Do no-purge sampling techniques yield a significant difference relative to traditional sampling techniques for shallow groundwater



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ABSTRACT

The potential implications of no-purge sampling from watertable monitoring wells on groundwater chemistry as evaluated analytically are indicated to be minimal. Groundwater chemistry between samples collected prior to and following purging from watertable monitoring wells is observed to be comparable, such that if analyte concentrations are compared to guideline values or used to determine the extent of a contaminant plume, the results would be similar.

RÉSUMÉ

Les implications potentielles de la méthode sans-purge utilisée dans les puits de la nappe phréatique sur la chimie des eaux souterraines étaient évaluées analytiquement et étaient observées d'être minimes. En plus, la chimie des eaux souterraines des échantillons obtenus avant et après la purge des puits de la nappe phréatique était relativement comparable. Alors, afin de comparer les concentrations aqueuses aux critères de qualité ou de déterminer l'ampleur de la contamination, les résultats seraient semblables.

1 INTRODUCTION

Prior to sample collection, purging is the generally accepted practice to remove stagnant groundwater present within the monitoring well that may not be representative of porewater within the surrounding geologic material. Purging consists of removing a given number of well volumes of groundwater, whereby a well volume has been operationally defined as the volume of water contained within the well casing, or wellbore, under hydrostatic conditions. Commonly, the number of well volumes purged prior to sample collection has been (to some degree arbitrarily) designated as three. Based on the absence of a rigorous foundation for the three well volume convention, the literature reports numerous studies that have attempted to determine the appropriate purge volume or sampling method required in order to obtain groundwater samples representative of subsurface conditions. For example, such studies have proposed techniques that minimize purge volumes and agitation within the monitoring well (e.g., low flow sampling) or eliminate purge volumes altogether (e.g. no purge sampling and passive samplers). The use of no-purge sampling techniques assumes that natural purging of the screened interval occurs due to ambient groundwater flow.

No-purge sampling is attractive because it eliminates the costs associated with collecting and disposing of purged groundwater. However, some researchers have suggested that no-purge sampling may not be broadly applicable, specifically in cases where the static water level resides above the screened interval (Varljen, 1997). Above the screened interval, the potential for slower flushing, or the occurrence of stagnant water has been demonstrated by previous authors (e.g. Robin et al., 1987). As such, the application of no-purge sampling appears most appropriate in cases where the well has been screened across the air-water interface (i.e. shallow wells screened across the water table).

In such cases, no-purge sampling may actually provide more representative results as turbidity caused by purging may be reduced, and volatilization of organic compounds during purging or turbulent recharge of the monitoring well may be reduced (turbulent recharge is assumed to occur in low permeability formations, after purging the well, water entering the well will cascade down the screen).

To evaluate the applicability of no-purge sampling with respect to shallow, water-table monitoring wells, a conceptual understanding of well hydraulics is presented, followed by a comparison of groundwater samples collected prior (pre-purge) and following (post-purge) purging of the monitoring well.

2 METHODOLOGY

2.1 Field Data Collection

Samples collected prior to purging were obtained using inertial pumps (i.e. waterra tubing), bailers, or passive samplers (i.e. Hydrasleeve[™] collection devices). Samples collected following purging were obtained using bailers or inertial pumps. Generally, purging consisted of

the removal of 3 casing volumes. Where necessary, samples obtained both prior to and following purging were filtered and preserved using standard industry methods, and placed in laboratory supplied containers for analysis.

Monitoring wells used for the study represent standard 5 cm (2") diameter, shallow monitoring wells, constructed of PVC. A filter pack or natural filter has been emplaced surrounding the screened interval overlain by an annular bentonite-based seal. Installation of the monitoring wells was typically done using solid/hollow-stem drilling rigs. Generally, monitoring wells were screened across or immediately below the water table.

To simplify comparisons, only pre and post-purge paired samples collected on the same or consecutive days have been considered to remove possible effects of seasonality, varying static water levels and groundwater recharge events on groundwater quality.

Two datasets were generated of pre and post-purge water chemistry data collected by WorleyParsons Komex personnel from various sites within Alberta, from 1998 to 2006. One set represents paired samples collected using bailers or inertial pumps and consisting of chloride (Cl), sulphate (SO₄), iron (Fe), benzene, ethylbenzene, and xylenes. The second set represents pre-purge samples collected using passive sampling devices, and focussing on dissolved aluminium (Al).

2.2 Dataset Manipulation

For each parameter, paired sets were developed of prepurge and post-purge concentrations exceeding the analytical detection limits. Data pairs below the detection limit were culled, while single values below detection limits in a data pair were assigned a concentration of one half the detection limit. Zeiner (1994) noted that assigning a value of zero to non-detected results would overestimate the sensitivity of the laboratory, while assigning a value equal to the detection limit would overestimate the parameter concentration.

Each dataset was log-transformed to capture concentration ranges over several orders of magnitude. Linear regression analysis was performed on the transformed data of pre and post-purge samples with more than 10 sample pairs). The intercept for all linear regression lines was not constrained.

3 CONCEPTUAL ANALYSIS

As an initial step, general well hydraulics under purged and no-purge conditions were considered with respect to their implications for no-purge sampling.

3.1 Radius of Influence

Groundwater chemistry may exhibit considerable spatial variability, even at scales on the order of centimetres (Cozzarelli et al., 2001), thus comparability of pre and post-purge samples may be affected by the area of the aquifer, from which groundwater is obtained in each case.

The area of a homogeneous, isotropic aquifer from which groundwater is obtained following purging is approximated using a volume displacement calculation. Accordingly, if three casing volumes are removed from a water table well with a given radius (r_c) and static water

column height (h), the radial extent of porous medium which must be drained (r_f) to replace this volume may be represented as (assuming 30% effective porosity):

$$3\pi r_c^2 h = \left(\pi (r_c + r_f)^2 h - \pi r_c^2 h\right) 30\%$$
(1)

Solving for r_f,

$$2.3r_c = r_f \dots$$

Thus, for a 5 cm (2") wide monitoring well, groundwater is predicted to be derived from a zone extending in a radial direction, approximately 6 cm from the casing (Equation 2).

This is similar to the radius of influence under natural hydraulic gradients, whereby a monitoring well installed in a homogeneous, isotropic porous medium will capture groundwater from a lateral area approximately twice its diameter (sources reviewed by Wilson et al., 1997). This phenomenon is termed borehole dilution and arises as flowlines within the less permeable geologic material converge towards the more highly permeable monitoring Specifically, for a standard 5-cm diameter PVC well. monitoring well (without filter pack), the capture zone would be a 10-cm wide section of the aquifer located hydraulically upgradient of the monitoring well. Thus samples obtained during pre and post-purge sampling should arise from areas of similar distances from the well. However, it should be noted that purging may obtain groundwater located hydraulically downgradient after having already passed through the well.

3.2 Flushing Rate

No-purge sampling assumes that purging of the screened interval occurs under natural gradients. To assess the rate of natural flushing, recall that the probable width of homogeneous, isotropic porous media from which groundwater flow converges from and enters a monitoring well is:

$$2d_f = d_{mw} \tag{3}$$

Where: d_f = width of the porous medium from which groundwater flow lines converge towards the monitoring well (after Wilson et. al., 1997) d_{mw} = width of the monitoring well

Assuming no change in storage,

$$Q_f = Q_{mw} \tag{4}$$

Where: Q_f = volumetric flowrate through a given cross sectional area of the porous medium

Q_{mw} = volumetric flowrate through a given cross sectional area of the monitoring well (screen)

Thus, if:

$$Q = (A)(q_x) \tag{5}$$

Where: A = cross sectional area (whereby monitoring well

is approximated as a square)

 q_x = horizontal Darcy flux

Assuming horizontal flow in the aquifer, the average linear groundwater flow velocity within the monitoring well can be determined based on:

$$q_{x(mw)} = \frac{(q_{x(f)})(2A_{mw})}{A_{mw}}$$
(6)

and thus

$$q_{x(mw)} = \left(q_{x(f)}\right)(2) \tag{7}$$

Where: A_f = cross sectional area of porous medium from which flowlines converge towards the monitoring well

 $q_{x(f)}$ = horizontal Darcy flux within the monitoring well, and also equivalent to the average linear horizontal groundwater velocity in the monitoring well (as porosity is 100%)

 $q_{x(f)}$ = horizontal Darcy flux within the porous medium and can be calculated via:

$$q_{x(f)} = \left(-K_x\right) (\nabla h) \tag{8}$$

Where: K_x = horizontal hydraulic conductivity of the porous medium

 $\mathbf{\nabla} \mathbf{h}$ = horizontal hydraulic gradient of the porous medium

Thus the residence time within a monitoring well is:

$$t_{R} = \frac{v_{x(mw)}}{d_{r(mw)}} = \frac{(-K_{x})(\nabla h)(2)}{d_{r(mw)}}$$
(9)

Where: t_r = residence time of groundwater within the monitoring well

 $d_{r(mw)} = diameter of the monitoring well$

Thus based on Equation 9, Figure 1 presents the residence times for groundwater flowing through a 5 cm (2") diameter monitoring, under a horizontal hydraulic gradient of 0.01 m/m and horizontal hydraulic conductivities ranging from 1 x 10^{-7} m/s to 1 x 10^{-4} m/s. Figure 1 estimates that monitoring wells screened across the water table can be flushed on the order of a day or less in aquifers of higher permeability (e.g. $>1 \times 10^{-4}$ m/s), or on the order of hundreds of days or more in aguifers of lower permeability (e.g. $<1 \times 10^{-7}$ m/s). These estimates provide an indication of the natural flushing rates which could be expected for monitoring wells installed in A similar calculation for homogeneous aquifers. inhomogeneous aquifers is more difficult, requiring consideration of inflow and outflow from the well through intervals of varying thickness and permeability.



Figure 1. Groundwater Residence Times in Monitoring Well and Associated Volatile Loss from 1 m Thick Water Column to Headspace

3.2.1 Volatile Loss from Stagnant Water Column

The implications of slower flushing rates on groundwater chemistry include increased opportunity for volatilization of organic compounds from the water column to the headspace of the well. Under ambient conditions, the amount of volatile loss will depend primarily on the contact time between groundwater and the headspace within the monitoring well (i.e., the residence time of groundwater within the monitoring well). Assuming that diffusion through the aqueous phase is rate limiting, and using benzene to illustrate the potential for volatile loss from the water column (after Schwarzenbach et al., 2003) the diffusive flux from the water column may be represented as:

$$DiffusiveFlux = D_{Benezene} \times \nabla C_{Benzene}$$
(10)

Where: $D_{Benzene}$ = aqueous diffusion constant for benzene in water, 5.24 x 10⁻⁶ cm²/s at 5°C (Interpolated from Table 20.3 in Schwarzenbach et al., 2003) $\mathbf{\nabla}$ C = concentration gradient

And,

$$\nabla C_{Benzene} = \frac{C_{BenzeneWater} - C_{BenzeneAir}}{\delta_{Water}}$$
(11)

$$\label{eq:Where:C_BenzeneWater} \begin{split} & \text{Where:} C_{\text{BenzeneWater}} = \text{Concentration of Benzene in Water} \\ & C_{\text{BenzeneAir}} = \text{Concentration of Benzene in Air} \\ & \overline{\delta_{Water}} = \text{Thickness of Diffusive Film in Water} \end{split}$$

And,

$$\delta_{Water} = \sqrt{D_{Benzene} \times t_r} \tag{12}$$

Generally, mass release (M_r):

$$M_{r} = D_{Benzene} \left(\frac{C_{BenzeneWater} - C_{BenzeneAir}}{\delta_{Water}} \right) \times t_{r} \times Area$$
(13)

Where: Area = Cross sectional area of air-water interface t_r = residence time of groundwater in well

To simplify, assume: $C_{BenzeneAir} = 0$

And,

$$Fraction \operatorname{Re} leased = \frac{M_r}{C_{InitWater} \times V_{Water}}$$
(14)

Where: C_{InitWater} = initial benzene concentration in the water column V_{Water} = volume of the water column

Thus,

$$Fraction \operatorname{Re} leased = \frac{\frac{D_{Benzene} \times t_r \times Area}{\delta_{Water}}}{V_{Water}}$$
(15)

Based on Equation 15, Figure 1 shows the fraction of total benzene loss through volatilization from a 1 m thick water column. Turbulent flow in the water column (Britt, 2005) is not captured by Equation 15, but its effects are expected to be minimal with respect to longer residence times, where volatile losses are larger.

As the residence time of groundwater within the well approaches 300 days, the estimated loss of benzene through volatilization is approximately 10% (Figure 1). Losses of this magnitude are not considered significant, being within acceptable reproducibility of field duplicates. While volatile losses could become significant for longer residence times than those shown in Figure 1,, it should be noted that turbulent recharge of the well after purging would likely induce volatile losses of similar or of greater magnitude. Furthermore, the percent loss of benzene will decrease linearly with increasing water column thickness, as the fraction released depends only on the volume of water volatilization rate is independent of water column thickness (Equation 15).

If a water column of significant thickness accumulates above the screened interval, it is less likely to be actively flushed by ambient flow of groundwater. Volatile loss could be compounded in such cases if the stagnant water above the screen is not completely removed during sampling and remains year over year.

3.2.2 Aeration of Stagnant Water Column

Similar to volatilization, aeration of the water column via diffusion of oxygen from the headspace will occur under ambient conditions. Anaerobic water could experience geochemical changes due to aeration of the water column. Dissolved hydrocarbons are typically degraded more quickly under aerobic relative to anaerobic conditions (Suarez et al., 1999), while dissolved metals (e.g., Fe) may precipitate under aerobic conditions.

Figure 2 illustrates the diffusive ingress of a conservative dissolved oxygen plume through an initially anaerobic water column. This scenario assumes the headspace immediately above the water column remains continuously saturated with oxygen and diffusion of oxygen through the water column is governed by:

$$\frac{C}{C_0} = ERFC\left(\frac{x}{\sqrt{4(D_{Oxygen})(t_r)}}\right)$$

Where: C/C_0 = Concentration at time(t), relative to initial concentration, C_0

x = Vertical distance along water column

 D_{Oxygen} = Aqueous diffusion constant for oxygen (1.30 x 10⁻⁵ cm²/s at 5°C, based on Schwarzenbach et al., 2003)

t_r = residence time



Figure 2. Oxygen Diffusion through Stagnant Water Column

Oxygen diffusion through the water column is slow. For example, a 10%, or greater increase in dissolved oxygen content is predicted to be limited to depths shallower than 0.5 m below the air-water interface in the well, after 1 year. (Figure 2). This result indicates that water samples collected sufficiently below the top of the water column (e.g., using inertial pumps) are unlikely to be notably affected by diffusive aeration. Although longer groundwater residence times within the monitoring well or in-well mixing processes could facilitate dissolved oxygen penetration, oxidation or biodegradation reactions between inorganic or organic compounds within the water column would be expected to consume dissolved oxygen as it entered the top of the water column. Also in low permeability formations, where residence times are largest, in-well mixing is expected to be minimal. Relative to the predicted degree of diffusive oxygenation, greater oxygenation is expected due to agitation and/or turbulent recharge associated with purging of the water column.

As with increased volatilization losses, oxygen diffusion into stagnant water above the screened interval, could allow a somewhat greater degree of oxygenation. Therefore to minimize possible effects, water samples should be collected within the screened interval.

3.2.3 Sorption and Desorption

Sorption-related interactions of dissolved constituents with the PVC well materials will likely only be influential when trying to interpret low dissolved concentrations (e.g., Barcelona and Helfrich, 1986). For most practical purposes, monitoring well casings and screens constructed of PVC are not expected to significantly affect dissolved concentrations of hydrocarbons and inorganic constituents. Rather, the suspended sediment within the water column bears the greatest potential for the sequestration or release of dissolved compounds via de-sorptive/dissolution sorptive or reactions. Sequestration of dissolved constituents may occur as a result of sorption to the increased amount of turbidity within the water column during purging or sampling activities. Conversely, artificially elevated concentrations of dissolved constituents (specifically dissolved metals) may occur if suspended sediment generated during sampling or purging activities is not removed during filtration of the groundwater sample prior to acid preservation.

Passive samplers are expected to generate less turbidity during sampling because they are not associated with surging action or rapid movement through the water column. It is unclear if turbidity will be significantly different during sample collection using bailers or inertial pumps under purge and no-purge conditions. Groundwater collection by inertial pumps is more likely to be associated with increased turbidity than collection via bailers, although the skill and experience of the operator are also influential.

4 FIELD DATA

Time series plots of water chemistry data, or scatter plots of pre- and post-purge data are presented in the following sections. For visual reference the following were included on the scatter plots:

• A line with a slope of one (1 to 1 line), passing through the origin of the x and y axes. This line represents the location where the points would plot if preand post-purge samples yielded identical values.

• A pair of lines representing deviations of +/- 20 relative percent difference (RPD) from the 1-to-1 line. These lines represent the maximum acceptable RPD between field duplicates of groundwater samples (Zeiner, 1994).

4.1 Pre- and Post-Purge Data Collected Using Bailers or Inertial Pumps

While the regression statistics reported below represent linear regressions on log-transformed data, presentation of the data occurs using either linear or log scales. Linear scales are more useful for illustrating absolute concentration differences, while log scales better illustrate proportional concentration differences and datasets spanning many orders of magnitude.

4.1.1 Selected Major Ions – Chloride

In most cases the pre- and post-purge concentrations of Cl plotted near the 1to1 line, and (while difficult to see) often differed by less than 20 RPD (Figure 3). Although the R^2 coefficient for Cl drops to 0.87 if the high concentration (>100 mg/L) data points are not considered, good correlation is still demonstrated. Cl is not subject to rapid concentration changes related to factors such as volatilization or changing redox conditions, thus greater comparability between pre- and post-purge samples is expected.

Larger proportional differences in CI concentrations are noted at lower concentrations (Figure 3). These larger differences are related to small absolute concentration differences (<10 mg/L) that are not considered significant for the purposes of compliance groundwater monitoring.



Figure 3. Pre and Post-Purge Chloride Concentrations (Slope: $0.98 \text{ R}^2=0.91 \text{ n}=64$)

4.1.2 Selected Redox Indicators - Sulphate and Iron

 SO_4 concentrations were relatively comparable near the higher end of the concentration range of the dataset, with most data points falling within or near the 20 RPD lines (Figure 4). As with Cl, greater scatter outside the 20 RPD lines occurs for lower concentrations, less than 10 mg/L. In contrast, there are more scattered points, likely reflecting the potential influence of biologically-mediated redox reactions on SO_4 concentrations, relative to the chemically conservative nature of Cl. Such reactions are expected to have a greater proportional impact on the lower SO_4 concentrations. Similar scatter of SO_4 concentrations is also noted between samples collected immediately after purging, versus those collected after sufficient time has elapsed to allow the water column to approach full recovery. This observation suggests that inwell reactions or mixing may affect SO_4 concentrations, and that variability is inherent in SO_4 concentrations, regardless of the sampling method.

Pre- and post-purge Fe samples show increased scatter compared to CI, with an apparent bias towards higher concentrations in the pre-purge samples (Figure 5). Similar to that noted for SO₄, the scatter observed in Fe concentrations may also be attributable to in-well mixing and reactions, and occur irrespective of the sampling method (Figure 6). Relative to more conservative parameters (e.g., CI), dissolved Fe is more reactive and subject to several reactions that may increase or decrease dissolved Fe concentrations: Fesulphide precipitation, Fe-carbonate precipitation, Fe-(oxy)hydroxide precipitation, and reductive dissolution reactions. Fe concentrations may also be affected by sample turbidity. It is hypothesized that lower dissolved Fe concentrations associated with post-purge samples are due to increased aeration of the water column related to purging during turbulent recharge. Additional data may help to better evaluate the scatter and potential bias towards pre-purge samples.

The scatter and potential bias is of minimal significance for the purposes of compliance monitoring, where Fe concentrations are compared versus a guideline (e.g. 0.3 mg/L; Alberta Environment 2007), or where the presence of Fe is used to indicate the potential for natural attenuation of petroleum hydrocarbons. The scatter and potential bias may be of greater significance for more quantitative applications, such as mass balance calculations or calculations of mineral saturation indices.



Figure 4. Pre and Post-Purge Sulphate Concentrations (Slope:0.86 R²=0.87 n=61)



Figure 5. Pre and Post-Purge Iron Concentrations (Slope:0.91 R^2 =0.78 n=45)



Figure 6. Analyte Concentrations in Samples Collected Immediately After Purging versus Samples Collected Following Additional Recovery of the Static Water Level

4.1.3 Monoaromatics – Benzene, Ethylbenzene and Xylenes

Only eight sample pairs were available for toluene, thus toluene has not been included in the discussion.

The lower R² values indicate less positive correlation for hydrocarbons (Figures 7, 8, 9) than was noted for inorganic parameters. The lower values are mainly related to the increased scatter of the data in the lower concentration ranges. As noted previously, in these cases, smaller absolute changes may appear larger on a proportional basis, compared to at higher concentrations. Better correlation is observed for data pairs at higher concentrations. Armstrong et al. (2004) observed that variability in groundwater samples could occur regardless of purging and was more apparent in sites where vertical redox and contaminant concentration gradients were greatest.

Similar to the conclusions reported by Newell et al. (2000), the correlation between pre- and post-purge concentrations for benzene, ethylbenzene and xylenes

(Figures 7, 8, and 9) is such that either pre- or post-purge concentrations would provide a similar portrayal of the location of a contaminant plume. For example, in cases where the sampling protocol at a given well has been updated to include no-purge sampling, consistency is noted with respect to analyte concentrations relative to historically collected samples (e.g., Benzene in Figure 10). While it is conceivable that repeated purging and sampling events on consecutive days could yield similar reproducibility, sufficient data were not available for confirmation.



Figure 7. Pre and Post-Purge Benzene Concentrations (Slope:0.77 R²=0.71 n=42)



Figure 8. Pre and Post-Purge Ethylbenzene Concentrations (Slope:0.78 R²=0.64 n=34)



Pre Purge (mg/L)

Figure 9. Pre and Post-Purge Xylenes Concentrations (Slope:0.76 R^2 =0.55 n=34)



Figure 10. Historical Benzene Concentrations at a Single Monitoring Well

4.2 Pre-Purge Data Collected with Passive Samplers and Post-Purge Data Collected with Bailers or Inertial Pumps.

Sample collection via passive sampling devices was limited to dissolved metals. The dataset in these regards is limited to 8 data points, thus regression analysis has not been performed.

The data available for pre-purge samples collected with passive samplers, suggests a positive bias for postpurge samples with respect to Al. The comparatively lower Al values associated with the passive sampler may be related to the lower turbidity generated by this method. As turbidity was not evaluated at the time of sampling, and the dataset is limited in size, additional investigation is required to explore this hypothesis.



Figure 11. Passive Sampler versus Post-Purge Aluminium Concentrations (n=8)

5 CONCLUSIONS

Results to date indicate that pre-purge sampling is applicable for sampling shallow water-table wells for compliance monitoring applications requiring:

- comparison to guidelines; and
- definition of the extent of a contaminant plume.

With respect to these applications, the results presented herein indicate that pre-purge data do not give significantly different results versus post-purge data. Pre-purge data may even give more representative sample concentrations, being less influenced by factors such as turbulent recharge.

It should be noted that there is no "correct" sampling method, because both purge or no-purge sampling methods bear the potential to introduce an artificial bias on groundwater chemistry. However, the differences between pre and post purge samples are often less or comparable than those observed in consecutive sampling events, most notably in cases where steeper analyte concentration gradients exist. As well, the difference between pre and post-purge samples and the bias introduced by each is considered minimal relative to the effects of:

- interception and in-well mixing between multiple water bearing intervals or water types via long (e.g., 3 m) screens;
- seasonal effects, such as varying groundwater recharge;
- spatial heterogeneity in groundwater chemistry; and
- skill of sampling personnel.

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REFERENCES

- Alberta Environment. 2007. Alberta Tier 1 Soil and Groundwater Remediation Guidelines. June 2007. Edmonton, Alberta.
- Armstrong, J., Biggar, K., Morin, D., and McLeish, K. 2004. Sampling-Method-Induced Variability and its Influence on Interpretation of Natural Attenuation of Hydrocarbons. Proceedings from the 5th Joint IAH-CNC/CGS Conference.
- Barcelona, M.J. and Helfrich, J.A., 1986. Well Construction and Purging Effects on Ground-Water Samples. Environmental Science and Technology, 20: 1179-1184.
- Britt, S.L., 2005. Testing the In-Well Horizontal Laminar Flow Assumption with a Sand-Tank Well Model. Groundwater Monitoring and Remediation. 25(3): 73-81.
- Cozzarelli, I.M., Bekins, B.A., Baedecker, M.J., Aiken, G.R., Eganhouse, R.P., and Tuccillo, M.E., 2001. Progression of natural attenuation processes at a crude-oil spill site 1. Geochemical evolution of the plume. Journal of Contaminant Hydrology, 53(3-4):369-385.
- Newell, C.J., Lee, R.S., and Spexet, A.H., 2000. No-Purge Groundwater Sampling – An Approach for Long-Term Monitoring. Published by the American Petroleum Institute. No. 12, October.
- Robin, M.J.L., and Gillham, R.W., 1987. Field Evaluation of Well Purging Procedures. Groundwater Monitoring and Remediation. 7(4): 85-93.
- Schwarzenbach, R.P., Śschwend, P.M, and Imboden, D.M., 2003. Environmental Organic Chemistry. Second Edition. John Wiley and Sons, Inc. Hoboken. New Jersey. 1313 pp.
- Suarez, M.P, and Rifai, H.S., 1999. Biodegradation Rates for Fuel Hydrocarbons and Chlorinated Solvents in Groundwater. Bioremediation Journal. 3(4):337-362.
- Varljen, M.D., 1997. No-Purge Sampling: Here's the Beef. Groundwater Monitoring and Remediation. 17(3):87-89.
- Wilson, R.D., Mackay, D.M., and Cherry, J.A., 1997. Arrays of unpumped Wells for Plume Migration Control by Semi-Passive In Situ Remediation. Groundwater Monitoring and Remediation, 17(3): 185-193.
- Zeiner, S.T., 1994. Realistic Criteria for the Evaluation of Field Duplicate Sample Results. Re-Print from Proceedings of Superfund XV. Washington, DC, 29 November to 1 December 1994.