Field comparison of the point velocity probe with other groundwater velocity measurement methods

W. Labaky,[1] J. F. Devlin,[2] and R. W. Gillham[3]

Received 3 April 2008; revised 11 December 2008; accepted 15 January 2009; published 14 March 2009.

[1] Field testing of a new tool for measuring groundwater velocities at the centimeter scale, the point velocity probe (PVP), was undertaken at Canadian Forces Base, Borden, Ontario, Canada. The measurements were performed in a sheet pile-bounded alleyway in which bulk flow rate and direction could be controlled. PVP velocities were compared with those estimated from bulk flow, a Geoflo® instrument, borehole dilution, colloidal borescope measurements, and a forced gradient tracer test. In addition, the velocity profiles were compared with vertical variations in hydraulic conductivity (K) measured by permeameter testing of core samples and in situ high-resolution slug tests. There was qualitative agreement between the trends in velocity and K among all the various methods. The PVP and Geoflo® meter tests returned average velocity magnitudes of 30.2 ± 7.7 to 34.7 ± 13.1 cm/d (depending on prior knowledge of flow direction in PVP tests) and 36.5 ± 10.6, respectively, which were near the estimated bulk velocity (20 cm/d). The other direct velocity measurement techniques yielded velocity estimates 5 to 12 times the bulk velocity. Best results with the PVP instrument were obtained by jetting the instrument into place, though this method may have introduced a slight positive bias to the measured velocities. The individual estimates of point velocity direction varied, but the average of the point velocity directions agreed quite well with the expected bulk flow direction. It was concluded that the PVP method is a viable technique for use in the field, where high-resolution velocity data are required.

Citation: Labaky, W., J. F. Devlin, and R. W. Gillham (2009), Field comparison of the point velocity probe with other groundwater velocity measurement methods, *Water Resour. Res.*, *45*, W00D30, doi:10.1029/2008WR007066.

1. Introduction

[2] The estimation of groundwater velocity is a fundamental requirement in contaminant hydrogeology. Typically, this estimation is achieved through Darcy's law, corrected for porosity,

$$v = \frac{K}{n}\frac{\Delta h}{\Delta l} \tag{1}$$

where K is hydraulic conductivity (L/T), v is average linear groundwater velocity (L/T), n is porosity, $h = \psi + z$ (L), z is elevation head (L), ψ is pressure head (L), and l is the distance over which the hydraulic head is observed to change (L). Units given are generalized units with L = length, T = time. This calculation is sometimes applied to obtain a site-wide velocity estimate, using simple hand calculations based on hydraulic head and an estimate of bulk hydraulic conductivity. Often, more sophisticated calculations are performed by software that solves the equations for three-dimensional flow. Numerical models of this kind can account for hydraulic conductivity variations in space, as well as anisotropy, making it possible to compute groundwater velocity fields that are of great value in predicting contaminant movement.

[3] Despite the strengths of the Darcy's law approach, situations occur in which alternative methods of velocity estimation offer advantages. For example, it sometimes arises that the gradient is inherently low and difficult to measure accurately, even over distances of tens of meters [*Devlin and McElwee*, 2007]. In other cases, the area of concern is too small to permit accurate hydraulic gradients to be determined. With the advent of permeable reactive barriers, the need for velocity estimations over short distances is acute. Groundwater velocities determine residence times, and these are intimately connected to barrier performance. In these cases, and in general, uncertainty in the value(s) of K used in the calculation is quite large [*Sudicky*, 1986; *Mas-Pla et al.*, 1997; *Gierczak et al.*, 2006], and this uncertainty transfers to the velocity estimate.

[4] Several techniques have been developed to measure groundwater velocities directly. An obvious basis for these kinds of measurements involves the use of tracers. The environmental tracers, including ^{2}H, ^{3}H, ^{18}O, ^{14}C and ^{36}Cl, Freon compounds, and heat, have been used to study flow and transport at a variety of scales from meters to hundreds of kilometers [*Robertson and Cherry*, 1989; *Clark and Fritz*, 1997; *Bartolino and Cole*, 2002; *Kalbus et al.*, 2006].

[1]Schlumberger Middle East, Abu Dhabi, United Arab Emirates.
[2]Department of Geology, Lindley Hall, University of Kansas, Lawrence, Kansas, USA.
[3]Department of Earth Sciences, University of Waterloo, Waterloo, Ontario, Canada.

0043-1397/09/2008WR007066

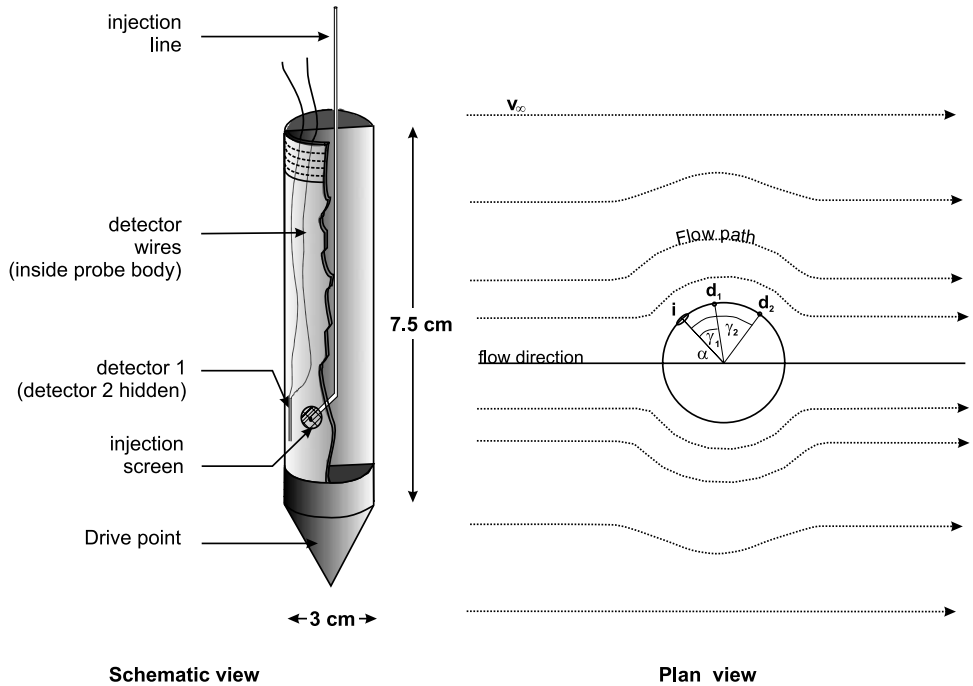

Figure 1. Schematic and plan views of the PVP. Flow travels around the probe body, picking up tracer at the injection port, i, and delivering it to two detectors, d_1 and d_2. Knowing the angles γ_1 and γ_2, apparent velocities averaged up to the detectors can be converted to the average linear velocity in the aquifer away from the probe, v_∞. In addition, the angle from the injection port to the average flow direction, α, can be calculated.

[5] The use of injected tracers to investigate natural gradient groundwater flow goes back to the turn of the twentieth century [*Schlichter*, 1905]. Controlled tests of these kinds, called natural gradient tracer tests, have been used to investigate groundwater movement and transport processes on relatively small scales, up to a few hundred meters [*Mackay et al.*, 1986; *LeBlanc et al.*, 1991]. Unintentional spills have sometimes led to longer plumes [e.g., *Perlmutter and Lieber*, 1970; *van der Kamp et al.*, 1994], but lack of control of the history of these plumes often limits what can be learned from them. The injected tracers used for groundwater velocity estimation have included such substances as chloride and bromide [*Mackay et al.*, 1986], and fluorescent dyes [*Kasnavia et al.*, 1999].

[6] Because of the time and cost of performing natural gradient tracer tests a variety of instruments have been developed to measure groundwater velocity at the scale of a single well. These include point (borehole) dilution devices [*Pitrak et al.*, 2007], the Geoflo meter® [*Kerfoot and Massard*, 1985], the In Situ Permeable Flow Sensor [*Ballard*, 1996; *Alden and Munster*, 1997], the Colloidal Borescope [*Kearl*, 1997] and the Laser Doppler Velocimeter [*Momii et al.*, 1993]. Most of these techniques were included in the work described here, and so are discussed in more detail below. Some of the methods require a well, while others depend on the instruments coming into direct contact with the aquifer material. Among the instruments in the latter group is the recently introduced point velocity probe (PVP) [*Labaky et al.*, 2007]. A PVP consists of a cylinder outfitted with a tracer release and detection system on its surface. By timing the arrival of the tracer at 2 or more detectors on the cylinder surface, at different distances from the injection port, both magnitude and direction of the average linear velocity vector can be determined (Figure 1). In addition, unlike other methods, PVPs provide velocity estimates relevant to the centimeter scale, a scale comparable to that of multilevel sampling, and a scale at which geochemical and hydrogeological effects of microbial activity can be observed [*Devlin and Barker*, 1996; *Schillig*, 2008]. The viability of the PVP method was demonstrated in laboratory tanks and with numerical modeling [*Labaky et al.*, 2007]. It remains to be demonstrated that the probes can be installed and used to advantage in a field setting.

[7] The purpose of this work was to test PVP performance in the field against several other established techniques for groundwater velocity measurement. The methods chosen for comparison included bulk estimates of velocity from known discharge rates, the Geoflo® meter [*Kerfoot and Massard*, 1985; *Guthrie*, 1986], borehole dilution from drive point and standard wells [*Drost et al.*, 1968], and the colloidal borescope [*Kearl*, 1997]. Details of the materials and procedures for all methods are given below. *Labaky et al.* [2007] noted that a challenge for the PVP method, in field applications, is for the instrument to be installed in a fashion that promotes good contact between the aquifer and the probe surface, with a minimal disturbed zone, i.e., minimization of skin effects. In this work, we begin to address the challenge by performing the comparisons presented here in a non-cohesive sand aquifer with a further goal of comparing methods of installing PVPs, including driving by vibrating

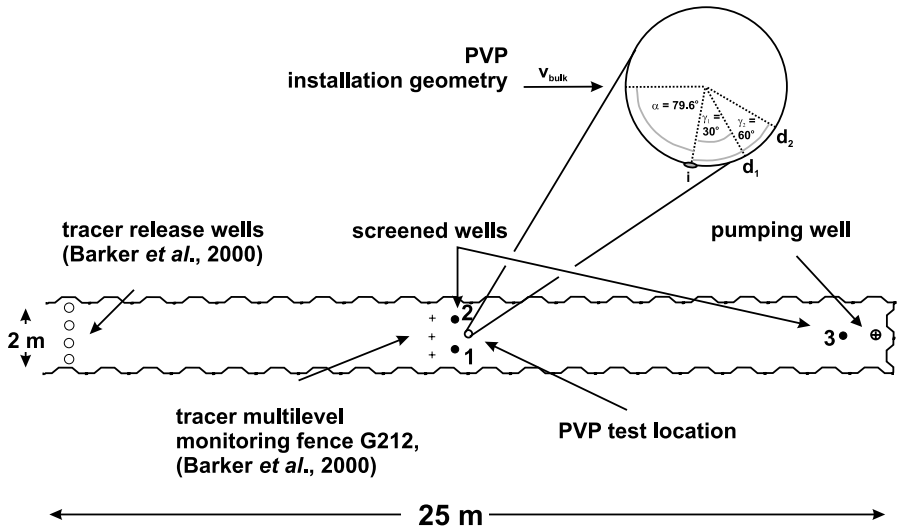

Figure 2. Experimental layout at the field site. Also shown, in plan view, is the orientation of the PVP during testing.

hammer and jetting. Additional work and evaluations will be needed in other aquifer settings.

2. Methods

[8] Field experiments were carried out in a sand aquifer at Canadian Forces Base, Borden, that has been described in detail elsewhere [*Mackay et al.*, 1986; *Ball et al.*, 1990]. The aquifer is a slightly heterogeneous deltaic beach sand of fine to medium grain size. The sand consists primarily of quartz and feldspar grains, with substantial amounts of carbonates and amphiboles also present. Hydraulic conductivity of the aquifer has been found to range between 6×10^{-4} cm/s and 2×10^{-2} cm/s on the basis of permeameter tests [*Sudicky*, 1986].

[9] All measurements were made in a section of the aquifer surrounded by sheet piling on three sides (Figure 2). The enclosed section was about 2 m wide, 3 m deep, with about 2.3 m of saturated aquifer underlain by a clay aquitard. The up-gradient side of the enclosure was open to admit groundwater, while the down-gradient side was sealed and pumped at about 216 mL/min to draw water through the alleyway.

[10] Two 7.6 cm diameter wells (boreholes 1 and 2, Figure 2) with 1.52 m long screens and 1.52 m riser pipes were used for the comparative tests described below. Cores collected during the installation of these wells were used to obtain depth-specific K by permeameter testing.

2.1. PVP Tests

[11] The PVP instrument was constructed of two half cylinders of stainless steel, 7.5 cm in length with a 3 cm outside diameter (OD), that were held together with machine screws (Figure 1). One half cylinder was hollow and contained the injection and detection ports. The other half cylinder was solid and contained a groove to stabilize the injection line. The two parts were threaded on both ends to connect the instrument with the drive tip at the bottom, and the extension rods at the upper end. The probe itself consisted of the cylinder body, an injection port and one or more tracer detectors. If the probe is constructed with two or more detectors, equations (2) and (3) can be used to estimate the ambient groundwater velocity (magnitude v_∞, and direction α, see Figure 1) [*Labaky et al.*, 2007]:

$$v_\infty = \frac{\nu_{app} \times \gamma}{2(\cos\alpha - \cos(\alpha + \gamma))} \tag{2}$$

$$\alpha = \tan^{-1}\left[\frac{\nu_{app1}\gamma(\cos\gamma_2 - 1) + \nu_{app2}\gamma_2(1 - \cos\gamma)}{\nu_{app1}\gamma\sin\gamma_2 - \nu_{app2}\gamma_2\sin\gamma}\right] \tag{3}$$

where v_{app} is the apparent velocity of the tracer measured at a detector on the probe surface (apparent velocity, L T^{-1}), and γ is the angle between a detector and the injection port (Figure 1). The subscripts 1 and 2 refer to the detectors.

[12] The tracer release system consisted of a stainless steel screen (0.055 cm mesh) welded onto a 0.6 cm outer diameter stainless steel nut that tightened against the injection line from the outside surface of the cylinder. The effective diameter of the injection screen was 0.3 cm. The injection line was an L-shaped stainless steel tube, 0.3 cm OD connected to a section of polyethylene tubing of the same diameter with a Swagelok® connector. The distal end of the polyethylene injection line was connected to a 60 mL plastic syringe filled with the tracer solution. A graduated roller clamp on the injection line was used to inject about 0.01 mL of the tracer solution at the onset of each measurement.

[13] Two pairs of detectors were installed on each side of the injector; detectors on only one side were needed for any particular measurement. The second pair was added to accommodate cases in which large changes in flow direction might occur, or in which flow direction was poorly known at the time of installation. The detectors were made from 0.075 cm gauge insulated copper wires positioned in individual grooves that were 0.3 cm apart on the surface of the probe with the insulation removed on the outward facing sides only, to prevent short circuiting through the body of the probe. These wires were connected at surface to a conductivity meter and a data logger. The angle (γ) between the injection port and each of the left side detectors was fixed at 30° and 60°, for detectors 1 and 2, respectively.

These detectors were used in all tests. The right side detectors were not used in this work.

[14] Preliminary laboratory testing and numerical modeling suggested that PVP measurements could be made with errors in the magnitude of the estimates of about ±9% and direction errors of about ±8°. However, the measurements were sensitive to skin effects in the porous medium next to the probe [*Labaky et al.*, 2007]. To address this issue, two methods of installing the PVP were compared: jetting and driving. In the jetting method, a 6 cm OD casing was driven to the bottom of the aquifer and the sand was then flushed out of the casing with a high-pressure stream of water. The PVP was lowered through the water column to the bottom of the casing, which was subsequently withdrawn, leaving aquifer material free to collapse around the body of the probe. Since a disturbance of this kind would be expected to increase the hydraulic conductivity of the aquifer near the probe [*Labaky et al.*, 2007], the inside diameter of the casing and the outside diameter of the probe were selected to be as close as possible. In this way the disturbance was minimized, as was any possible bias to the velocity measurements. The probe was pulled up incrementally between measurements, which were made at 10 cm intervals.

[15] The experimental procedure, involving tracer injection and detection, was similar to the laboratory method described by *Labaky et al.* [2007]. However, because the background conductance was significantly higher in the field test than in the laboratory, 0.01 mL of a 6000 mg/L NaCl solution (replacing the 600 mg/L solution in the laboratory tests) served as the tracer for each measurement. Density driven tracer movement was not a concern because the position of the side detectors was such that only the horizontal component of flow could be measured by them. As long as the tracer pulse did not sink so rapidly that it entirely missed the detector, the horizontal component of the velocity could be measured. This expectation was corroborated by modeling the migration of a 6000 mg/l salt tracer around the probe [*Labaky*, 2004] using the finite element model SALTFLOW [*Molson and Frind*, 1994]. The model showed that over short distances, such as that between the injection port and the detectors (0.9 to 1.2 cm), no density effects were discernable.

[16] Measurements were interpreted (1) assuming the flow direction was parallel to the bounding sheet piles and (2) assuming that the flow direction was not known. In the latter case, the flow direction was determined using data from the two left-side detectors and equations (2) and (3), as described by *Labaky et al.* [2007].

[17] Finally, a second set of experiments was performed in which the PVP was driven into the ground with a vibrating hammer. At selected depths on the way down, the descent was interrupted for measurements.

2.2. Permeameter Tests

[18] The depth-specific hydraulic conductivity of the formation was determined in the laboratory by falling-head permeameter tests on segments of core collected from the aquifer. The segments were 5 cm in length and the core was retrieved during the drilling of boreholes 1 and 2. The tests included 25 samples from depths between 122 and 263 cm below ground surface (bgs). These tests provided estimates of hydraulic conductivity, and comparisons were made between vertical trends in the magnitudes of K and v (where v was measured using the various methods of this study). The K values could not be converted accurately to highly localized velocities because applicable hydraulic gradients were not known.

2.3. High-Resolution Slug Tests

[19] Depth-specific high-resolution slug test measurements were carried out at boreholes 1 and 2. Slug tests were performed on a series of 7.6 cm sections of the well screens. Each test section was isolated from the rest of the well with inflatable packers mounted on a central 3.5 cm diameter PVC pipe [*Zemansky and McElwee*, 2005]. Measurements were made at 7.6 cm increments, which was also the length of the test interval. A slug consisted of a 30 cm to 120 cm long column of water (depending on the proximity to the static water level) held in the pipe under vacuum. Tests were initiated by opening a valve in the central pipe, which released the slug into the isolated section of the well screen. The change in pressure in the isolated section was monitored over time with a pressure transducer. The tests were conducted over the depth interval of 222 to 275.6 cm bgs in borehole 1 and between 209 and 270 cm bgs in borehole 2. Measurements were limited to these depths because at shallower depths, close to the water table, drawdown during the collection of slug water resulted in air entering the system. Results were interpreted according to the method given by *McElwee and Zenner* [1998] and *McElwee* [2001]. As with the permeameter tests, trends in K values were later compared with trends in v determined by other methods.

2.4. Geoflo® Meter

[20] The Geoflo® meter operates on the principle that a heat pulse tracer, released in the center of a ring of thermistors within a well, can be used to estimate magnitude and direction of groundwater flow on the basis of the arrival time of the heat pulse at the thermistor(s) and the location on the ring of the thermistors that receive the signal. A model 40L Geoflo® was calibrated in a slot 10 PVC tube, Monoflex® well screen with a 5.1 cm diameter and a 0.64 cm slot interval. The screen was placed at the center of a calibration tank provided by the manufacturer and the remaining volume was filled to within 7 cm of the top with Borden aquifer sand. The tank was filled with water and left to equilibrate overnight. A peristaltic pump was used to regulate flow across the tank.

[21] Calibration was carried out for velocities of 0, 15, 30 and 60 cm/d. Each calibration experiment consisted of two parts. In the first part, thermistor 1 was down gradient from the heat source and aligned with the direction of flow. The heat pulse was generated and the readings were noted. The second part saw the repetition of the same procedure with the probe rotated by 180°. This method was used to compensate for potential deviations of thermistors from the central heat unit. A period of 1 h was allowed to elapse between each half test to allow the thermistors to cool and reequilibrate. The calibrations were repeated between 2 and 6 times, with the greater number of repetitions applied to the lower velocities.

[22] Field experiments were carried out in boreholes 1 and 2 (Figure 2), which were constructed with PVC screens identical to those used in the calibration tests. Measurements were made beginning with the deeper locations and

moving upward. Each measurement was repeated between 2 and 4 times. A correction factor (0.85), to account for difference in temperature between the calibration experiments (25°C) and the field measurements (10°C), was recommended in the manufacturer's manual and was applied to all measured velocities [*K-V Associates*, Inc., 1983].

2.5. Borehole Dilution Tests

[23] The principles of borehole dilution testing were introduced as early as 1916 [*Halevy et al.*, 1967]. Details of the method are described elsewhere [*Drost et al.*, 1968; *Gaspar and Oncescu*, 1972]. The method works by relating the rate a tracer is flushed from a well (or packed-off section of a well) to groundwater velocity. Borehole dilution experiments were conducted at boreholes 1 and 2, and using drive points emplaced with a vibrating hammer. The downhole dilution instrument was 5 cm in diameter and 15 cm in length. An inflatable packer on each end of the mixing chamber was used to isolate the chamber from the rest of the borehole. The mixing chamber was connected to the inlet and outlet ends of a peristaltic pump at ground surface. A closed circulation system was maintained by pumping water out through one tube and reinjecting it through the other to ensure continuous mixing of the tracer solution. The electrical conductivity of the solution was monitored with an in-line electrical conductivity cell.

[24] Experiments were initiated by lowering the instrument down the borehole to the desired depth. Circulation of the water was then begun to eliminate entrapped air in the lines and in the mixing chamber. Next, the packers were inflated to a pressure of 207 kPa (2 atm) and background conductivity of the groundwater was monitored for a minimum of 20 min, or until the readings stabilized. Depending on the background conductivity, the injected volume of NaCl tracer solution was between 5 and 10 mL, yielding a maximum solution conductance of 3 to 5 times the background concentration. Each experiment was run until 33% dilution of the original tracer pulse was attained, or for a maximum of 180 min [*Le Sieur*, 1999]. As is common practice with this method, a correction factor (2.29 in this case) was applied to each velocity estimate to account for well construction effects on flow on the basis of the equation given by *Halevy et al.* [1967].

[25] Drive point borehole dilution tests were conducted in a fashion similar to the well-based tests [*Le Sieur*, 1999]. The tip of the drive point dilution probe was made from a Johnson Filtration System, Inc. well screen of 0.025 cm slot size. The screen was 18 cm long and 3.8 cm in diameter. The drive point tip was attached to drilling rods and driven to the desired depth with a vibrating hammer. During the tests the screen was isolated from the standpipe with an inflatable packer. The correction factor for well construction effects in these experiments was 2.27, using the equation from *Halevy et al.* [1967].

2.6. Colloidal Borescope Tests

[26] The colloidal borescope was operated by AquaVISION Environmental LLC. An attempt was made to collect data at 15 cm intervals in boreholes 1 and 2, starting from the bottom and moving upward. Additional data were gathered at 30 cm intervals in borehole 3 (Figure 1) (data not shown).

[27] To make a measurement, the instrument was positioned in the well and allowed to stand for 30 min, allowing turbulence to dissipate. If after 30 min monitored particles were still affected by turbulence (i.e., no clear direction of flow could be discerned), the borescope was raised to another level and a new measurement was attempted. Colloidal sized particles in the water that passed through the instrument were monitored at the surface on a television screen. Data were collected for at least an hour per depth of measurement.

3. Results

3.1. PVP Installation Experiments

[28] The velocity associated with bulk flow in the sheet pile alleyway was calculated to be 20 ± 1.7 cm/d on the basis of the following equation:

$$v = \frac{Q}{An} \tag{4}$$

where Q is discharge through the alleyway (216 mL/min), measured at the pumping well (Figure 1), A is the cross-sectional area of the saturated portion of the alleyway (4.6 m^2, including the capillary fringe estimated to be 35 cm high [*Xie*, 1994]) and n is the porosity (0.33 as determined by *Mackay et al.* [1986] and *Ball et al.* [1990]). Individual measurements of velocity ranged from 19.5 to 63.7 cm/d. It should be noted that individual PVP derived velocities are not expected to necessarily agree with the bulk velocity, since the former reflect conditions at specific points in space where flow might be quite different than bulk flow in the aquifer. Averages should show increasing agreement, and that was found to be the case here. The average velocity for all depths measured with the PVP was found to be 30.2 ± 7.5 cm/d for single detector tests and 34.7 ± 13.1 cm/d in two-detector tests (Table 1), where the reported errors represent one standard deviation in each case. With the measurement uncertainty taken into account, there is reasonable agreement between the bulk velocity and the PVP averages, though a slight positive bias cannot be ruled out. The possibility of a bias is discussed further in the section comparing direct velocity measurements.

[29] The PVP method permits velocity magnitudes to be determined on the basis of either single- or multiple-detector tests. The former tests require a foreknowledge of the flow direction, while the latter tests permit velocity direction to be estimated from the data. The magnitudes of the velocities estimated in single detector tests, assuming the average flow direction to apply (see Table 1; detector 2 is shown in Figure 3), showed variations with depth that generally matched those estimated by the two-detector method (for which velocity directions were computed from the data) (Figure 3). The magnitudes sometimes differed, but with the exception of a single point at about 275 cm depth the differences were not usually more than a few percent. In contrast, measured velocity directions varied considerably from those based on the orientation of the PVP relative to the sheet piles (Table 1). On a point by point basis there is no independent way to know for certain whether or not the PVP directions are accurate. However, the average directions (79.6° on the basis of the orientation of the injection port relative to the expected bulk flow direction, and 75.9° from the PVP measurements) showed remarkable agreement,

Table 1. Summary of Results for Assessing Single- Versus Two-Detector Velocity Estimates and Direction Estimates[a]

Depth (cm)	Detector	α Assumed ±10° (deg)	Darcy Velocity ±1.7	Single-Detector PVP Velocity (cm/d)	Two-Detector PVP Velocity (cm/d)	α Two-Detector PVP Direction (deg)
180	1	80	20.0	30.3	36.3	109
	2	80	20.3	25.5	36.3	109
190	1	80	20.3	39.7	31.6	38
	1	85	20.4	25.6	31.6	38
	2	85	20.7	31.4	31.6	38
200	1	80	20.7	46.1	23.3	48
	1	80	20.3	21.0	23.3	48
	2	80	20.6	24.4	23.3	48
210	1	80	20.3	38.2	49.7	115
	2	80	20.2	30.2	49.7	115
220	1	90	20.3	31.5	40.4	34
	2	90	20.4	41.7	40.4	34
230	1	90	20.2	19.4	19.9	56
	2	90	20.2	22.7	19.9	56
240	1	90	20.3	25.8	25.1	73
	2	90	20.5	28.0	25.1	73
250	1	70	20.7	31.5	31.3	76
	2	70	20.8	30.4	31.3	76
260	1	70	20.4	40.6	46.7	105
	2	70	20.4	33.2	46.7	105
270	1	70	20.4	41.1	63.7	125
	2	70	20.4	27.0	63.7	125
280	1	70	20.6	19.5	20.4	56
	2	70	20.7	20.6	20.4	56
Averages		79.6	20.4	30.2	34.7	73.2

[a]Note that the alpha angle is the angle between the flow direction and the injection port on the PVP instrument.

supporting the notion that the flow directions derived from the detectors were meaningful.

[30] Estimates of velocity magnitudes in the single detector tests tended not to be strongly affected by the assumed flow direction, but they were found to be sensitive to installation methods. On the day that the PVP was driven into the ground with a vibrating hammer, the bulk velocity within the alleyway was about 24 cm/d. The velocities measured with the PVP were about 5 cm/d, and exhibited low variability with depth. These data suggest that the porous medium was altered, perhaps by compaction, during the hammering. As discussed by *Labaky et al.* [2007], a skin of reduced K next to the probe would be expected to cause a negative bias in estimated velocity. On the basis of these data, it appears that more representative velocity estimates are obtained when PVPs are installed using the jetting technique rather than driving, notwithstanding the possible positive bias noted above. Further testing is necessary to evaluate direct push methods and augering; however, the latter method is expected to introduce positive biases to the measurements due to the zone of disturbed material next to the instrument that would follow the removal of the augers.

3.2. Comparisons of v Trends With K Trends

[31] The average hydraulic conductivity from the permeameter testing was 6.06×10^{-3} cm/s with a standard deviation of 2.4×10^{-3} cm/s. The maximum variation in K was about an order of magnitude (Figure 4). The observed trends in K with depth, i.e., peaks and valleys, were found to correspond very well with those of the PVP estimated velocities, although the latter only varied in magnitude by a factor of about 2.

[32] Hydraulic conductivity was also estimated from slug tests and again the results are quite encouraging. The permeameter- and slug test-derived K values are similar in magnitude and appear to follow similar trends, though the slug test data were less variable with depth. This dampening is thought to be due to a larger volume of aquifer being sampled during the slug tests, despite the small screened

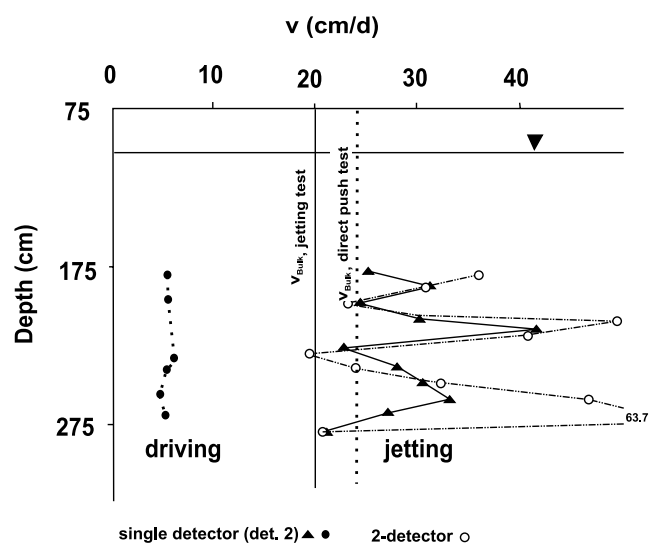

Figure 3. Summary of results for PVP comparison tests. Here "driving" refers to a PVP installed by driving with a vibrating hammer, and "jetting" refers to a PVP installed by driving an open casing to the bottom of the aquifer, flushing it out with a high-pressure stream of water, emplacing the PVP, and then withdrawing the casing.

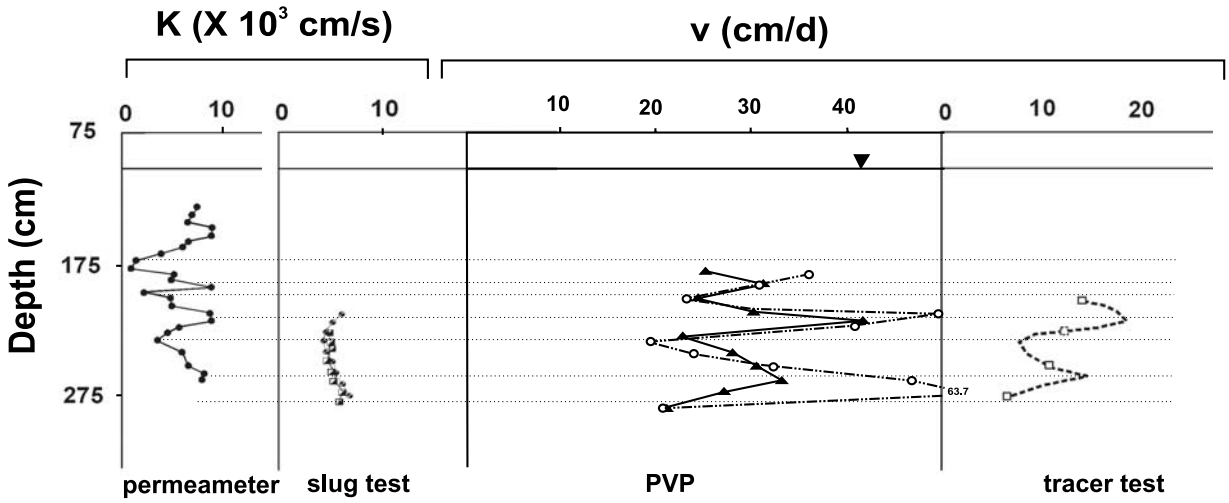

Figure 4. Comparison of vertical trends in hydraulic conductivity determined by permeameter and slug testing with velocities determined by PVP measurements and a tracer test [*Barker et al.*, 2000]. The dashed line connecting the data points of the tracer test was drawn with the sole purpose of illustrating possible compatibility with the other data sets.

interval, compared to the other methods. An attempt was made to correct for this at the data interpretation stage by including a generalized Hvorslev shape factor in the K estimation calculations [*Hvorslev*, 1951]. However, this correction assumes a homogeneous porous medium, a limiting assumption in these tests. Despite the dampened nature of the slug test data, a K minimum at a depth of about 235 cm is common to the permeameter and slug tests, and corresponds with a zone of lower velocity as reflected in the PVP measurements. Thus, these data sets appear to agree with one another remarkably well.

[33] The results discussed above can be compared to the results of a tracer test performed by *Barker et al.* [2000]. In that test, a conservative tracer was released from wells at the open end of the sheet pile alleyway (Figure 2). The tracer movement was tracked at a multilevel fence (G212) located about halfway along the alleyway (Figure 2). Average tracer velocities were determined as a function of depth (Figure 4). The spacing of the tracer detection points was about 20 cm, and as a result the details of the velocity profile are lost. There appears to be an overall decline in velocity with depth in the tracer data, and despite the limited number of data points it is possible to qualitatively reconcile the observed trend with those observed in K and v determined from PVP measurements (Figure 4). The differences in magnitude between the tracer velocities and the PVP velocities is the result of different pumping rates at the end of the alleyway on the days of the experiments ($Q_{tracer\ test}$ = 130 mL/min and $v_{bulk} \sim 12$ cm/d). It must be remembered that while the PVP velocities represent point velocities, the tracer estimates represent velocities averaged from the source wells to the G212 monitoring wells. Direct comparisons between these methods is not likely to be meaningful in all cases. Any comparability of the results here is attributable to the relatively homogeneous nature of the Borden aquifer over the scale of the alleyway.

3.3. Comparison of Direct Velocity Measurements

[34] Five methods of measuring groundwater velocity directly were compared in this work: the PVP and GeoFlo® instrument methods, two methods of borehole dilution, and the colloidal borescope. In general, the trends in the relative magnitudes of the measured velocities appeared to be consistent among the methods. Although only 4 of the colloidal borescope measurements were of sufficient quality to be converted into velocity estimates, the velocity minima and maxima were consistent with other data. Where the methods differed was in the absolute magnitudes of the velocity estimates (Figure 5).

[35] The two data sets that agreed best were those from the PVP and Geoflo® instrument tests. Average velocities were 30 to 35 cm/d in the PVP tests (Table 1), and 35 cm/d, in the Geoflo® tests. This agreement between independent tests with different installation methods, suggests that the bulk velocity may have underestimated the local flow rate near the PVP and test well locations (Figure 2). Between these two methods, an advantage of the PVPs is that velocities were generated without calibration or other empirical correction factors, the same was not true of the Geoflo® meter velocities. The borehole dilution velocities from the drive point wells appeared to underestimate the true values in a fashion similar to the drive point PVP measurements (see section 3.1). Borehole dilution experiments conducted at the screened wells (boreholes 1 and 2) appeared to overestimate the groundwater velocity by almost an order of magnitude in some cases. The colloidal borescope measurements also overestimated the velocities by an order of magnitude or more.

[36] The reasons for the poor performances of the borehole dilution and colloidal borescope methods are not known for certain, although in the former case the possibility that density flow contributed to this error cannot be ruled out (i.e., tracer movement may have occurred out of the well screen because of density driven flow in addition to background groundwater flow). Both techniques have provided useful data in other experiments, at other sites, or in higher-velocity regimes. Nevertheless, the results of these comparisons show that the PVP field measurements match or exceed the quality of measurements from other established

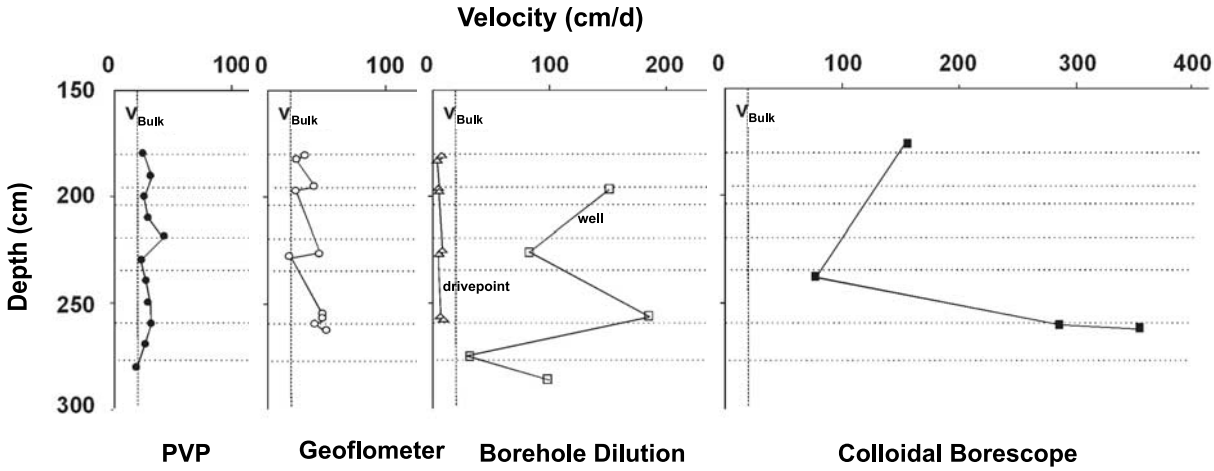

Figure 5. Comparison of velocity trends with depth as measured with various direct velocity measurement instruments. Here v_{Bulk} refers to a bulk average velocity between the sheet pile boundaries shown in Figure 1. The PVP data presented here are the same data plotted as the single-detector measurements in Figures 3 and 4. The scale was altered for ease of comparison with the various other methods.

methods of groundwater velocity estimation. Furthermore, this work demonstrates that the PVP instrument is capable of measuring groundwater velocity to a reasonable degree of accuracy in a field setting, despite its possible sensitivity to skin effects as discussed by *Labaky et al.* [2007]. In addition, the PVP methods offers the opportunity to collect highly detailed velocity data sets with unmatched vertical resolution.

4. Conclusions

[37] This work has shown that PVPs can measure groundwater velocities in a field setting with accuracy equal to or exceeding (on the basis of a comparison to the estimated bulk velocity) other methods of direct velocity measurement, provided the instrument is installed with minimal disturbance to the aquifer sediments. Emplacing PVPs by driving with a vibrating hammer led to negative biases in the measured velocities, compared to the estimated bulk velocity. A similar result was obtained when borehole dilution wells were driven into the aquifer. Jetting the PVPs into place may have introduced a slight positive bias in velocities compared to the bulk velocity, but a positive bias was also found in the Geoflo® meter data collected from a developed well. Therefore, the positive "biases" may be at least partly due to velocity variations in the alleyway, rather than being true biases. These findings are consistent with expectations for PVP performance based on laboratory work, and they establish the jetting method as the preferred method for installing PVPs in noncohesive, unconsolidated porous media.

[38] This work has demonstrated that the PVP is useful in sand where flow is predominantly horizontal. Additional work is needed to extend the method to other environments. Individual measurements may take minutes to several hours to complete, depending on the groundwater velocity. The method as implemented here returns velocities averaged over those time intervals, rather than point-in-time velocities, and this should be taken into account when measurements are made in rapidly changing environments. However, the PVP method offers the opportunity for automation and the acquisition of detailed records of velocity variations through time, on the scale of days or weeks, and variations in space, on the scale of centimeters (individual measurements) or greater.

[39] **Acknowledgments.** The NSERC/Motorola/ETI Industrial Research Chair in Groundwater Remediation, NSERC, CRESTech, the National Council for Scientific Research of Lebanon (NCSR), OGSST, and NSF under grant 0134545 are acknowledged for funding this work. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the National Science Foundation. Carl McElwee performed the high-definition slug tests; Peter Kearl performed the colloidal borescope measurements; and Bob Ingleton, Paul Johnson, and Greg Friday assisted with probe construction and field installation.

References

Alden, A. S., and C. L. Munster (1997), Field test of the in situ permeable ground water flow sensor, *Ground Water Monit. Rem.*, *17*(4), 81–88, doi:10.1111/j.1745-6592.1997.tb01267.x.

Ball, W. P., C. H. Buehler, T. C. Harmon, D. M. Mackay, and P. V. Roberts (1990), Characterization of a sandy aquifer material at the grain scale, *J. Contam. Hydrol.*, *5*, 253–295, doi:10.1016/0169-7722(90)90040-N.

Ballard, S. (1996), The in situ permeable flow sensor: A ground-water flow velocity meter, *Ground Water*, *34*(2), 231–240, doi:10.1111/j.1745-6584.1996.tb01883.x.

Barker, J. F., B. J. Butler, E. Cox, J. F. Devlin, R. Focht, S. M. Froud, D. J. Katic, M. McMaster, M. Morkin, and J. Vogan (2000), *Sequenced Reactive Barriers for Groundwater Remediation*, 376 pp., Lewis, Boca Raton, Fla.

Bartolino, J. R., and J. C. Cole (2002), Ground-water resources of the middle Rio Grande Basin, *U. S. Geol. Surv. Circ.*, *1222*, 132 pp.

Clark, I. D., and P. Fritz (1997), *Environmental Isotopes in Hydrogeology*, 328 pp., CRC Press, Boca Raton, Fla.

Devlin, J. F., and J. F. Barker (1996), Field investigation of nutrient pulse mixing in an in situ biostimulation experiment, *Water Resour. Res.*, *32*(9), 2869–2877, doi:10.1029/96WR01128.

Devlin, J. F., and C. M. McElwee (2007), Effects of measurement error on horizontal hydraulic gradient estimates in an alluvial aquifer, *Ground Water*, *45*(1), 62–73, doi:10.1111/j.1745-6584.2006.00249.x.

Drost, W., D. Klotz, A. Koch, H. Moser, F. Neumaier, and W. Rauert (1968), Point dilution methods of investigating ground water flow by means of radioisotopes, *Water Resour. Res.*, *4*(1), 125–145, doi:10.1029/WR004i001p00125.

Gaspar, E., and M. Oncescu (1972), *Radioactive Tracers in Hydrology*, transl. from Romanian by M. Marinescu, 342 pp., Elsevier, Amsterdam.

Gierczak, R., J. F. Devlin, and D. Rudolph (2006), Combined use of laboratory and in situ hydraulic testing to predict preferred flow paths of solutions injected into an aquifer, *J. Contam. Hydrol.*, *82*, 75–98, doi:10.1016/j.jconhyd.2005.09.002.

Guthrie, M. (1986), Use of a Geo flowmeter for the determination of ground water flow direction, *Ground Water Monit. Rev.*, *6*(2), 81–86, doi:10.1111/j.1745-6592.1986.tb01244.x.

Halevy, E., H. Moser, O. Zellhofer, and A. Zuber (1967), A critical review, in *Isotopes in Hydrology*, pp. 531–564, Int. At. Energy Agency, Vienna.

Hvorslev, M. J. (1951), Time lag and soil permeability in groundwater observations, *Bull. 36*, Waterw. Exp. Stn., Corps of Eng., U. S. Army Corps of Eng., Vicksburg, Miss.

K-V Associates, Inc. (1983), Groundwater flowmeter system: Operations and maintenance manual, 21 pp., Mashbee, Mass.

Kalbus, E., F. Reinstorf, and M. Schirmer (2006), Measuring methods for groundwater-surface water interactions: A review, *Hydrol. Earth Syst. Sci.*, *10*, 873–887.

Kasnavia, T., D. Vu, and D. A. Sabatini (1999), Fluorescent dye and media properties affecting sorption and tracer selection, *Ground Water*, *37*(3), 376–381, doi:10.1111/j.1745-6584.1999.tb01114.x.

Kearl, P. M. (1997), Observations of particle movement in a monitoring well using the colloidal borescope, *J. Hydrol.*, *200*, 323–344, doi:10.1016/S0022-1694(97)00026-7.

Kerfoot, W. B., and V. A. Massard (1985), Monitoring well screen influences on direct flowmeter measurements, *Ground Water Monit. Rev.*, *5*(4), 74–77, doi:10.1111/j.1745-6592.1985.tb00942.x.

Labaky, W. (2004), Theory and testing of a device for measuring point-scale groundwater velocities, Ph.D. dissertation, Dep. of Earth Sci., Univ. of Waterloo, Waterloo, Ont., Canada.

Labaky, W., J. F. Devlin, and R. W. Gillham (2007), Probe for measuring groundwater velocity at the centimeter scale, *Environ. Sci. Technol.*, *41*(24), 8453–8458, doi:10.1021/es0716047.

LeBlanc, D. R., S. P. Garabedian, K. M. Hess, L. W. Gelhar, R. D. Quadri, K. G. Stollenwerk, and W. W. Wood (1991), Large-scale natural gradient tracer test in sand and gravel, Cape Cod, Massachusetts: 1. Experimental design and observed tracer movement, *Water Resour. Res.*, *27*(5), 895–910, doi:10.1029/91WR00241.

Le Sieur, M. E. (1999), Characteristics of remaining solvent DNAPL distribution and mass discharge after frequent removal from a sandy aquifer: A field study, M.Sc. thesis, 140 pp., Dep. of Earth Sci., Univ. of Waterloo, Waterloo, Ont., Canada.

Mackay, D. M., D. L. Freyberg, and P. V. Roberts (1986), A natural gradient experiment on solute transport in a sand aquifer: 1. Approach and overview of plume movement, *Water Resour. Res.*, *22*(13), 2017–2029, doi:10.1029/WR022i013p02017.

Mas-Pla, J., T.-C. J. Yeh, T. M. Williams, and J. F. McCarthy (1997), Analyses of slug tests and hydraulic conductivity variations in the near field of a two-well tracer experiment site, *Ground Water*, *35*(3), 492–501, doi:10.1111/j.1745-6584.1997.tb00110.x.

McElwee, C. D. (2001), Application of a nonlinear slug test model, *Ground Water*, *39*(5), 737–744, doi:10.1111/j.1745-6584.2001.tb02364.x.

McElwee, C. D., and M. A. Zenner (1998), A nonlinear model for analysis of slug-test data, *Water Resour. Res.*, *34*(1), 55–66, doi:10.1029/97WR02710.

Molson, J. W., and E. O. Frind (1994), SALTFLOW user guide, Waterloo Cent. for Groundwater Res., Univ. of Waterloo, Waterloo, Ont., Canada.

Momii, K., K. Jimmo, and F. Hirano (1993), Laboratory studies on a new laser Doppler-velocimeter system for horizontal groundwater velocity measurements in a borehole, *Water Resour. Res.*, *29*(2), 283–291, doi:10.1029/92WR01958.

Perlmutter, N. M., and M. Lieber (1970), 1970Dispersal of plating wastes and sewage contaminants in ground water and surface water, South Farmingdale–Massapequa area, Nassau county, New York, *U. S. Geol. Surv. Water Supply Pap.*, *1879-G*, 67 pp.

Pitrak, M., S. Mares, and M. Kobr (2007), A simple borehole dilution technique in measuring horizontal ground water flow, *Ground Water*, *45*(1), 89–92, doi:10.1111/j.1745-6584.2006.00258.x.

Robertson, W. D., and J. A. Cherry (1989), Tritium as an indicator of recharge and dispersion in a groundwater system in central Ontario, *Water Resour. Res.*, *25*(6), 1097–1109, doi:10.1029/WR025i006p01097.

Schillig, P. (2008), Microbial activity during biodegradation and its effects on groundwater velocity in a contaminated aquifer, M. S. thesis, 136 pp., Dep. of Geol., Univ. of Kans., Lawrence.

Schlichter, C. S. (1905), Field measurements of the rate of movement of underground waters, *U. S. Geol. Surv. Water Supply Pap.*, *140*, 85 pp.

Sudicky, E. A. (1986), A natural gradient experiment on solute transport in a sand aquifer: Spatial variability of hydraulic conductivity and its role in the dispersion process, *Water Resour. Res.*, *22*(13), 2069–2082, doi:10.1029/WR022i013p02069.

van der Kamp, G., L. D. Luba, J. A. Cherry, and H. Maathuis (1994), Field study of a long and very narrow contaminant plume, *Ground Water*, *32*(6), 1008–1016, doi:10.1111/j.1745-6584.1994.tb00940.x.

Xie, X. (1994), Solute transport and remediation in the interface zone: Mathematical modelling and field investigation, Ph.D. thesis, 135 pp., Dep. of Earth Sci., Univ. of Waterloo, Waterloo, Ont., Canada.

Zemansky, G. M., and C. D. McElwee (2005), High-resolution slug testing, *Ground Water*, *43*(2), 222–230, doi:10.1111/j.1745-6584.2005.0008.x.

J. F. Devlin, Department of Geology, University of Kansas, Lindley Hall, 1475 Jayhawk Boulevard, Lawrence, KS 66049, USA. (jfdevlin@ku.edu)

R. W. Gillham, Department of Earth Sciences, University of Waterloo, Waterloo, ON N2L 3G1, Canada.

W. Labaky, Schlumberger Middle East, 6th Floor, Al Masaood Tower, P.O. Box 21, Hamdan Street, Abu Dhabi, United Arab Emirates.