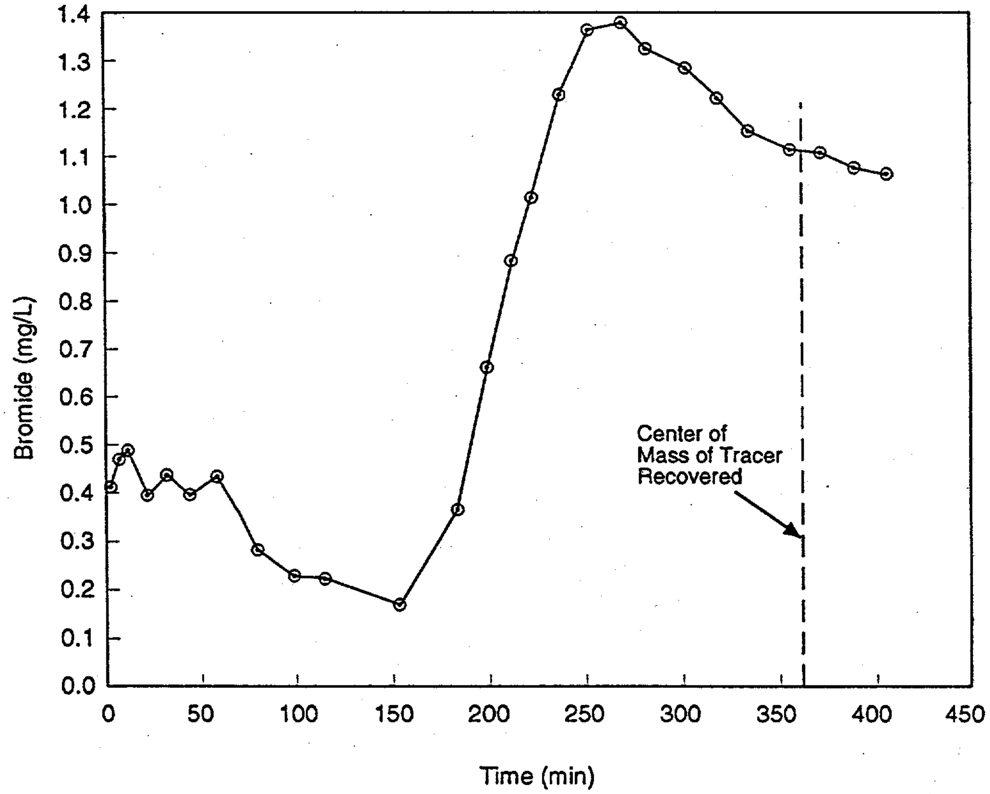

1.4
1.3
1.2
1.1
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Bromide (mg/L)
0
50
100
150
200
250
300
350
400
450
Time (min)
Center of
Mass of Tracer
Recovered

FIG 5

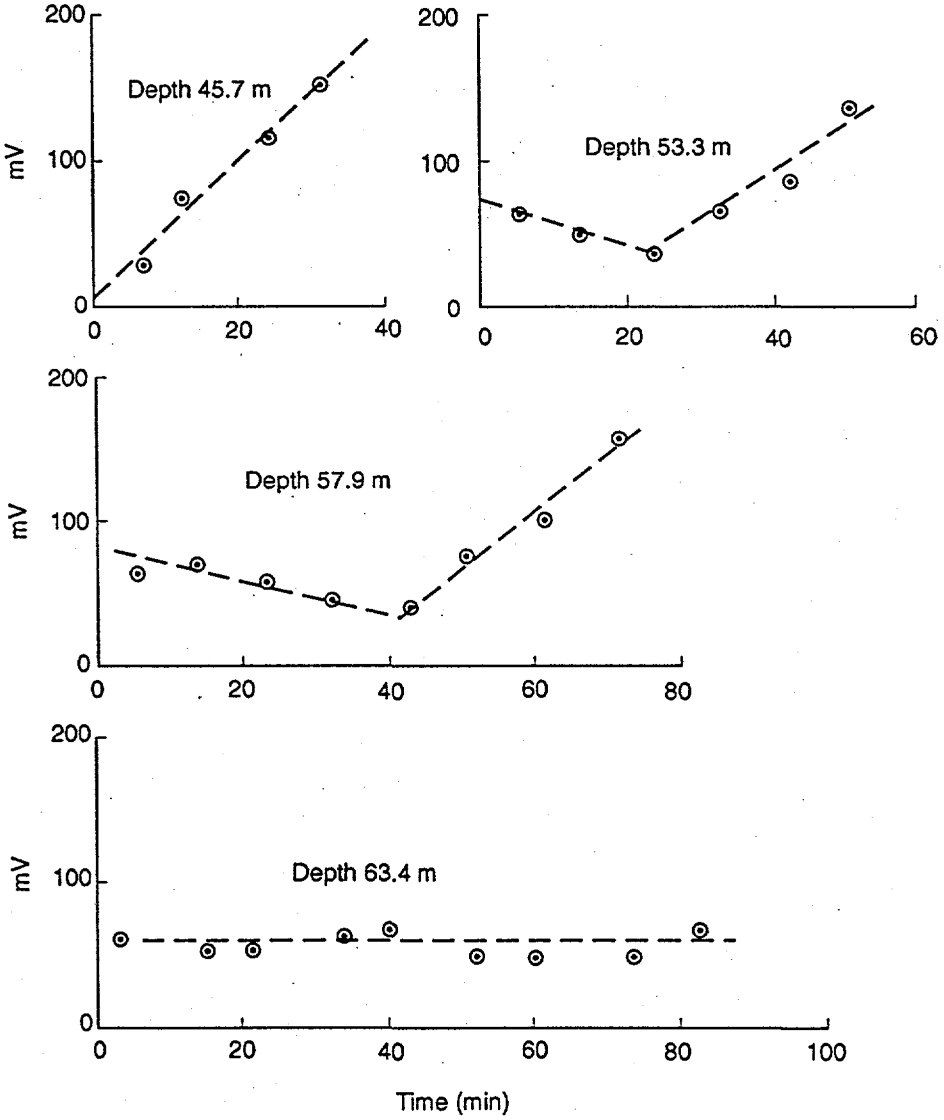
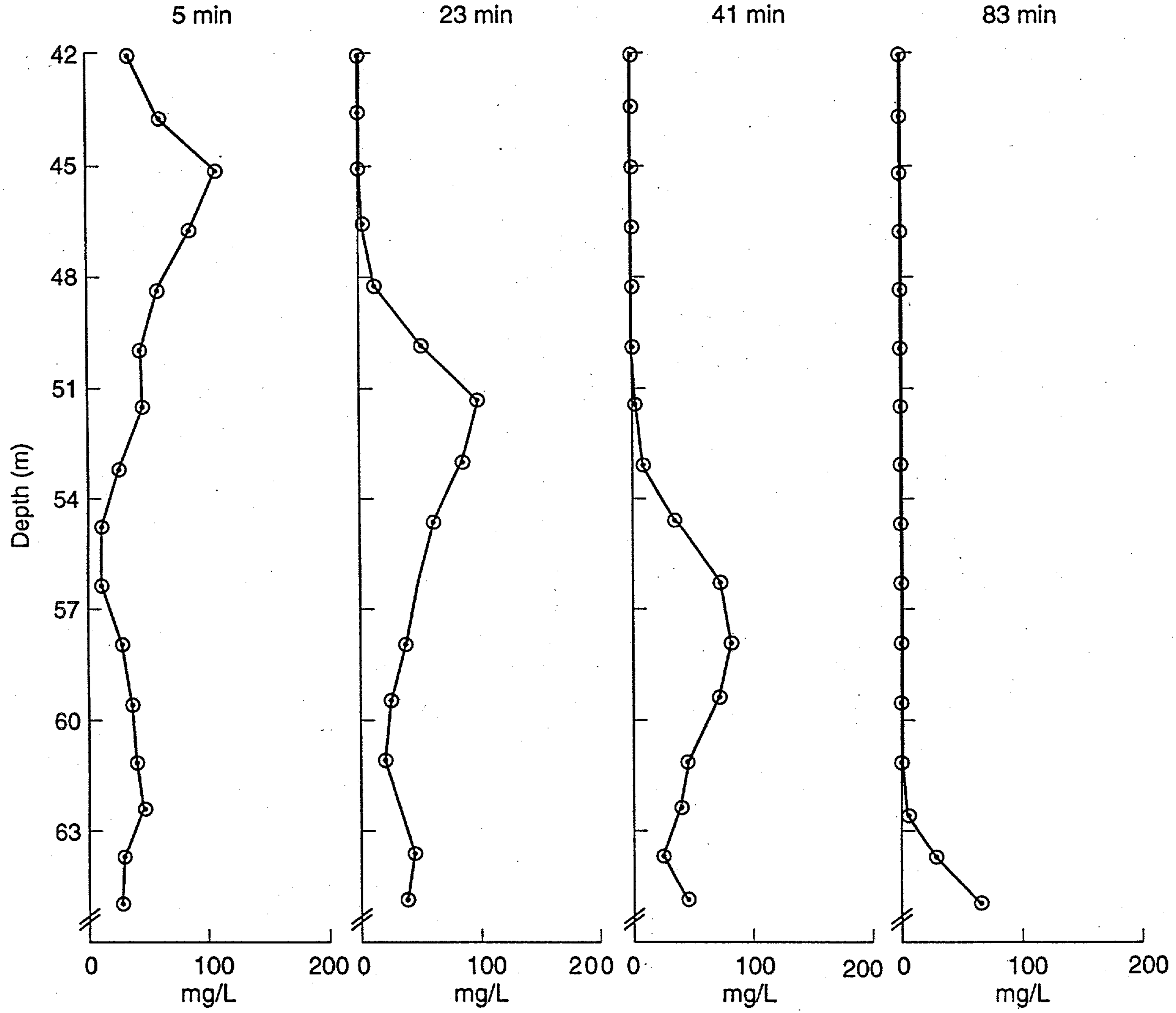

200
100
0
mV
Depth 45.7 m
0
20
40
200
100
0
Depth 53.3 m
0
20
40
60
200
100
0
mV
Depth 57.9 m
0
20
40
60
80
200
100
0
mV
Depth 63.4 m
0
20
40
60
80
100
Time (min)

FIG 6



5 min
23 min
41 min
83 min
Depth (m)
42
45
48
51
54
57
60
63
0
100
200
mg/L

FIG 7

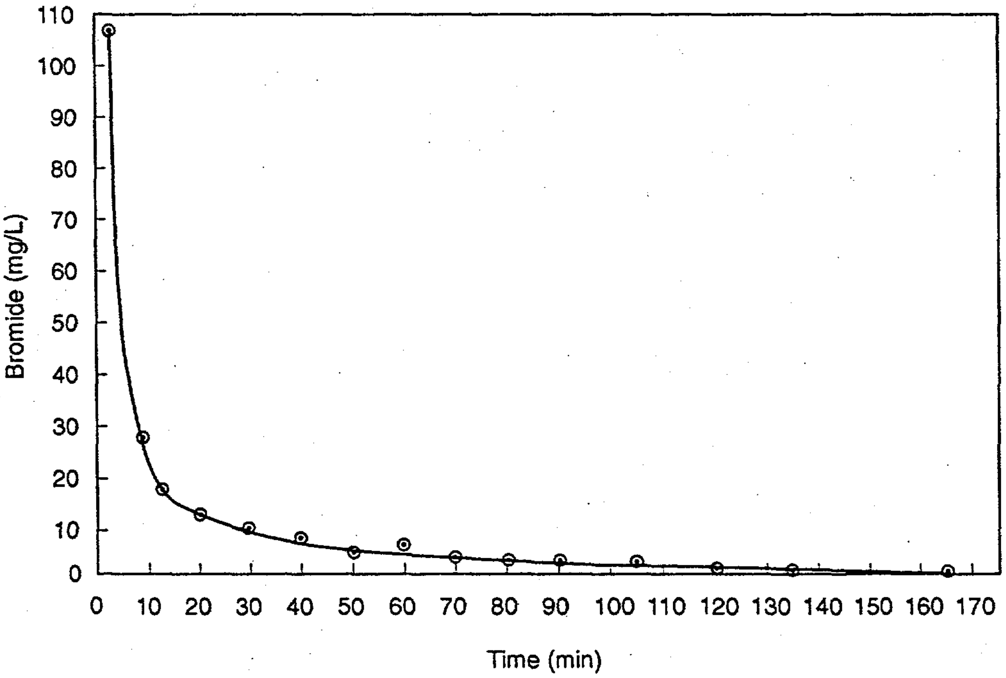
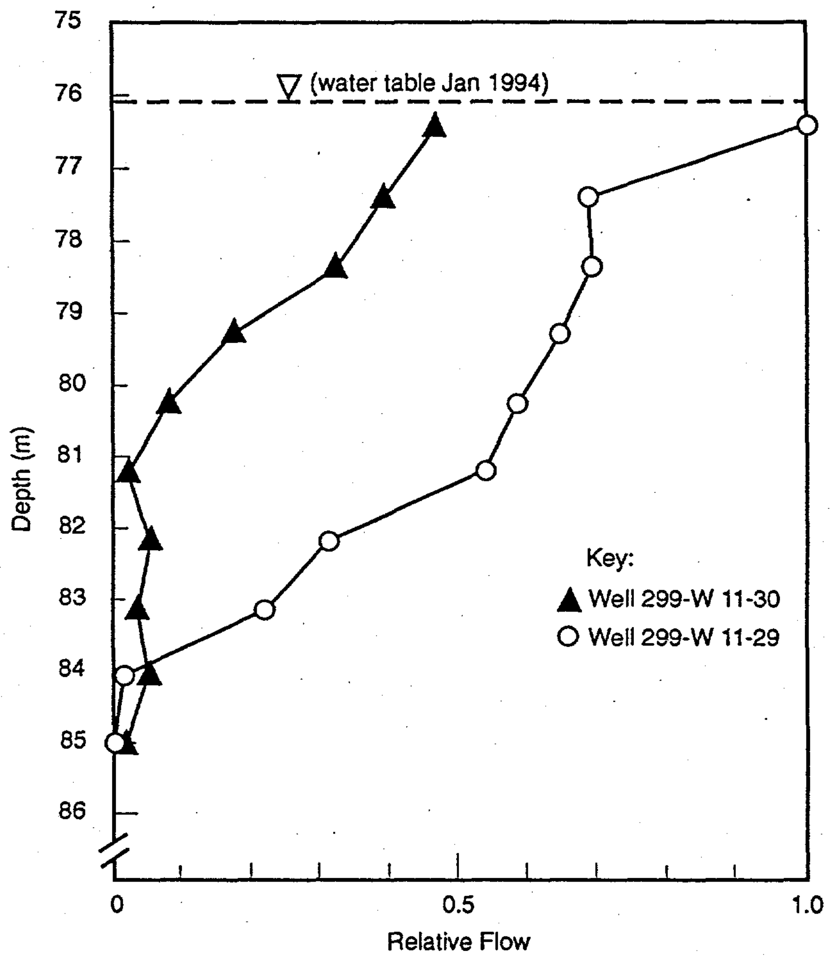
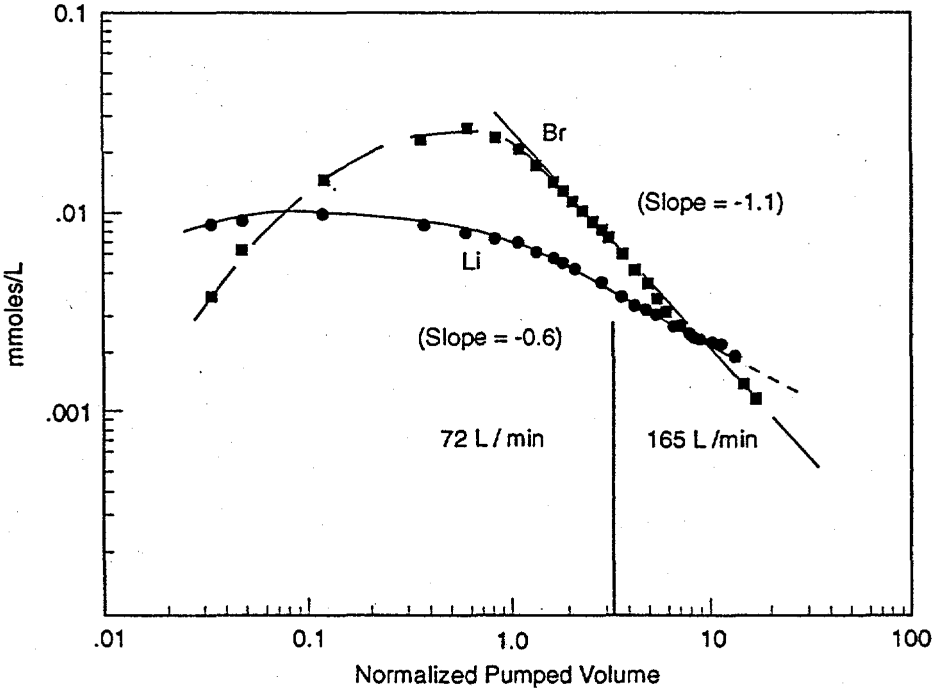

Bromide (mg/L)
110
100
90
80
70
60
50
40
30
20
10
0
0
10
20
30
40
50
60
70
80
90
100
110
120
130
140
150
160
170
Time (min)

FIG 8



75
76
77
78
79
80
81
82
83
84
85
86
Depth (m)
▽ (water table Jan 1994)
Key:
▲ Well 299-W 11-30
○ Well 299-W 11-29
0
0.5
1.0
Relative Flow

FIG 9


0.1
.01
.001
mmoles/L
Br
(Slope = -1.1)
Li
(Slope = -0.6)
72 L / min
165 L /min
.01
0.1
1.0
10
100
Normalized Pumped Volume