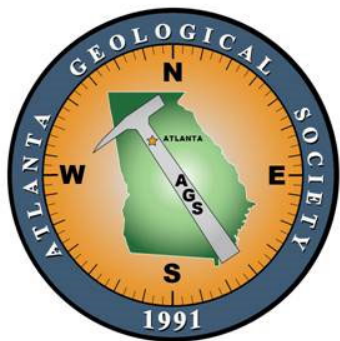




Atlanta Geological Society Newsletter

ODDS AND ENDS

Dear AGS members,

Zanclean – that's a word I had never heard before this week. You may have heard that the Mediterranean Sea had dried up about 6 million years ago. That was known as the Messinian Salinity Crisis. With a combination of tectonics and climate, the connection with the Atlantic Ocean was closed and there was less recharge into the basin. Then, 5.33 mya, the climate changed, some subsidence occurred between Spain and Morocco and the Atlantic came pouring in. Hence the Zanclean Flood, named after a pre-Roman city of Zancle in Sicily.

This is one of the things I love about geology; there's always something new to learn. I think that is one of major benefits of the AGS, the diversity of topics that presented in our presentations and in the newsletter. This could be part of your continuous learning process; not necessarily mandated professional development hours but rather the fulfillment of the goal of the best geologist is the one who has seen the most rocks, knows the most geology.

And since this is January and the start of the AGS membership year, this is my plug for you to renew your membership, to keep learning about more geology. I just renewed my dues; it took all of 2 minutes. <https://squareup.com/store/atlanta-geological-society>

Hope to see you on Tuesday.
Ben Benthowski, President

January Meeting

Join us **Tuesday, January 28, 2020** at the Fernbank Museum of Natural History, 760 Clifton Road NE, Atlanta GA. The social starts at 6:30 pm and the meeting starts at 7:00 p.m.

This month our presentation is “**A 1,200-yr Pollen Record of Environmental Changes in the Northern Gulf of Mexico**” presented by Dr. Larry Kiage. Please find more information about the presentation and Dr. Kiage's bio on the next page.

Please come out, enjoy a bite to eat, the camaraderie, an interesting presentation and perhaps some discussion on the importance of accurate mineral characterization. Also, the differences that can exist between mineralogical, industrial and regulatory definitions for minerals.

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Atlanta Geological Society January Speaker

A 1,200-yr pollen Record of Environmental Changes in the Northern Gulf of Mexico

Speaker Dr. Larry Kiage's Bio:

Dr. Larry Kiage's research has focuses on studying the history of climate change, vegetational changes and human impacts on environments, during the late Holocene through the examination of multi-proxy records of fossil pollen, diatoms, phytoliths, fungal spores, and microscopic charcoal in lake-sediment cores. In addition to paleoenvironmental research he also studies recent environmental changes, especially land use and land cover changes using remote sensing techniques.

Dr Kiage employs the use of remote sensing techniques to study environmental impacts due natural and anthropogenic forces. He has used remote sensing tools to study hurricane impacts on the environment in Louisiana and Alabama along the Gulf of Mexico. He has also used remote sensing techniques to study eutrophication problems in Lake Maracaibo, Venezuela.

Dr. Kiage is actively involved in the new and exciting science of paleotempestology, which employs geological proxy techniques, including overwash sand layers, diatoms, foraminifera, pollen, and phytoliths collected from coastal lakes and marshes to study past hurricane activities. He is presently studying the spatial and temporal variations in hurricane activity in coastal Georgia at interannual to millennial timescales and their relationship to global and regional climate changes.

Abstract:

Palynological studies from the northern Gulf of Mexico (GOM) have primarily focused on understanding long-term climatic changes. Very few studies have addressed paleoenvironmental changes during the common era. This study used stratigraphic, loss-on-ignition, and pollen data from Bay Jimmy Island, to investigate environmental changes in coastal Louisiana since ca. 1090 AD. The results show that the wetland that presently defines Bay Jimmy area, LA, was formed at ca. AD 1090 and has mostly been influenced by prevailing climate conditions, including rare extreme events, and human activities. At least four major storms impacted the site during the past 1200 years, including two in recent times. The results indicate that the northern GOM was warm and dry during the Medieval Warm Period (ca. AD 950-1250). After AD 1090, the landscape was characterized by scattered vegetation communities of *Pinus* and *Juglans*, which were later succeeded by a closed forest that included *Quercus*, and *Morus*. The results show that red mangroves (*Rhizophora*) were established in the vicinity of Bay Jimmy until shortly after cal AD 1450 and 1640. The pollen record indicates that the LIA period (AD. 1550-1850) was cold and dry, and defined by an open vegetation community. The presence of *Ambrosia* and *Cheno/Am* pollen throughout the record suggests that anthropogenic influence has been part of the fabric of the southern Louisiana landscapes throughout the 1200 years.

Pyrite Oxidation Halts Migration of a Phosphorus Plume

Abstract

We have established a monitoring record of phosphate (PO_4^{3-}) migration in the Long Point, ON campground septic system plume that now spans 26 years. Previously, at year 16 (2006), a P plume 16 m in length was documented and provided a good fit with an analytical advection dispersion model when a P migration velocity of 0.8 m/yr was used (retardation factor of 37) and when P behaved in an otherwise conservative manner (sorption only). However, between years 16 and 26 (2016), the P plume length expanded by only 2 m (0.2 m/yr) and increased in depth by only 0.5 m. The zone of abrupt P depletion at depth occurs close to the zone where SO_4^{2-} concentrations increase in response to NO_3^- oxidation of pyrite. Scanning electron microscope images of sand grains from the nose of the P plume reveal abundant authigenic mineral coatings of considerable thickness (~ 5 to $20\ \mu\text{m}$), with Fe as the dominant cation and containing 1 to 3 wt % P. This evidence suggests that P is now being attenuated along a reaction front that coincides with the zone where pyrite oxidation is occurring. P migration may now be controlled by the rate of migration of the pyrite oxidation front and this is several times slower than the previously indicated rate in the shallower, sorption-controlled portion of the plume. Monitoring at Long Point has demonstrated the danger of embracing an overly simplistic conceptual model when attempting to predict wastewater P migration in groundwater and also highlights the unique insight provided by a long-term monitoring record.

Introduction

Recent cyanobacteria outbreaks in Lake Erie and elsewhere have heightened awareness of phosphorous (P) loading to lakes and streams (Jarvie et al. [2017](#)), including the potential for P loading from on-site sewage disposal in septic systems (Lemonds and McCray [2003](#)). Phosphate concentrations in domestic sewage (1.2 to 22 mg/L, McCray et al. [2005](#)) are more than 100 times higher than concentrations that can stimulate algae growth (Dillon and Rigler [1974](#)). The possibility that this phosphorus can remain mobile in subsurface flow systems is one of the considerations that may influence the choice of wastewater treatment options, particularly the use of on-site treatment systems vs. installation of centralized sewers. To help inform these decisions, watershed scale models are now being used to assist in quantifying potential P loads from septic systems relative to other P loads that may impact surface water bodies (Lemonds and McCray [2003](#); McCray et al. [2009](#); Geza et al. [2010](#); Sinclair et al. [2014](#)). These are usually GIS based models where the locations of individual septic systems can be assigned and solute travel times to receiving water bodies can be estimated from flow paths and groundwater velocities determined from water table maps, sediment characteristics, and so forth. Phosphorus migration in groundwater is normally highly retarded however, because of surface layer sorption reactions and retardation factors ranging from 10 to 100 have been reported for P migration in septic system plumes (Robertson et al. [1998](#); Parkhurst et al. [2003](#)). Phosphorus sorption isotherms for various sediment types are available from the literature (McCray et al. [2005](#)) and modeling studies have tended to focus on sorption as the dominant factor controlling P mobility (Lemonds and McCray [2003](#); Geza et al. [2010](#); Domagalski and Johnson [2011](#); Sinclair et al. [2014](#)). Sorption reactions are normally fast and reversible however and consequently, sorbed phosphorus remains mobile in the subsurface and over time, will ultimately discharge to a surface water body. A few studies have attempted to include other potential P attenuation mechanisms such as slower “irreversible” sorption (Spiteri et al. [2007](#)) and direct mineral precipitation reactions (Parkhurst et al. [2003](#)), but so far, these processes have been included in plume scale models only.

Surprisingly, only a few studies have actually documented P loading to surface water bodies from septic system plumes. Corbet et al. ([2002](#)) conducted plume studies near three septic systems located along the shore of Apalachicola Bay, FL, and estimated P fluxes into the bay ranging from 1 to 24 mg/m²/yr. A small kettle lake on

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)

Cape Cod, MA, Ashumet Pond, is estimated to have received P loads of approximately 400 kg/yr in recent years, from infiltration beds located approximately 600 m from the pond that received wastewater from a military base during 1936 to 1995. This septic system plume has been studied extensively (LeBlanc [1984](#)) and in recent years has discharged PO_4^{3-} -rich groundwater (1 to 2 mg/L P) along a 200 m reach of the pond shoreline. A 3D reactive transport model was used to gain insight into the processes controlling P migration (Parkhurst et al. [2003](#)).

Phosphorus concentrations and loads discharging to the pond in the 1990s were found to be only about one third the values expected if sorption was the only process affecting P mobility. However, if precipitation of the Fe-P mineral vivianite, which attains saturation in portions of the plume, was also considered, then simulated values were closer to the measured field values. Furthermore, desorption tests on sediment samples from the core of the plume, revealed that only 10 to 50% of the P was readily desorbable, with the remainder presumably sequestered in insoluble mineral forms.

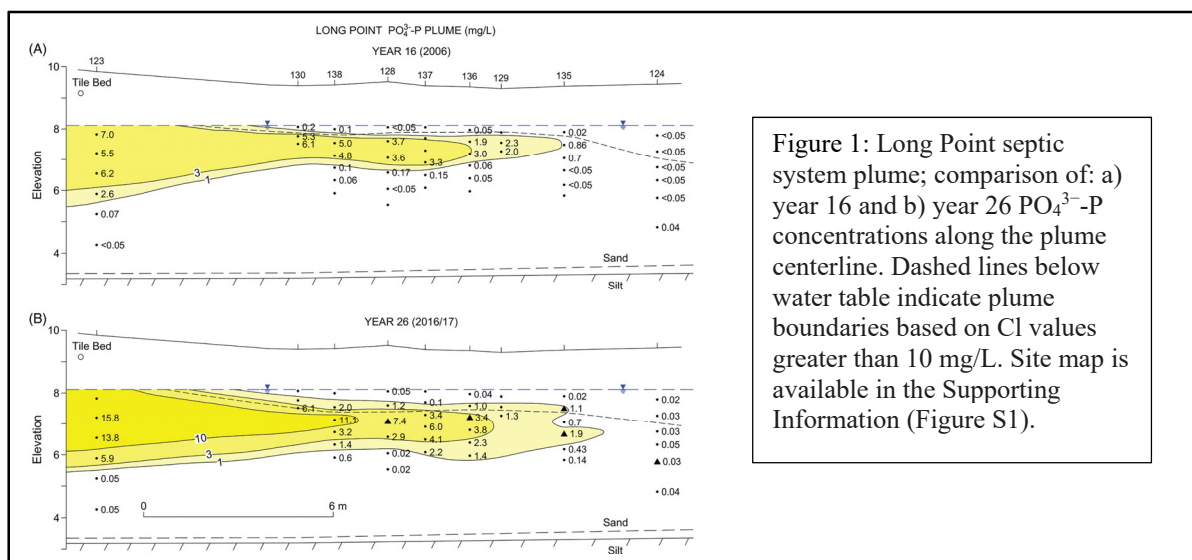
In the Cape Cod study discussed above, mineral precipitation played an important role and served to decrease P loading to well below that predicted by sorption alone, yet watershed scale models remain primarily focused on sorption. If mineral precipitation reactions are occurring, sorption focused models will overestimate P loading rates. Long-term monitoring of P plume behavior is crucial in revealing the importance of these various processes. However, in addition to the Cape Cod study, only a few other field studies are available with extended monitoring records and this paint a contrasting picture of P mobility. Monitoring of the Long Point and Cambridge septic system plumes over periods of several years indicated that P was retarded by sorption, but was still migrating at a rate of approximately 1 m/yr (Robertson [1995](#); Robertson [2008](#)), whereas at the Parry Sound site, monitoring over a 5-year period revealed that the P plume was highly attenuated by mineral precipitation reactions (>90% P removal) and was virtually stagnant (Robertson [2012](#)). Clearly, there is a need for addition field studies and particularly, long-term observational records that bring new insight into the factors controlling the ultimate fate of PO_4^{3-} in wastewater plumes.

In this study we present the results of recent monitoring undertaken at the Long Point, ON septic system site in 2016/17, which now represents 26 years of plume development. These new results paint a decidedly different picture of P mobility compared to that presented 10 years previously, one that is much slower and where a sorption-only model no longer applies.

Long Point Site

Long Point is a large seasonal-use campground located on the north shore of Lake Erie and here wastewater is treated on-site in a septic system. We have had the opportunity to have maintained a semi-continuous monitoring record of Tile Bed 2 at this site since it was commissioned in 1990. At that time a network of six multilevel bundles was installed beneath the tile bed and over time, the network was expanded to ultimately include approximately 250 monitoring points in 33 multilevel bundles that traced the plume 200 m down gradient within a 5 m thick sand aquifer, where it then discharged to L. Erie (Robertson et al. [2013](#)). Tile Bed 2 (Figures [1](#) and [S1](#)), has received high wastewater loads of up to 10 cm/day during peak loading in the summer (Robertson and Cherry [1992](#)). Water limiting fixtures installed in the washroom in approximately 2004, resulted in relatively high concentrations of nitrogen in the wastewater after that date (~100 mg/L, Table [S1](#)), which is about twice the average value for septic tank effluent (McCray et al. [2005](#)). For most of the operational period, concentrations of PO_4^{3-} -P in the effluent (5 to 7 mg/L, Table [S1](#)) have been in the range considered typical for septic tank effluent (McCray et al. [2005](#)). However, in 2014 the system was upgraded with the installation of a tertiary aeration module (Bionest™) and since that time PO_4^{3-} -P concentrations in the effluent have approximately doubled to 15 mg/L and nitrogen has been substantially nitrified (Table [S1](#)).

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)



The plume core zone of interest in this study normally exhibits minimal dilution compared to the septic tank effluent. Although not measured in the current study, previous studies have consistently shown that the plume is suboxic ($\text{DO} < 1 \text{ mg/L}$, Robertson et al. 2012), has slightly acidic to neutral pH (6.7 to 7.0), low DOC ($\sim 3 \text{ mg/L}$) and variable Eh in the range of +290 to +400 mV (Robertson et al. 1998; Robertson 2008). Plume ages near the tile bed were estimated previously from NaBr tracer tests carried out in 1990 and again in 2008 and by observing seasonal variations in plume electrical conductivity (Figure S2). Seasonally, there is a strong vertical component of flow downward below the tile bed (up to $\sim 10 \text{ cm/day}$, Robertson et al. 2012) as a result of the high sewage loading rate in the summer.

Plume monitoring has focused on a variety of parameters, including PO_4^{3-} (e.g., Robertson et al. 2013). In an earlier study (Robertson 2008) the mobility of PO_4^{3-} over the initial 16-year period of operation, from 1990 to 2006, was assessed. During that time a distinct PO_4^{3-} plume 16 m in length developed. $\text{PO}_4^{3-}\text{-P}$ concentrations in the proximal zone approached concentrations in the septic tank effluent ($\sim 6 \text{ mg/L}$) and concentrations along the entire length of the P plume were well matched using an analytical advection dispersion model when a P retardation factor of 37 was used in conjunction with the field-indicated groundwater velocity of 28 m/yr and when no P “decay” (mineral precipitation or irreversible sorption) occurred. Thus, a sorption-only conceptual model was successful in simulating P plume development at that time.

Methods

Plume Sampling and Analyses

Site characterization and groundwater sampling methods have been described previously (Robertson and Cherry 1992; Aravena and Robertson 1998; Robertson 2008). In the current study, renewed sampling of the PO_4^{3-} plume was undertaken during November 2015 to January 2017 (eight sampling events). Groundwater samples were collected using a peristaltic pump with silicone tubing and were filtered ($0.45 \mu\text{m}$) in-line, prior to atmospheric exposure. Samples for NH_4^+ and PO_4^{3-} analyses were acidified to $\text{pH} < 2$ immediately after collection, whereas samples for Cl^- , SO_4^{2-} , and NO_3^- were left untreated. All samples were immediately transported to the laboratory where they were either refrigerated at approximately 4°C or frozen (PO_4^{3-} and NH_4^+ samples) until analysis. Nitrate, Cl^- , and SO_4^{2-} were analyzed in the Environmental Geochemistry Laboratory (EGL), Department of Earth

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)

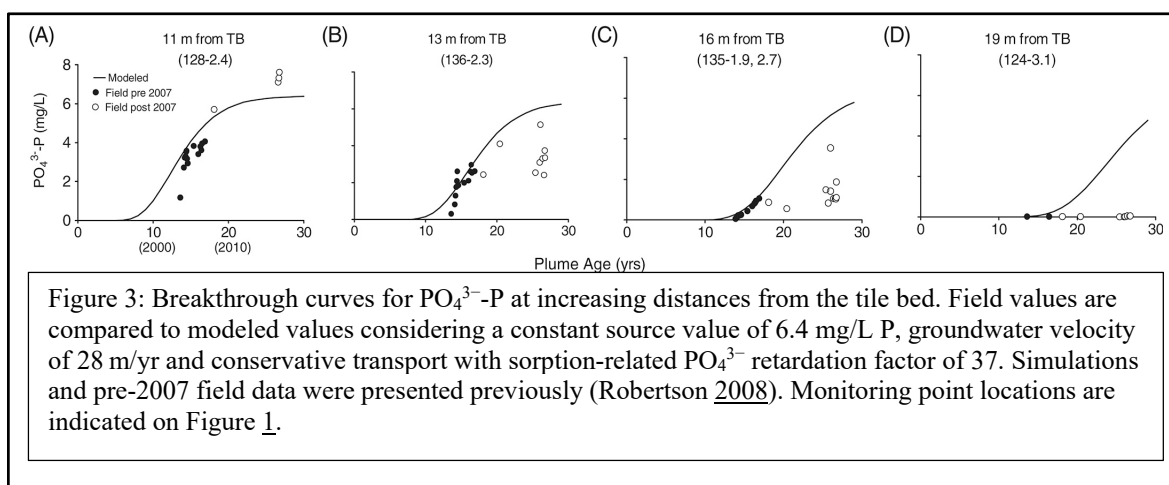
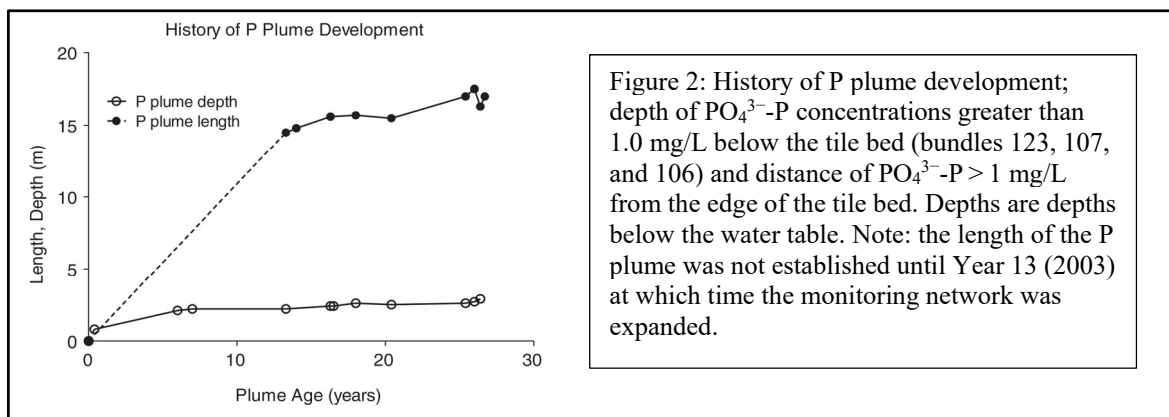
and Environmental Sciences, University of Waterloo, ON, by ion chromatography using a Dionex ICS-90 (Dionex, Sunnyvale, California), which provided an analytical reporting limit of lesser than 0.5 mg/L for each of these parameters. NH_4^+ was also analyzed at EGL, using a colorimetric technique with a Beckman DU 530 spectrophotometer, which provided a detection limit of lesser than 0.05 mg/L N. Soluble reactive phosphorus (PO_4^{3-}) was analyzed colorimetrically (molybdonate blue technique, APHA 1998) using a spectrophotometer (Cary100 UV-Vis, Agilent Technologies, Santa Clara, California), which provided a detection limit of 0.5 $\mu\text{g/L}$ P. Sediment samples from the P plume zone were obtained using 5 cm diameter aluminum core tubes that were advanced into the subsurface by percussion. Morphology and composition of sand grains from the cores were determined using a scanning electron microscope with energy dispersive X-ray (EDX) elemental analysis. These analyses were conducted using a Leo 440 scanning electron microscope with Quartz XOne EDX system (Carl Zeiss Microscopy, Oberkochen, Germany) at Surface Science Western, London, ON.

Results and Discussion

P Mobility Trends

Figure 1 shows the 18 m long zone in the plume where elevated PO_4^{3-} concentrations are present. Groundwater ages in this zone, estimated from previous tracer tests and seasonal variations in EC, range from about 0.2 years 8 m from the tile bed edge (well 138), to 0.7 years 19 m from the tile bed (well 124, Figure S2). One of the previous tracer tests occurred over an 8-month period from July 2008 to April 2009 (Robertson et al. 2012) and thus would have factored in any transience of the flow system resulting from variations in wastewater loading or from seasonal fluctuations of the water table. Two PO_4^{3-} snapshots are shown representing Year 16 (2006) and year 26 (2016/17). Several trends are noted. PO_4^{3-} -P concentrations in the proximal zone, within 10 m of the tile bed, were in the range of approximately 5 to 7 mg/L in Year 16 but then increased to 11 to 16 mg/L in Year 26. These values are very close to the effluent values for those periods (Table S1). This indicates that there is little P loss in the vadose zone here, in contrast to some other sites where substantial vadose zone P loss has been noted (Robertson et al. 1998). Secondly, the size of the P plume has only increased marginally over the last 10-year period. In Year 16, the plume length, as defined by concentrations greater than 1 mg/L, was 16 m, but then this increased by only about 2 m to 18 m by year 26. Likewise, the P plume depth below the tile bed remained almost fixed at about 4 m depth over this period which is well above the bottom of the overall plume. Figure 2 chronicles the expansion of the P plume length and depth over the 26-year period of operation. An abrupt decrease in P mobility is apparent after about year 16. In Figure 3, PO_4^{3-} temporal trends are shown for several monitoring points at increasing distances from the tile bed. These data are compared to modeled breakthrough curves that were presented previously at Year 16 (Figure 3). At that time, the modeled base case provided a good match with the field data when utilizing the field-indicated groundwater velocity of 28 m/yr, a PO_4 retardation factor of 37 (sorption effect), a source concentration of 6.4 mg/L P, longitudinal dispersivity of 0.5 m and when PO_4 behaved in an otherwise conservative manner (no “degradation”). Additional simulations show the effects expected if first order degradation reactions, with half lives of 5 and 30 yrs., were also occurring. The degradation scenerios did not match the field data at that time (Figure 3). However, recent monitoring has now revealed a distinctly different pattern of P mobility. Whereas PO_4^{3-} values within 11 m of the tile bed have responded in an apparently conservative manner, farther away values become progressively less than previously predicted. At nest 124, 19 m from the tile bed, PO_4^{3-} -P values were barely above detection (~ 0.05 mg/L) in 2016/17, even though much higher values of approximately 4 mg/L had been predicted. A comparison of PO_4^{3-} concentrations along the plume centreline in Years 16 and 26 (Figure 4) further illustrates the divergent behavior of the plume in recent years. At year 16, the field data were well matched using the conservative, sorption-only approach, but by year 26 this was not the case. In Year 26, PO_4^{3-} was abruptly attenuated at the nose of the plume, along what appears to be a reaction front. The modeled simulations shown in Figure 4 used the same transport parameters for the year 26 simulations that were used previously for year 16

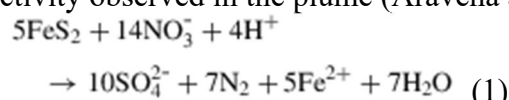
Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)



(groundwater velocity of 28 m/yr, P retardation factor of 37 and longitudinal dispersivity of 0.5 m). However, it should be noted that after water limiting fixtures were introduced in the washroom in 2004 (Year 14), it is estimated that the tile bed loading rate decreased by about half, which likely resulted in decreased flow velocities near the tile bed. However, the velocity decrease was likely less than a factor of two, because the flow system is driven by precipitation recharge as well, particularly farther away from the tile bed. It appears unlikely that a potential modest decrease in groundwater velocity could account for the abrupt attenuation of P noted at the nose of the plume. Furthermore, aquifer dispersivity typically increases at greater travel distances (Sudicky 1986) and at the nose of the plume, this would tend to counterbalance the effect of any velocity decrease. Possible changes in redox and pH conditions that might have occurred as a result of enhanced oxidation of the wastewater after 2014, could also potentially affect P mobility. However, it is unlikely that any such changes would preferentially affect the distal portion of the P plume while leaving the proximal zone unaffected.

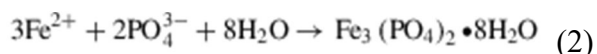
Pyrite Oxidation Front

Trace amounts of pyrite are present in the Long Point aquifer (0.02 wt %, Robertson and Cherry 1992) and this contributes to robust denitrification activity observed in the plume (Aravena and Robertson 1998);



Ferrous iron in solution can attenuate PO_4^{3-} by the precipitation of vivianite;

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)



or, more likely, if some NO_3^- remains, ferrous iron will be further oxidized and Fe(III)-P minerals such as strengite may precipitate;

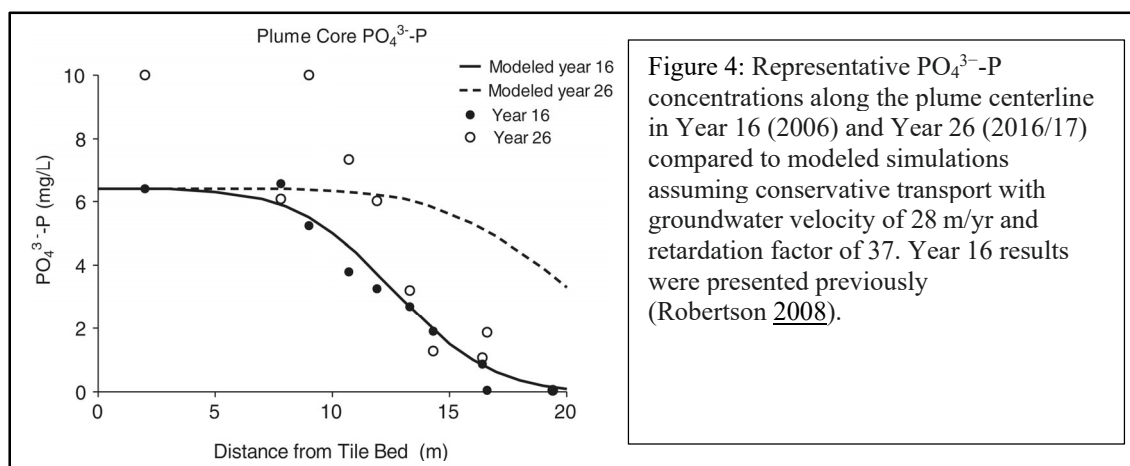
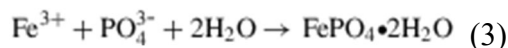


Figure 4: Representative $\text{PO}_4^{3-}\text{-P}$ concentrations along the plume centerline in Year 16 (2006) and Year 26 (2016/17) compared to modeled simulations assuming conservative transport with groundwater velocity of 28 m/yr and retardation factor of 37. Year 16 results were presented previously (Robertson [2008](#)).

Pyrite oxidation is revealed here and elsewhere by distinct increases in SO_4^{2-} concentrations (Korom [1992](#); Aravena and Robertson [1998](#); Bohlke et al. [2002](#)). Figure [5](#) compares depth profiles of $\text{PO}_4^{3-}\text{-P}$, NO_3^- -N and SO_4^{2-} below the tile bed in years 4 to 6 (1994 to 1996) and year 25 (2015). In both cases NO_3^- -N concentrations decrease abruptly beginning at about 3 to 4 m depth and SO_4^{2-} concentrations increase coincidentally. Previous mass balance estimates based on the amount of SO_4^{2-} increase, indicated that pyrite oxidation was responsible for 22 to 30% of the observed NO_3^- loss at Long Point, with the remainder attributed to denitrification utilizing trace amounts of organic carbon that is also present in the aquifer (Robertson and Cherry [1992](#); Aravena and Robertson [1998](#)). The elevated SO_4^{2-} also has an isotopic signature that is consistent with biogenic pyrite oxidation (Aravena and Robertson [1998](#)). Of note, the PO_4^{3-} plume terminates at the same depth where SO_4^{2-} begins to increase (4 m depth, Figure [5](#)). This provides strong circumstantial evidence that pyrite oxidation could be playing a role in PO_4^{3-} immobilization at Long Point.

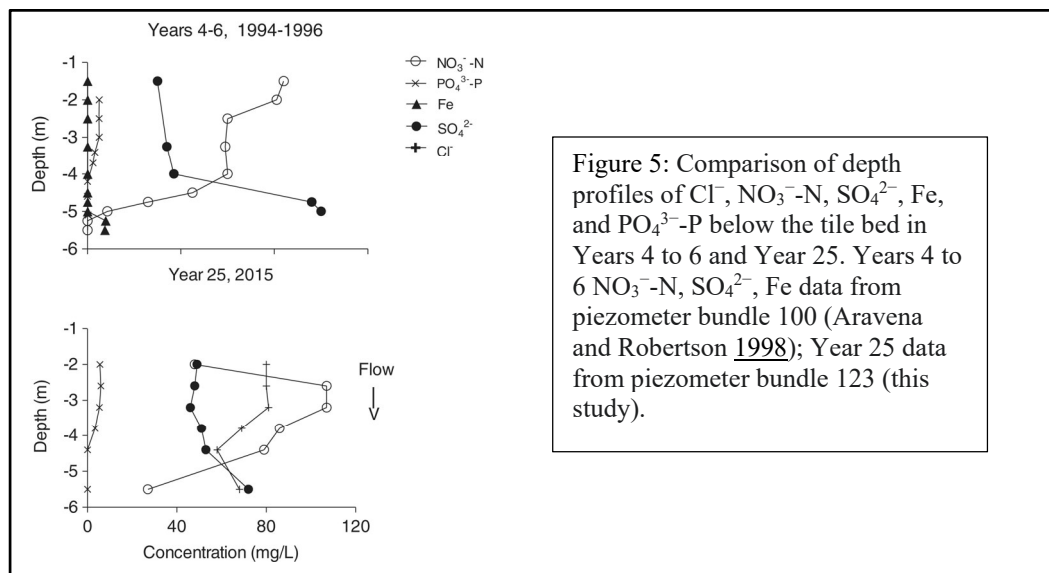


Figure 5: Comparison of depth profiles of Cl^- , NO_3^- -N, SO_4^{2-} , Fe, and $\text{PO}_4^{3-}\text{-P}$ below the tile bed in Years 4 to 6 and Year 25. Years 4 to 6 NO_3^- -N, SO_4^{2-} , Fe data from piezometer bundle 100 (Aravena and Robertson [1998](#)); Year 25 data from piezometer bundle 123 (this study).

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)

Authigenic Mineral Coatings

Scanning electron microscope images provide additional evidence that mineral precipitation reactions likely play an important role in immobilizing PO_4^{3-} at Long Point. Figure 6 shows an electron image and X-Ray elemental composition of a dolomite grain from the nose of the P plume which has a distinct authigenic mineral coating of substantial thickness (5 to 20 μm). The coating is variable in composition, with Fe (15.6 to 29.5 wt %), Ca (1.9 to 22.6 wt %), and Al (1.9 to 11.3 wt %) as the dominant cations and containing 1.0 to 1.4 wt % P (Figure 6). Similar coatings were observed on other sand grains of varying mineralogy (quartz, feldspar, magnetite, not shown) and these also contained 1 to 3 wt % P. The morphology and composition of these coatings is generally consistent with that expected from pyrite oxidation and subsequent Fe(III)-P mineral precipitation, as indicated in Equations 1 and 3 above. Additionally, geochemical modeling undertaken previously (Robertson 2008), indicated that the P plume core zone was consistently supersaturated with respect to the Ca-P mineral hydroxyapatite (saturation index 7.4 to 8.1) and the proximal zone was supersaturated with both of the Fe-P minerals vivianite (SI 1.3) and strengite (SI 2.7 to 5.0). Thus, several minerals could potentially be contributing to the formation of these coatings, although their variable composition suggests these are likely amorphous precursors rather than distinct mineral phases.

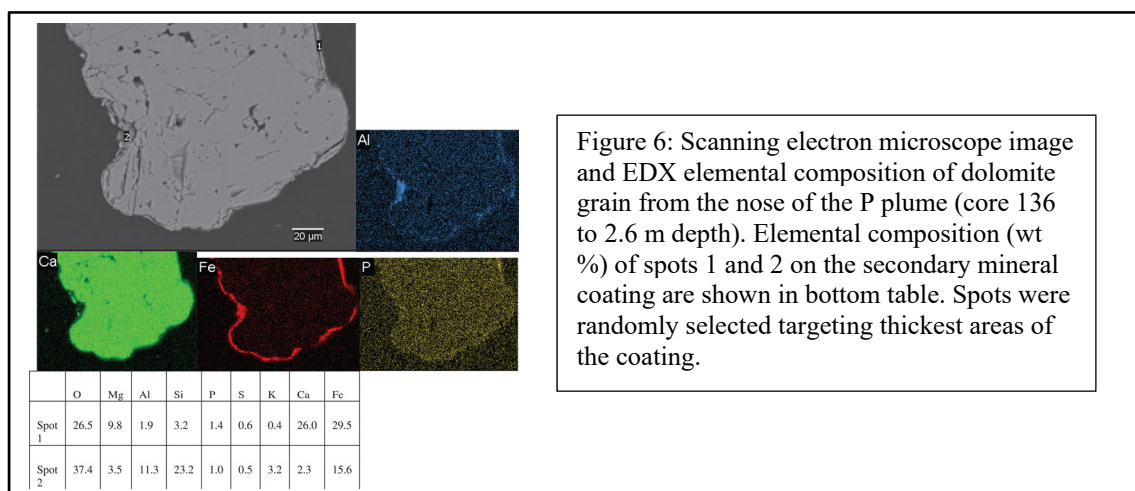


Figure 6: Scanning electron microscope image and EDX elemental composition of dolomite grain from the nose of the P plume (core 136 to 2.6 m depth). Elemental composition (wt %) of spots 1 and 2 on the secondary mineral coating are shown in bottom table. Spots were randomly selected targeting thickest areas of the coating.

Long-Term PO_4^{3-} Migration Rate

Although recent monitoring has indicated that pyrite oxidation may now be limiting P migration, this effect was apparently less so during earlier plume development in the shallower portion of the aquifer (i.e., at Year 16). This may indicate that the upper portion of the aquifer has a lower pyrite content than deeper zones. This could be the result of past oxidation in the shallow zone associated with natural precipitation recharge or from oxidation that may have occurred during past low water table events prior to construction of the tile bed. Note that as the plume moves farther away from the tile bed, it migrates progressively deeper into the aquifer (Robertson et al. 2013) in response to ongoing recharge from precipitation. In this region, precipitation (~ 90 cm/yr) exceeds potential evapotranspiration (~ 60 cm/yr, Morton 1983) by about 30 cm/yr. Assuming the aquifer has porosity of 0.35, this amount of potential water table recharge would result in a vertical component of flow, in the shallow water table zone, of about 1 m/yr. Thus, the plume should be migrating progressively deeper into the aquifer as it migrates away from the tile bed area where potentially, it could be encountering zones of higher pyrite content at depth.

If pyrite oxidation is now controlling PO_4^{3-} mobility, then the long-term rate of PO_4^{3-} migration will likely coincide with the rate at which pyrite is being consumed by denitrification activity. If one considers the S content of the aquifer (0.02 wt %), the vertical component of flow at approximately 4 m depth below the tile bed (~ 15 m/yr from Figure S2), NO_3^- -N loss at 3 to 5 m depth of approximately 50 mg/L, from Figure 5 and assuming that 25% of this

Pyrite Oxidation Halts Migration of a Phosphorus Plume (continued)

loss (13 mg/L) is due to pyrite oxidation, then the pyrite oxidation front should be moving downward at a rate of approximately 30 cm/yr. Sulfate profiles shown on Figure 5 indicate that between Years 4 to 6 and Year 25, the pyrite oxidation front may have moved deeper by only about 1 to 2 m, indicating a somewhat slower downward migration rate of 5 to 10 cm/yr. However, the vertical groundwater velocity inferred from Figure S1 is representative of a period of active sewage loading and this only occurs for about 4 months during the year. Vertical flow rates are much less for the remainder of the year and consequently, migration rate of the pyrite oxidation front appears generally consistent with the rate expected.

Conclusions and Implications

Nitrate attenuation by pyrite oxidation is a commonly observed phenomena in Canada (Robertson et al. 1996; Sebol et al. 2007) and elsewhere (Postma et al. 1991; Korom 1992; Bohlke et al. 2002). It appears likely that similarly robust P immobilization will occur in other nitrate rich septic system plumes when pyrite is encountered and ferrous iron is liberated (Equations 1 to 3). This attenuation mechanism may have particular relevance for watershed scale modeling studies with longer flow paths and longer time scales involved. For example, in a study of a *L. Huron* beach town with a large number of decommissioned septic systems present, Roy et al. (2017) found evidence of legacy P plumes present near two of four decommissioned tile beds studied in detail. Except for these two locations however, which were close to the beach, there was a absence of obvious septic system P loading at other locations along the beach. This was attributed to the slow rate of migration of the P plumes (estimated at 1 to 5 m/yr, considering groundwater flow of 100 m/yr and a P retardation range of 20 to 100), such that the P plumes from decommissioned systems located farther from the beach had not yet reach the shoreline. Under this scenario, P loading to the beach would continue to increase in coming decades and centuries. However, Roy et al. also speculated that the lack of septic P impact could reflect precipitation of vivianite, as the deeper groundwater at this location is anaerobic with elevated Fe present. Experience at Long Point and at other sites such as the Cape Cod, suggests that the potential for mineral precipitation reactions should also receive an appropriate degree of attention when attempting to predict the long-term fate of P in septic system plumes.

Long Point now adds to the growing list of sites where the long-term mobility of P in septic system plumes is dominated by mineral precipitation rather than by sorption reactions. In some cases on-site treatment in septic systems may be the preferred choice for wastewater disposal because they can potentially provide treatment at lower cost and with lower energy consumption compared to the installation of centralized sewers. To successfully predict P loading from septic systems, watershed scale models need to also consider the potential for mineral precipitation reactions and not just focus on sorption. The Long Point site also highlights the unique insight that can be gained from maintaining a long-term monitoring record of a contaminant plume.

Read more about this article at: <https://ngwa.onlinelibrary.wiley.com/doi/10.1111/gwat.12877>

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane

Abstract

Vapor intrusion (VI) involves migration of volatile contaminants from subsurface through unsaturated soil into overlying buildings. In 2015, the US EPA recommended an approach for screening VI risks associated with gasoline releases from underground storage tank (UST) sites. Additional assessment of the VI risk from petroleum hydrocarbons was deemed unnecessary for buildings separated from vapor sources by more than recommended vertical screening distances. However, these vertical screening distances did not apply to potential VI risks associated with releases of former leaded gasoline containing 1,2-dichloroethane (1,2-DCA), because of a lack of

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empirical data on the attenuation of 1,2-DCA in soil gas. This study empirically evaluated 144 paired measurements of 1,2-DCA concentrations in soil gas and groundwater collected at 47 petroleum UST sites combined with BioVapor modeling. This included (1) assessing the frequency of 1,2-DCA detections in soil gas below 10^{-6} risk-based screening levels at different vertical separation distances and (2) comparing the US EPA recommended vertical screening distances with those predicted by BioVapor modeling. Vertical screening distances were predicted for different soil types using aerobic biodegradation rate constants estimated from the measured soil-gas data combined with conservative estimates of source concentrations. The modeling indicates that the vertical screening distance of 6 feet (1.8 m) recommended for dissolved-phase sources is applicable for 1,2-DCA below certain threshold concentrations in groundwater, while 15 feet (4.6 m) recommended for light nonaqueous phase liquid (LNAPL) sources is applicable for sites with clay and loam soils in the vadose zone, but not sand, if 1,2-DCA concentrations in groundwater exceed $150 \mu\text{g/L}$. This dependence of the predicted vertical screening distances on soil type places added emphasis on proper soil characterization for VI screening at sites with 1,2-DCA sources. The soil-gas data suggests that a vertical screening distance of 15 feet (4.6 m) is necessary for both dissolved-phase and LNAPL sources.

Introduction

A combination of 1,2-dichloroethane (1,2-DCA, also called ethylene dichloride) and 1,2-dibromoethane, also called ethylene dibromide (EDB) was added to leaded gasoline as lead scavengers to prevent lead oxide deposits from fouling the internal combustion engines of automobiles (Falta et al. 2005). Use of lead scavengers in automobile gasoline started in the mid-1920s and continued until it was phased out beginning in the late 1980s (Falta 2004; US EPA 2006). Leaded automobile gasoline contained 1,2-DCA and EDB at concentrations in the range of several hundreds of mg/L (Falta et al. 2005).

Concentrations of 1,2-DCA and EDB in groundwater at former leaded gasoline release sites could reach as high as approximately 3,700 and 1,900 $\mu\text{g/L}$, respectively, based on equilibrium partitioning with leaded gasoline (Falta 2004). In 2010, U.S. states were prompted by the US EPA to monitor for lead scavengers in groundwater at sites where leaded fuels were either stored or marketed prior to 1986 and where there was a suspected threat to drinking water resources. A nationwide survey of state leaking underground storage tank (UST) programs in the United States identified lead scavengers as a contaminant of concern in groundwater at up to 20% of petroleum UST sites (ASTSWMO 2014).

In 2015, the US EPA Office of Underground Storage Tanks (OUST) published a technical guide (US EPA 2015a) recommending the use of vertical screening distances for assessing petroleum vapor intrusion (PVI) risks at leaking UST sites. No further PVI assessment was required at UST sites if the vertical separation distance between a dissolved-phase or light nonaqueous phase liquid (LNAPL) vapor source and an overlying building was ≥ 6 feet (1.8 m) or 15 feet (4.6 m), respectively. These distances were based on the vertical screening distances derived for benzene, which required the greatest source-building vertical separation distance for concentrations in soil gas to fall below risk-based screening levels of concern for PVI. However, no data were available to estimate vertical screening distances for lead scavengers. Hence, the US EPA guidance recommended the collection of soil-gas data at UST sites with gasoline releases with lead scavengers and comparison of measured concentrations to vapor intrusion screening levels (VISLs) available at: <https://www.epa.gov/vaporintrusion/vapor-intrusion-screening-level-calculator> (US EPA 2014). The VISLs are based on the OSWER Technical Guide for Assessing and Mitigating the Vapor Intrusion Pathway from Subsurface Vapor Sources to Indoor Air (US EPA 2015b). The VISLs for 1,2-DCA and EDB in indoor air, soil gas, and groundwater are presented in Table 1.

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Table 1

Comparison of Vapor Intrusion Screening Level (VISL) Values in Indoor Air, Soil Gas, and Groundwater at the 10^{-6} Risk Level for 1,2-DCA and EDB

Compound	Indoor Air ($\mu\text{g}/\text{m}^3$)	Soil Gas ($\mu\text{g}/\text{m}^3$)	Groundwater ($\mu\text{g}/\text{L}$)
1,2-Dichloroethane (1,2-DCA)	0.11	3.6	2.2
1,2-Dibromoethane (EDB)	0.0047	0.16	0.18

To help improve the understanding of vapor intrusion risks from lead scavengers, Ma et al. (2016) used a 3-D numerical model to validate the applicability of the vertical screening distances recommended by US EPA OUST for 1,2-DCA and EDB. The modeling was used to predict the potential for vapor intrusion assuming a range of first order rate constants reported in the literature (for aerobic/anaerobic biodegradation and chemical degradation), different soil types (sand, silt, and clay), and measured groundwater concentrations reported in

Falta et al. (2005). Model results indicated that a vertical source/building separation distance of 15 feet (4.6 m) was generally sufficient to screen sites for EDB, but not for 1,2-DCA if the concentrations in groundwater exceeded approximately 3 and 30 $\mu\text{g}/\text{L}$, along with sand and silt respectively, in the vadose zone. Their study noted the importance of developing an empirical dataset to evaluate the potential for vapor intrusion of 1,2-DCA and EDB.

The goal of this study was to fill this data gap by assessing the applicability of the US EPA recommended vertical screening distances for 1,2-DCA and EDB through an evaluation of paired soil gas and groundwater concentration data collected at 47 UST sites with likely releases of leaded gasoline (based on UST operational history).

Unfortunately, a meaningful evaluation of vertical screening distances for EDB could not be undertaken because soil-gas concentrations were all nondetect at reporting limits ($2 \mu\text{g}/\text{m}^3$) that exceeded the soil-gas VISL for EDB at the 10^{-6} risk level ($0.16 \mu\text{g}/\text{m}^3$). The data for 1,2-DCA were evaluated by two complementary methods. The first method involved empirical evaluation of the percentage frequency of 1,2-DCA detections in soil gas below risk-based screening levels at different vertical separation distances above dissolved-phase and LNAPL sources. This was analogous to the method used to derive vertical screening distances for petroleum hydrocarbons (Lahvis et al. 2013; USEPA 2013). The second method involved (1) calibration of first-order aerobic biodegradation rate constants from soil-gas concentration data using the BioVapor model (DeVaul 2007; API 2010) associated with the known soil types, and measured concentrations of 1,2-DCA in groundwater and (2) prediction of vertical screening distances using these rate constants in combination with the 95th percentile value of the 1,2-DCA concentrations in groundwater in this study as the source concentrations.

Figure 1 compares the cumulative frequency distribution of 1,2-DCA detections in groundwater in this study ($N = 114$, lowest detection limit of $0.5 \mu\text{g}/\text{L}$) with the maximum site concentration reported between 2016 and 2018 from 471 petroleum UST sites in the California Geotracker database. These two distributions of 1,2-DCA concentrations in groundwater are very similar (median concentrations of 12 vs. 9 $\mu\text{g}/\text{L}$ and 95th percentile concentrations of 360 vs. 474 $\mu\text{g}/\text{L}$, respectively for this study and UST sites in California). This suggested that conclusions of this study will be applicable to a broader population of petroleum UST sites. Figure 1 also indicated that concentrations of 1,2-DCA in groundwater exceed the groundwater VISL at the 10^{-6} risk level ($2.2 \mu\text{g}/\text{L}$) at approximately 80% of petroleum UST sites with 1,2-DCA detections in groundwater. Understanding the applicability of the US EPA recommended vertical screening distances to assess the potential vapor intrusion risk from 1,2-DCA will be extremely beneficial in optimizing future site investigations at such sites.

Study Approach

Empirical Evaluation of Applicability of US EPA Recommended Vertical Screening Distances for 1,2-DCA

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

Data from UST sites were compiled from three main sources: (1) California (GeoTracker database—<http://geotracker.waterboards.ca.gov/>), (2) site data from other states across the United States and provinces in Canada, and (3) sites in Minnesota (MN) as facilitated by Minnesota Pollution Control Agency. The following criteria were used to filter data for the empirical evaluation.

1. Data from locations suggesting presence of LNAPL sources in the vadose zone soils were eliminated. These were identified by reviewing the associated soil boring logs for presence of elevated soil PID readings (e.g., more than 100 ppm) as well as observations of staining and odor due to hydrocarbons (Lahvis et al. 2013; ITRC 2014). In addition, soil-gas sampling locations within 20 feet (6.1 m) of UST, dispenser island or piping were excluded.
2. Data were excluded if the soil-gas sampling location and the paired groundwater well were separated laterally by more than 30 feet (9.1 m). In addition, data were excluded if the groundwater and soil-gas sampling dates were more than 90 days apart.
3. Site groundwater elevation maps were reviewed to ensure that the resulting pairs of soil gas and groundwater data represented site conditions where 1,2-DCA in soil gas were sourced from either a dissolved-phase or LNAPL source at the water table.
4. Only data collected using appropriate soil-gas sampling techniques and analytical protocols were used (California EPA DTSC 2015; NJDEP 2018). The data met all the following criteria:
 - a. samples collected from properly constructed permanent soil-gas probes installed using either a hollow stem auger or a direct push tool, typically with a bentonite slurry seal in the annulus between the tubing and borehole wall above the screened interval;
 - b. samples collected using Nylon® or Teflon® tubing into laboratory certified Summa® canisters at recommended sampling flow rates (less than 200 mL/min);
 - c. samples collected using appropriate leak detection protocols using either helium, 1,1-difluoroethane or isopropanol as a tracer compound;
 - d. samples analyzed using either EPA method TO-15, or 8260B.

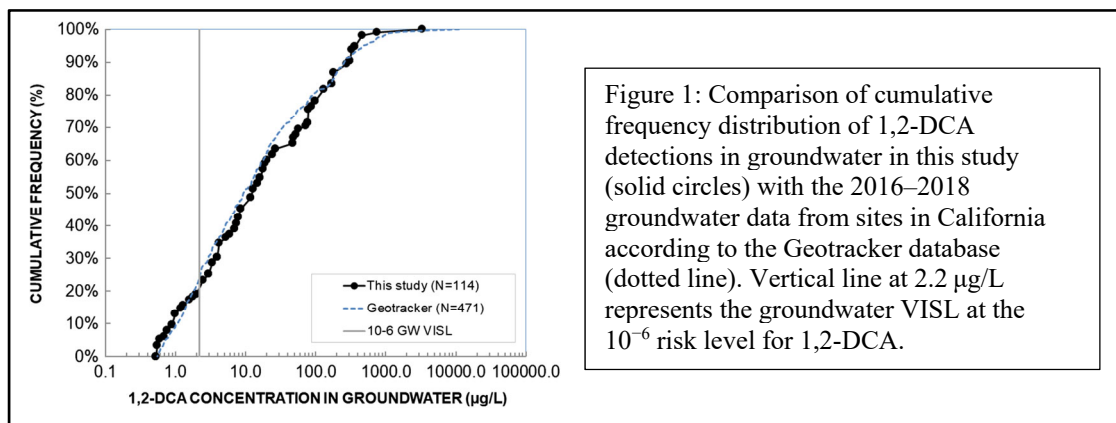


Figure 1: Comparison of cumulative frequency distribution of 1,2-DCA detections in groundwater in this study (solid circles) with the 2016–2018 groundwater data from sites in California according to the Geotracker database (dotted line). Vertical line at 2.2 µg/L represents the groundwater VISL at the 10^{-6} risk level for 1,2-DCA.

This data filtering resulted in 200 pairs of soil gas and groundwater data that were suitable for empirical evaluation. The distribution of sites across states, number of data pairs and the range of 1,2-DCA concentrations in soil gas and groundwater are summarized in Table 2. Only data with concentration of 1,2-DCA in groundwater (or reporting limit) greater than 2.2 µg/L (groundwater VISL at the 10^{-6} risk level) were used for empirical evaluation or BioVapor modeling. This reduced the total number of data pairs to 144. These represented 55 monitoring wells along with the adjacent soil vapor probes (some installed at multiple depths) that were sampled

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

between 2004 and 2017. The predominant vadose zone soil types observed at these locations included—sand (28 locations), loam/silt (16 locations), sandy clay (2 locations), silty clay (8 locations), and clay (1 location).

Table 2				
Summary of Soil Gas and Groundwater Concentration Data Pairs for 1,2-DCA (N = 200) that Satisfy Data Filtering Criteria for the Purpose of Empirical Assessment of Vertical Screening Distance				
State/Province	Number of Sites	Number of Concurrent Soil Gas and Groundwater Concentration Data	Range of Soil-Gas Concentrations of 1,2-DCA ($\mu\text{g}/\text{m}^3$)	Range of 1,2-DCA Concentrations in Groundwater ($\mu\text{g}/\text{L}$)
California	32	126	<0.53 to <4400	<0.5 to 3300
Western Canada ¹	5	31	1 to 120	1 to 360
Washington DC and certain states in the US ²	8	38	<0.3 to 510	0.5 to 13
Minnesota	2	5	<0.81 to <1600	<1 to 8.6
TOTAL	47	200		

¹Includes British Columbia, Alberta, and Manitoba.

²Includes North Carolina, Michigan, Alaska, South Carolina, and Pennsylvania.

Criteria to Distinguish Dissolved Sources from LNAPL Sources at the Water Table

Distinguishing the vapor source type (LNAPL vs. dissolved phase) is essential for choosing the appropriate vertical screening distance (Lahvis et al. [2013](#); EPA [2015a](#)). A number of LNAPL indicator criteria have been proposed to make this distinction (Peargin and Kolhatkar [2011](#); CRC CARE [2013](#); Lahvis et al. [2013](#); ITRC [2014](#); USEPA [2015a](#); NJDEP [2018](#)).

In this study, we used a concentration of 1000 $\mu\text{g}/\text{L}$ benzene in groundwater as the threshold to distinguish between dissolved-phase contamination in groundwater and LNAPL as the source of contamination in the soil gas. This was empirically derived as the 5th percentile of the distribution of average benzene concentrations in groundwater observed in time series data from 269 wells (83 sites) that historically showed measurable LNAPL at some point in time (Peargin and Kolhatkar [2011](#); ITRC [2014](#)). This meant that if the benzene concentration in groundwater exceeded 1000 $\mu\text{g}/\text{L}$, then there was a 95% probability of residual LNAPL being present, even in the absence of measurable LNAPL in the well.

Applicability of US EPA Recommended Vertical Screening Distances Based on Soil Vapor Transport Modeling
Vapor transport modeling was conducted using BioVapor (DeVaull [2007](#); API [2010](#)) to further support the empirical data assessment of vertical screening distances. BioVapor is an analytical vapor intrusion model that simulates one-dimensional diffusive vapor transport to indoor air and includes oxygen-limited biodegradation in the vadose zone (DeVaull [2007](#); API [2010](#)). The modeling was conducted to calibrate first order aerobic biodegradation constants (in the aqueous phase) for 1,2-DCA, and then to apply these rate constants to estimate vertical screening distances assuming conservative source concentration and transport parameters and an indoor air VISL at the 10^{-6} risk level for 1,2-DCA ($0.11 \mu\text{g}/\text{m}^3$). The model approach is described in greater detail in the following sections.

Estimation of First-Order Aerobic Biodegradation Rate Constants for 1,2-DCA

First-order aerobic biodegradation rate constants for 1,2-DCA were estimated by a best fit of the measured and simulated soil-gas concentration data. A conceptualization of the model domain and boundary conditions used for model calibration is shown in Figure 2 (panels A and B). The model input and parameter values used in transport modeling are summarized in Table 3. The key model assumptions used in the derivation of rate constants are listed below.

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

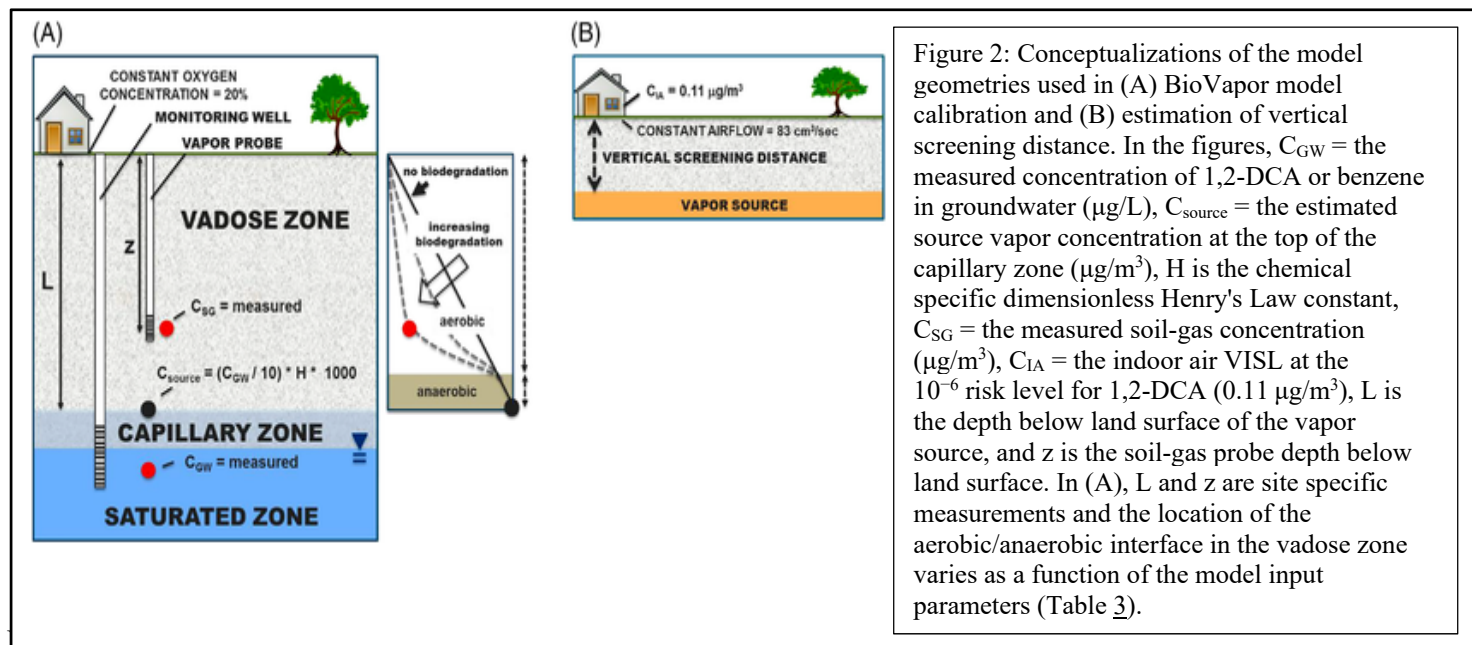


Figure 2: Conceptualizations of the model geometries used in (A) BioVapor model calibration and (B) estimation of vertical screening distance. In the figures, C_{GW} = the measured concentration of 1,2-DCA or benzene in groundwater ($\mu\text{g}/\text{L}$), C_{source} = the estimated source vapor concentration at the top of the capillary zone ($\mu\text{g}/\text{m}^3$), H is the chemical specific dimensionless Henry's Law constant, C_{SG} = the measured soil-gas concentration ($\mu\text{g}/\text{m}^3$), C_{IA} = the indoor air VISL at the 10^{-6} risk level for 1,2-DCA ($0.11 \text{ } \mu\text{g}/\text{m}^3$), L is the depth below land surface of the vapor source, and z is the soil-gas probe depth below land surface. In (A), L and z are site specific measurements and the location of the aerobic/anaerobic interface in the vadose zone varies as a function of the model input parameters (Table 3).

$\geq 1\%$ v/v (API 2009). At concentrations of oxygen less than 1% v/v, only diffusive transport was considered. A 20% v/v constant concentration oxygen boundary condition was assumed at the land surface to be representative of the conditions at soil-gas sampling locations. Four constituents (1,2-DCA, benzene, the aliphatic fraction of Total Petroleum Hydrocarbon [TPH], and the aromatic fraction of TPH) were modeled to account for total biological oxygen demand resulting from a release of weathered gasoline.

It was assumed that the source vapor concentrations for 1,2-DCA, benzene, and the aliphatic and aromatic fractions of TPH were constant at the top of the capillary zone. These were estimated using measured concentrations of 1,2-DCA and benzene in groundwater as follows. Source vapor concentrations for benzene and 1,2-DCA were estimated assuming equilibrium partitioning according to Henry's Law (assuming a constant and uniform subsurface temperature of 15°C) and a 10-fold reduction in vapor concentrations across the capillary zone (McCarthy and Johnson 1993).

A constant and uniform effective diffusion coefficient was estimated for the vadose zone based on site-specific soil types reported in the data base and default soil properties (total porosity, bulk density, and moisture content) recommended by Environmental Quality Management (2004) for VI modeling applications, and air-phase tortuosity values based on Millington-Quirk relations (Millington, 1959). A soil respiration rate was estimated based on an assumed fraction organic carbon value of 0.002 (g organic carbon/g soil) for all soil types (US EPA 1996).

Default values for building dimensions, foundation properties, and exposure parameters recommended by Environmental Quality Management (2004) for VI modeling applications were used. Advection caused by density or pressure effects in the vadose zone (associated with processes such as barometric pumping, fluctuations in water level table and methanogenesis) were assumed to be negligible compared to diffusion. Additional oxygen demand associated with the oxidation of methane resulting from methanogenesis was not considered in the BioVapor modeling. This was based on the analysis of the US EPA and Australian PVI databases

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

(<https://www.epa.gov/ust/petroleum-vapor-intrusion-database>) which indicated that methane concentrations over 1% v/v were rarely detected in soil gas at petroleum UST sites (in 33 out of 519 samples, or 6%).

When 1,2-DCA was not detected in the soil-gas sample, the associated detection limit was used for calibration. The 1,2-DCA soil-gas concentration data used for model calibration were initially filtered to eliminate groundwater/soil-gas data pairs for which first-order aerobic biodegradation rate constants could not be calibrated. This occurred when the measured soil-gas concentrations were greater than those predicted by the BioVapor model based on Fick's Law of diffusion without biodegradation. This could be caused by several factors, including error or bias in the measurement of the concentrations of 1,2-DCA in groundwater which resulted in reported concentrations that were less than the true concentrations (e.g., due to the effect of borehole dilution), or the presence of vadose-zone sources of vapors that could not be identified based on the available site data. In addition, soil-gas concentration data collected directly above locations (soil-gas probes) where soil-gas concentrations were nondetect were not used for model calibration.

After data filtering, only 30 of the 144 groundwater/soil-gas data pairs were suitable for BioVapor model calibration. This was done by varying the first-order aerobic biodegradation rate constant until best fits of model-predicted and measured soil-gas concentration data were achieved. Rate constants were estimated for dissolved-phase and LNAPL sources of petroleum hydrocarbons, which were differentiated using the benzene groundwater concentration threshold criterion (1000 µg/L) described earlier (Peargin and Kolhatkar 2011; ITRC 2014). The rate constants (and associated vertical screening distances) estimated by transport modeling were assumed to be conservative, yet suitable for screening-distance derivation, given the model was calibrated to largely nondetect soil-gas concentrations at analytical reporting limits greater than actual concentrations (28 out of 30 instances). The predominant types of vadose zone soils reported at these 30 locations were: sand (16 locations), loam (3 locations), silt (5 locations), sandy clay (2 locations), and silty clay (4 locations).

Prediction of Vertical Screening Distances Using the BioVapor Model

Vertical screening distances were determined for 1,2-DCA by varying the vertical source-building separation distance until predicted concentrations in indoor air were less than the indoor air target concentration at 10^{-6} risk level ($0.11 \mu\text{g}/\text{m}^3$) as shown in Figure 2b. Conservative estimates of the vertical screening distances were derived for sand, loam, and clay soils and dissolved-phase and LNAPL sources assuming (1) aerobic biodegradation rate constants determined by model calibration for dissolved-phase and LNAPL sources, (2) 95th percentile source vapor concentrations of 1,2-DCA, benzene, and TPH aliphatic and aromatic hydrocarbons for dissolved-phase and LNAPL sources, (3) a constant airflow of $83 \text{ cm}^3/\text{s}$ through a slab-on-grade building foundation, and (4) the model assumptions and parameter values previously presented (Table 3).

The source vapor concentrations for 1,2-DCA and benzene were determined based on the 95th percentile groundwater concentrations in the current dataset, assuming equilibrium partitioning according to Henry's Law and attenuation factor of 0.1 across the capillary fringe (McCarthy and Johnson 1993). The source vapor concentrations of aliphatic and aromatic hydrocarbons were estimated by applying these source vapor concentrations of benzene to the constituent concentration relationships for weathered gasoline (API 2010). The 95th percentile source vapor concentrations were used for petroleum hydrocarbons (benzene and TPH aliphatic and aromatic fractions) to maximize oxygen demand (i.e., limit the potential for aerobic biodegradation of 1,2-DCA in the vadose zone) and thereby yield longer, more conservative vertical screening distance estimates. The poor correlation between 1,2-DCA and benzene concentrations in groundwater ($R^2 = 0.27$, Figure S1) indicates that the combined use of 95th percentile concentrations of 1,2-DCA and benzene to define source vapor concentrations is a very conservative assumption.

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

Table 3

BioVapor Model Input Used in the Derivation of first-Order Aerobic Biodegradation Rate Constants and Vertical Screening Distances for 1,2-DCA

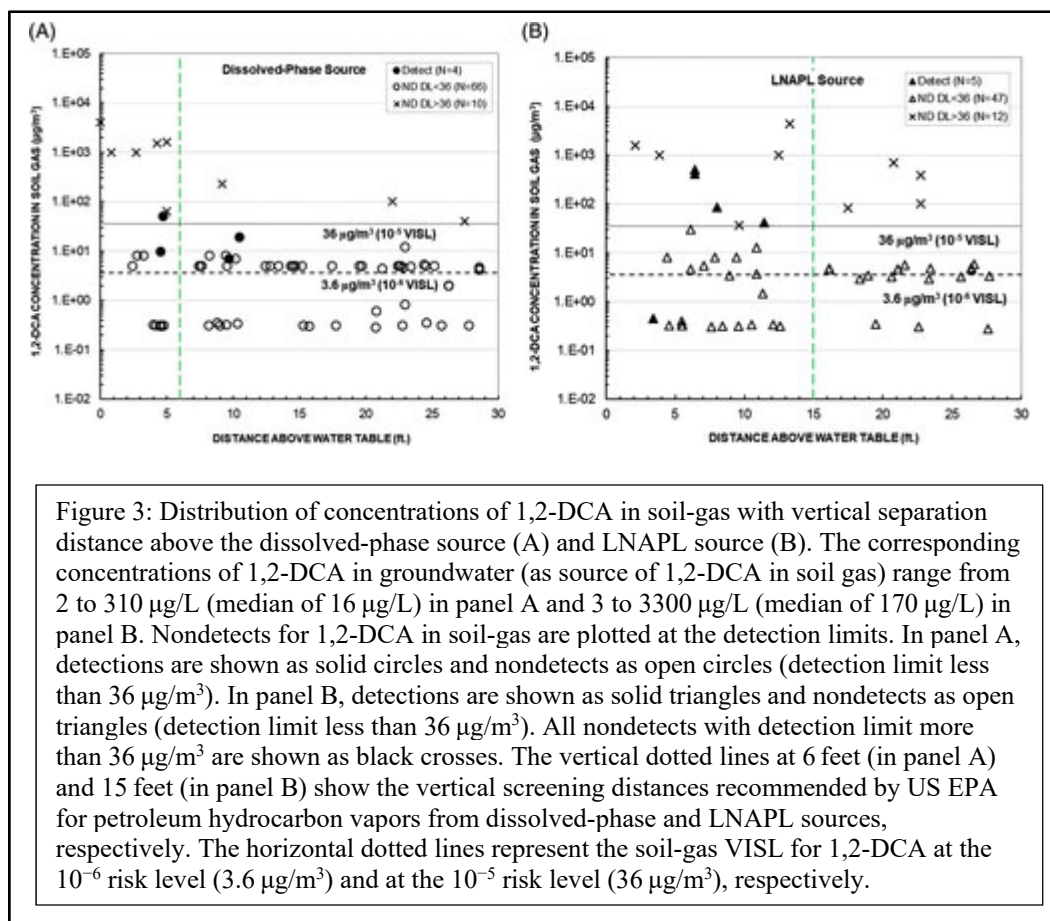
Parameter	Value	Units
<i>Exposure and risk factors</i>		
Target hazard quotient for individual chemicals	1	Dimensionless
Target excess individual lifetime cancer risk	0.000001	Dimensionless
Carcinogen averaging time	70	yrs
Non-carcinogen averaging time	30	yrs
Body weight—adult	70	kg
Exposure duration	30	yrs
Exposure frequency	350	days/yr
Indoor inhalation rate exposure adjustment	1	Dimensionless
<i>Building parameters</i>		
Indoor mixing height	244	cm
Air exchange rate	6	1/day
Foundation thickness	15	cm
Foundation area	1,060,000	cm ²
Foundation crack fraction	0.000377	cm ² -cracks/cm ² -total
Total porosity (soil-filled cracks)	1	cm ³ -void/cm ³ -soil
Water filled porosity (soil-filled cracks)	0	cm ³ -water/cm ³ -soil
Airflow through basement foundation ³	83	cm ³ -air/s
Oxygen concentration at the land surface ¹	20	% v/v
<i>Vadose zone parameters</i>		
Soil Type	Soil Porosity ² (cm ³ -void/cm ³ -soil)	Soil Water Content ² (cm ³ -water/cm ³ -soil)
Sand	0.375	0.054
Loam	0.399	0.148
Clay	0.415	0.215
Soil organic carbon fraction		0.002
Annual median soil temperature		15
Baseline soil oxygen respiration rate		variable
Depth to source (from bottom of foundation)		variable
Minimum oxygen concentration for aerobic biodegradation		1
<i>95% maximum constituent source groundwater concentrations (µg/L)</i>		
Constituent	Dissolved-Phase Source	LNAPL Source ⁵
1,2 DCA	310	760
Benzene	930	19,000
<i>95% maximum constituent source vapor concentrations (µg/m³)⁶</i>		
Constituent	Dissolved-Phase Source	LNAPL Source
1,2 DCA	769	1880
Benzene	14,100	289,000
TPH Aliphatics ⁷	804,000	16,400,000
TPH Aromatics ⁷	63,400	1,340,000

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

Results and Discussion

Empirical Evaluation of the Applicability of US EPA Recommended Vertical Screening Distances for 1,2-DCA

The vertical screening distances for benzene and the other petroleum hydrocarbons were derived as the vertical separation distances above the dissolved-phase and LNAPL sources required to reduce the contaminant concentration in soil gas below the risk-based screening level with $\geq 95\%$ probability (EPA 2013; Lahvis et al. 2013). The applicability of these recommended vertical screening distances for 1,2-DCA was evaluated by examining the percentage frequency of 1,2-DCA detections in soil gas below the soil-gas VISL at the 10^{-6} risk level ($3.6 \mu\text{g}/\text{m}^3$) for different vertical separation distances for dissolved-phase and LNAPL sources, respectively. Figure 3 shows concentration of 1,2-DCA in soil gas plotted as a function of the vertical separation distance (distance above water table) from dissolved-phase (panel A) and LNAPL sources (panel B). Concentrations of 1,2-DCA in groundwater for these data sets ranged from 2.2 to 310 $\mu\text{g}/\text{L}$ for dissolved-phase sources and 3 to 3,300 $\mu\text{g}/\text{L}$ for LNAPL sources.



As seen in panel A, 1,2-DCA was detected in soil gas in four instances (5%) and was nondetect in 76 instances (95%) associated with a dissolved-phase source. Concentrations of 1,2-DCA in soil gas in two out of these four detections were greater than $3.6 \mu\text{g}/\text{m}^3$ (soil-gas VISL at the 10^{-6} risk level) and occurred at vertical separation distances greater than 6 feet (1.8 m). The highest vertical separation distance for one detection (at $19 \mu\text{g}/\text{m}^3$) was observed at 10.5 feet (3.2 m). The two detections exceeding $3.6 \mu\text{g}/\text{m}^3$ at vertical separation distances greater than 6 feet (1.8 m) occurred at locations with predominantly sandy soil type in the vadose zone.

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

In the case of soil gas above LNAPL sources (panel B), 1,2-DCA was detected in five instances (8%) and was nondetect in 59 instances (92%). There were no detections of 1,2-DCA in soil gas at vertical separation distance greater than 15 feet (4.6 m). The highest vertical separation distance where 1,2-DCA was detected was 11.4 feet (3.5 m) at the concentration of $42 \mu\text{g}/\text{m}^3$. The large percentage of nondetect data (more than 90%) in this dataset indicated significant attenuation of 1,2-DCA within 15 feet (4.6 m) in the vadose zone above both dissolved and LNAPL sources.

Table 4 summarizes the frequency distribution of 1,2-DCA detections in soil gas at concentrations less than 3.6 and $5 \mu\text{g}/\text{m}^3$ at different vertical separation distances above the source. A large number of the soil-gas data were nondetect at a detection limit of $5 \mu\text{g}/\text{m}^3$, which is just above $3.6 \mu\text{g}/\text{m}^3$ (soil-gas VISL at the 10^{-6} risk level) and were included in the comparison. If 95% of the measured concentrations were less than the VISL at a particular separation distance, that distance can be considered protective. This analysis indicated that 6 feet (1.8 m) vertical screening distance may not be applicable for 1,2-DCA vapors from a dissolved-phase source (only 90% frequency of detection at $\leq 3.6 \mu\text{g}/\text{m}^3$ while 96% frequency of detection at $\leq 5 \mu\text{g}/\text{m}^3$), while 11 feet (3.4 m) might be adequately protective based on the current dataset (100% frequency of detection at $\leq 3.6 \mu\text{g}/\text{m}^3$). On the other hand, the 15 feet (4.6 m) vertical screening distance recommended by US EPA could be applied to 1,2-DCA vapors from LNAPL source (100% frequency of detection at $\leq 3.6 \mu\text{g}/\text{m}^3$).

First-Order Aqueous Phase Biodegradation Rate Constants for 1,2-DCA

The distribution of first order aerobic biodegradation rate constants for 1,2-DCA determined by model calibration are plotted in Figure 4 as a function of the entire data set (ALL) and source type (dissolved-phase and LNAPL sources). The distributions do not consider two outlier values, one estimated for dissolved-phase source (9 1/h) and another for LNAPL source (4 1/h). These were very high rate constants that could not be explained and thus were excluded from the analysis. The median aerobic biodegradation rate constants for a dissolved-phase source (9.54×10^{-4} 1/h, half-life = 30 days) and an LNAPL source (1.12×10^{-3} 1/h, half-life = 27 days) are nearly equivalent. The median rate constants in this study were approximately twofold lower than the geometric mean of the rate constants documented by Ma et al. (2016) based on numerous laboratory studies (2.6×10^{-3} 1/h, half-life = 11 days) and approximately two orders of magnitude lower than the rate constants reported for petroleum hydrocarbons, such as benzene (7.9×10^{-1} 1/h, half-life = 0.04 days) (DeVaul 2007).

Table 4
Occurrence of 1,2-DCA Detections in Soil-Gas at Concentrations $\leq 3.6 \mu\text{g}/\text{m}^3$ (Soil-Gas VISL for 1,2-DCA at the 10^{-6} Risk Level) and $\leq 5 \mu\text{g}/\text{m}^3$ Above Different Vertical Separation Distances from Dissolved-Phase and LNAPL Sources in This Dataset¹

Source Type (Data Points with Detection Limit $< 36 \mu\text{g}/\text{m}^3$)	Vertical Separation Distance (feet)	Number of Detections > 3.6 and $> 5 \mu\text{g}/\text{m}^3$	Frequency of 1,2-DCA Detections in Soil-Gas $\leq$ Specified Concentration (%) ²			
			Total Number of Data Points ¹		$3.6 \mu\text{g}/\text{m}^3$	$5 \mu\text{g}/\text{m}^3$
Dissolved source (70)	≥ 0	4	29	61	86	93
	≥ 6	2	21	52	90	96
	≥ 8	2	21	49	90	96
	≥ 10	1	16	43	94	98
	≥ 11	0	14	41	100	100
LNAPL source (52)	≥ 0	4	30	42	87	90
	≥ 8	2	22	33	91	94
	≥ 10	1	18	29	94	97
	≥ 12	0	15	23	100	100
	≥ 15	0	13	23	100	100

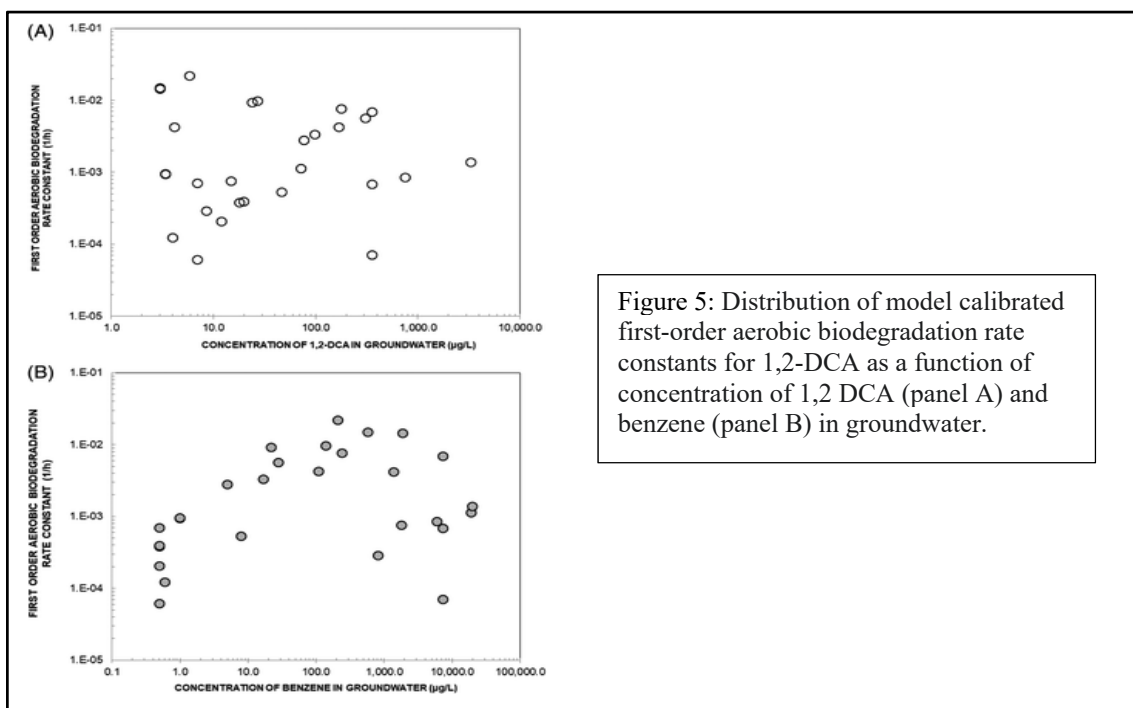
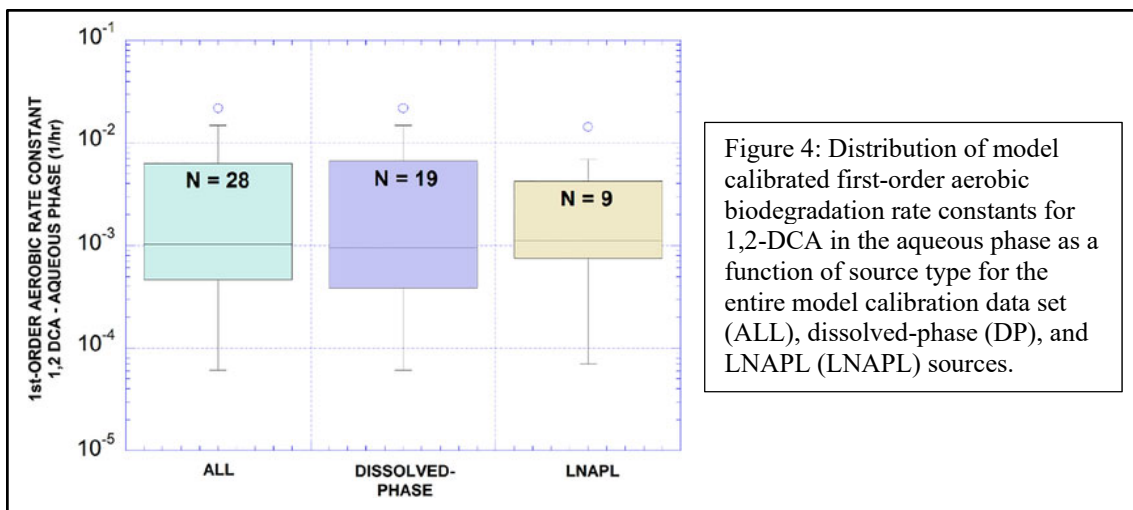
¹Includes nondetect data only when detection limit ≤ 3.6 and $\leq 5 \mu\text{g}/\text{m}^3$, respectively.

²% Frequency = $(1 - (\text{Number of detections})/(\text{Total number of data points})) \times 100$.

³The shaded portions highlight data for > 6 ft. (for dissolved source) and > 15 ft. (LNAPL source).

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

The lower rate constant for 1,2-DCA compared to literature values from Ma et al. (2016) likely reflects the conservatism in the model calibration approach used in this study. The rate constants also showed little variability as a function of 1,2-DCA concentration in groundwater (3–3300 $\mu\text{g/L}$) and source type (see Figure 5, panel A). Figure 5 (panel B) also shows the lack of dependence of the biodegradation rate constants as a function of benzene concentration in groundwater, which ranged from 0.5 to 1000 $\mu\text{g/L}$ (for dissolved-phase sources) and 1000 to 20,000 $\mu\text{g/L}$ (for LNAPL sources) in this dataset. Any meaningful enhancement in the biodegradation rate of 1,2-DCA due to co-metabolism of petroleum hydrocarbon substrates (Hazen 2010) is not evident from this analysis. The median calibrated biodegradation rate constants for dissolved-phase (9.54×10^{-4} 1/h) and LNAPL (1.12×10^{-3} 1/h) sources were used as input in the derivation of vertical screening distances for sand, loam, and clay vadose zone soils as described in the following section.



Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

Predicted Vertical Screening Distances for 1,2-DCA Using BioVapor

The vertical screening distances for 1,2-DCA predicted by BioVapor modeling ranged from 5 to 28 feet (1.5 to 8.5 m) for dissolved-phase sources and 6 to 50 feet (1.8 to 15 m) for LNAPL sources (Table 5). The vertical screening distances estimated for 1,2-DCA were generally equal to or less than the 6 feet (1.8 m) and 15 feet (4.6 m) screening distances recommended by US EPA (2015a) for all sites except for sites with sand soils in the vadose zone. This was generally consistent with the empirical observation that the only two detections of 1,2-DCA in soil gas more than 3.6 $\mu\text{g}/\text{m}^3$ at vertical distance more than 6 feet (1.8 m) from dissolved-phase sources (Figure 3a) occurred at locations with sandy soil. The BioVapor model results were conservative compared to the empirical analysis of the vertical separation distances in this soil-gas data. This is expected given that vadose zones at most sites are likely to contain some interbeds of non-sand soils (e.g., loam, silt, and clay).

Table 5

Summary of Estimated Vertical Screening Distances for 1,2-DCA as a Function of Soil Type and Source Type (Dissolved-Phase or LNAPL) based on Conservative Vapor Transport Modeling

Soil Type	Vertical Screening Distance—Feet (Meters)	
	Dissolved-Phase Source (Median Rate Constant = $9.54 \times 10^{-4} \text{ h}^{-1}$, Half Life = 30 Days)	LNAPL Source (Median Rate Constant = $1.12 \times 10^{-3} \text{ h}^{-1}$, Half Life = 27 Days)
Sand	28 (8.5)	50 (15.2)
Loam	7 (2.1)	8 (2.4)
Clay	5 (1.5)	6 (1.8)

The assumptions include first-order aerobic biodegradation rate constants for 1,2-DCA for dissolved-phase and LNAPL sources, 95% maximum source vapor constituent (Benzene and TPH aliphatic and aromatic hydrocarbons) concentrations for dissolved and LNAPL sources, and default vadose zone soil moisture and total porosity values (Table 3).

The sensitivity of the vertical screening distance to soil type was not observed for hydrocarbons such as benzene that aerobically biodegrade with rate constants that are more than 2 orders of magnitude higher (Lahvis et al. 2013; US EPA 2013). The dependence of the vertical screening distance on soil type highlights the importance of proper characterization and classification of vadose zone soils (e.g., geotechnical analyses of soil type, soil properties) when assessing vapor intrusion risks from 1,2-DCA.

Figure 6 shows source concentrations of 1,2-DCA in groundwater and vertical separation distances in sand, loam, and clay soils that result in indoor air concentrations of 0.11 $\mu\text{g}/\text{m}^3$

(indoor air VISL at the 10^{-6} risk level for 1,2-DCA). The nomograph was developed based on conservative BioVapor modeling using the median values of first-order rate constants ($9.54 \times 10^{-4} \text{ 1/h}$ for dissolved-phase sources and $1.12 \times 10^{-3} \text{ 1/h}$ for LNAPL sources), 95th percentile source vapor concentrations for benzene and TPH aliphatic and aromatic hydrocarbons in combination with default soil and building parameter values (Table 3). The horizontal dash-dot line represents the maximum expected concentration of 1,2-DCA in groundwater (3700 $\mu\text{g}/\text{L}$) in equilibrium with leaded automobile gasoline (Falta 2004). Measured concentrations of 1,2-DCA in groundwater greater than this value indicate a potential vapor source other than leaded automobile gasoline.

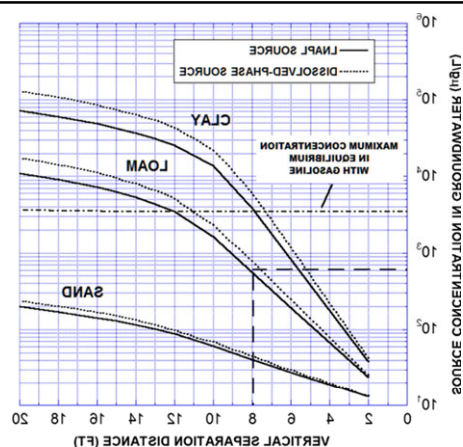


Figure 6: Source concentrations of 1,2-DCA in groundwater and vertical separation distances in sand, loam and clay soils that result in indoor air concentrations of 1,2-DCA of 0.11 $\mu\text{g}/\text{m}^3$ (indoor air VISL at the 10^{-6} risk level). The horizontal dash-dot line represents the concentration of 1,2-DCA in groundwater in equilibrium with leaded automobile gasoline (approximately 3,700 $\mu\text{g}/\text{L}$). The dashed lines show the vertical separation distance needed (i.e., vertical screening distance of 8 feet) for a LNAPL source with 500 $\mu\text{g}/\text{L}$ 1,2-DCA in groundwater, and loam soil in the vadose zone. If the source concentration in groundwater and the vertical separation distance (or depth to groundwater) at a site falls below the line for the type of vapor source and soil type in the vadose zone, then the indoor air concentration of 1,2-DCA in an overlying building will be less than 0.11 $\mu\text{g}/\text{m}^3$. No further evaluation of vapor intrusion from 1,2-DCA will be needed under these conditions.

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

The nomograph shown in Figure 6 can be used for site screening purposes based on the concentration of 1,2-DCA in groundwater, type of vapor source (dissolved-phase vs. LNAPL), and the predominant soil type in the vadose zone. For example, a minimum vertical separation distance of 8 feet (2.4 m) is needed to screen out a site with a LNAPL source, concentration of 1,2-DCA in groundwater of 500 $\mu\text{g/L}$, and a loam soil in the vadose zone (see dashed lines). No additional investigation of vapor intrusion risk from 1,2-DCA would be necessary for any combination of 1,2-DCA concentration in groundwater and vertical separation distance (or depth to groundwater) that plots below the individual curve for a given soil and source type.

Figure 6 can also be used to indicate the range of conditions for which the US EPA recommended vertical screening distances are applicable. The 6 feet (1.8 m) screening distance for dissolved-phase sources appears applicable only for concentrations of 1,2-DCA in groundwater lower than 30, 250, and 1100 $\mu\text{g/L}$ at sites with sand, loam, and clay soils in the vadose zone, respectively. The vertical screening distance of 15 feet (4.6 m) for petroleum UST sites with LNAPL sources appears applicable across the entire range of expected source concentrations of 1,2-DCA in groundwater (less than 3700 $\mu\text{g/L}$) at sites with loam or clay soils, but not sandy soils in the vadose zone. These threshold groundwater concentration limits defined in Figure 6 are consistent with those reported by Ma et al. (2016) in model simulations that assumed aerobic biodegradation of 1,2-DCA across the entire vadose zone at rates of 2.6×10^{-3} 1/h.

In summary, soil type and source concentration are important factors to consider in screening vapor intrusion risks for 1,2-DCA. More refined estimates of vertical screening distances can also be obtained using BioVapor with site-specific soil property data (moisture content, total porosity, bulk density, and fraction organic carbon) and source concentrations (groundwater or soil gas) as input.

Conclusions

To our knowledge, this is the first study using soil-gas data for 1,2-DCA from petroleum UST sites to demonstrate that the US EPA recommended vertical screening distance of 15 feet (4.6 m) is applicable to 1,2-DCA at the 10^{-6} risk level. Empirical data analysis was applied to 144 pairs of soil gas and groundwater concentration data from 47 UST sites to support this conclusion. These data were representative of conditions when 1,2-DCA soil vapors were sourced from either LNAPL or dissolved-phase sources at the water table (similar approach to that described in Lahvis et al. 2013 and US EPA 2015a for establishing vertical screening distances for benzene and other petroleum hydrocarbons).

At the 47 UST sites, 1,2-DCA was rarely detected in the soil gas above dissolved-phase and LNAPL sources (5% and 8% of locations, respectively) suggesting significant attenuation due to aerobic biodegradation in the vadose zone. In order to assess the applicability of the US EPA recommended vertical screening distances to 1,2-DCA, the frequency of occurrence of 1,2-DCA in soil gas at concentrations $\leq 3.6 \mu\text{g/m}^3$ (soil-gas VISL at the 10^{-6} risk level) at different vertical separation distances from the source was evaluated (Table 4). This analysis indicated that 6 feet (1.8 m) as the vertical screening distance for dissolved-phase source was not applicable for 1,2-DCA (soil-gas concentrations $\leq 3.6 \mu\text{g/m}^3$ in only 90% of samples at separation distances ≥ 6 feet (1.8 m)). However, 15 feet (4.6 m) as the vertical screening distance for LNAPL source was applicable for 1,2-DCA (soil-gas concentrations $\leq 3.6 \mu\text{g/m}^3$ in 100% of samples at separation distances ≥ 15 feet [4.6 m]).

First order aerobic biodegradation rate constants for 1,2-DCA were estimated by the best fit between the measured soil-gas concentrations and the predicted soil-gas concentrations using the BioVapor model. The median aerobic rate constants for 1,2-DCA (approximately 1×10^{-3} 1/h) were twofold lower than those reported by Ma et al. (2016) and approximately two orders of magnitude lower than those reported for benzene (DeVaull 2007). The actual aerobic

Vertical Screening Distance Criteria to Evaluate Vapor Intrusion Risk from 1,2-Dichloroethane (Continued)

biodegradation rate constants for 1,2-DCA are likely to be greater than these estimates given that the soil-gas concentrations used for model calibration were largely nondetect.

The BioVapor model was used to predict vertical screening distances for 1,2-DCA in soil gas from dissolved-phase and LNAPL sources at the water table using conservative estimates of source concentrations (Table 5). Given the lower aerobic biodegradation rate constants for 1,2-DCA, the vertical screening distances were also dependent on the soil type in the vadose zone and the source concentration of 1,2-DCA in groundwater (Figure 6), in contrast to the US EPA recommended vertical screening distances for petroleum hydrocarbons. The relationships shown in Figure 6 can be used to screen sites based on the concentration of 1,2-DCA in groundwater and the vertical separation distance (or depth to groundwater) for the relevant source type and the predominant soil type in the vadose zone. This approach can potentially reduce unnecessary soil-gas sampling at numerous sites with historic releases of leaded fuels.

The modeling also indicated that the US EPA recommended vertical screening distance of 6 feet (1.8 m) for dissolved-phase sources was applicable for sites only below certain threshold concentrations of 1,2-DCA in groundwater (30, 250, and 1100 $\mu\text{g/L}$ at sites with sand, loam, and clay soils in the vadose zone, respectively). Similarly, the modeling indicated that the US EPA recommended vertical screening distance of 15 feet (4.6 m) for LNAPL sources was applicable for sites with clay and loam soils in the vadose zone, but not sites with sand soil, if the concentration of 1,2-DCA in groundwater was more than 150 $\mu\text{g/L}$. This likely reflected the conservatism in the BioVapor modeling, as the empirical data representing multiple soil types (with approximately 50% locations with sand) suggested significant attenuation of 1,2-DCA vapors above dissolved-phase and LNAPL sources. The soil-gas data also supported the applicability of the US EPA recommended vertical screening distance of 15 feet (4.6 m) for both dissolved and LNAPL sources.

Although EDB was not detected in the soil gas in this dataset, the reporting limits were greater than 1.6 $\mu\text{g/m}^3$ (soil-gas VISL for EDB at the 10^{-5} risk level). As a result, this dataset could not be used to empirically evaluate the applicability of US EPA recommended vertical screening distances for EDB. The conventional analytical methods (EPA TO-15 or EPA 8260B) used for soil-gas analysis in this dataset were not sensitive and selective enough to detect EDB at low enough concentrations to be meaningful for risk assessment, likely due to matrix interference from other constituents (i.e., BTEX and other gasoline range hydrocarbons). To address this data gap, there is a need to develop a more sensitive and selective analytical method for soil-gas analysis or to configure existing sampling procedures and analytical protocols to be able to detect EDB at or below 0.16 $\mu\text{g/m}^3$ (soil-gas VISL at the 10^{-6} risk level). To evaluate vertical screening distance criteria for EDB using the approach described here, additional field studies using such an analytical method are needed to gather concurrent soil gas and groundwater data at sites with EDB detections in groundwater over 0.18 $\mu\text{g/L}$ (groundwater VISL at the 10^{-6} risk level).

Read more about this article at: <https://ngwa.onlinelibrary.wiley.com/doi/10.1111/gwmmr.12351>

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

Introduction

The influence of the migration of groundwater on building foundations, tunnels, and slope stability should not be neglected as there could be detrimental consequences, such as foundation and structural damage (Yang et al. 2018). However, it is currently difficult to determine water seepage in rock and soil masses in real time, as well as the different levels of seepage in different areas of the rock and soil masses. Therefore, a thorough

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

understanding of water seepage in rock and soil masses is very important for engineering practices (Jiang et al. 2013; Sun et al. 2014; Chambers et al. 2015; Uhlemann et al. 2016; Dimech et al. 2019).

Geoelectric fields, including natural and artificial geoelectric fields, are induced on the surface of the Earth. They are important as they drive currents through conductor networks and therefore an important component of the electrical prospecting method, which measures surface potential. When groundwater infiltrates cracks in rocks and soil, there is interaction between the groundwater and rock and soil media (Zhang et al. 1995) and therefore an increase in the volume of ions in the rock or soil (Shen 1987). A higher ion content means higher electrical conductivity. In addition, groundwater becomes diffused, adsorbed, and filtered, and undergoes redox reactions during the infiltration process, which significantly change the geoelectric fields of the solid and liquid media (Zheng 2006; Liu et al. 2009). In the natural environment, the presence of geoelectric fields makes it possible to use an instrument that is placed at two points on the ground to measure the potential difference, which is related to the natural ground electric currents. One of the geophysical measurements that is relevant to the groundwater flow field is spontaneous potential (SP). SPs are electrical potential differences that are naturally found in the Earth. They occur when a semi-permeable body inhibits ion diffusion through the voids of rocks or naturally, when a conducting fluid flows through rocks. The use of SP is a generally acceptable method to determine the flow of fluid. The SPs in soil, clay, or rock can be examined through diffusion, electro-filtration, and oxidation-reduction potentials (Sun et al. 2014; Geng et al. 2017). Therefore, applying the geoelectric method helps to identify and monitor groundwater seepage to prevent damage to structures, infrastructures, landslides, etc.

Previous work in the literature over the last few decades has applied geoelectric fields, such as examining water seepage in coal mining (Liu et al. 2013a), conducting mineral exploration (Corry 1985), determining changes in hydrothermal systems (Legaz et al. 2009), examining contaminant transport, conducting tunnel construction, excavating foundation pit, and managing slope seepage (Geng et al. 2017). The further exploitation of deep underground caves and prevalent use of advance mining have led to the increasing importance of innovative techniques for observing groundwater seepage (Liu et al. 2013b), of which technologies for measuring the geoelectric field have been of paramount importance. Among the different technologies, the resistivity method has been very successful (Nowroozi et al. 1999; Petalas and Diamantis 1999; Nassir et al. 2000; Somay and Gemici 2009; Gemici et al. 2016; Zhou et al. 2017; Dimech et al. 2019), and the use of SP has also provided very robust results in determining the amount of groundwater seepage (Bogoslovsky and Ogilvy 1970; Sill 1983; Birch 1994; Hashimoto and Tanaka 1995; Doussan et al. 2002; Revil et al. 2005; Jardani et al. 2006). Zohdy (1969) used the resistivity method in a groundwater exploration program for two locations near El Paso, Texas in the United States and concluded that the static water table was found at a depth of about 91.4 m, while the freshwater interface varied from 91.4 m to about 518.2 m. Nowroozi et al. (1999) found that the Schlumberger sounding resistivity method is an efficient means of investigating the interface between salt water and freshwater in the geological setting of the eastern shore of Virginia in the United States. Jougnot et al. (2015) simulated the SP response to the infiltration of rain water and groundwater in soil with a hydrogeophysical model and concluded that using SP can detect fluxes of fluid flow. All of these efforts have proven that geoelectric field signatures are closely related to water flow during infiltration. However, only a small number of hydrological studies have monitored the dynamic changes in groundwater seepage (Pellerin 2002; Chandra et al. 2008; Sun et al. 2014; Dimech et al. 2019). Moreover, there is the need for long-term monitoring, and more in-depth quantitative analyses of groundwater seepage should be carried out.

In response to this research gap, a physical model for groundwater seepage was designed to dynamically monitor the transient response of SPs, excitation currents and apparent resistivity in the seepage process by using an electrical resistivity meter. Pattern of changes in the geoelectric field parameters are discussed quantitatively, which provides a procedure for monitoring groundwater.

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

Experimental Methods

A three-dimensional physical model with dimensions of 60 cm in length, 60 cm in width, and 50 cm in height constructed from 5 mm thick glass was used for the experiment. The model was filled with clay and fine sand, which were placed into four layers. The top layer (Layer I) is 3 cm of thick fine sand to prevent erosion. The second layer (Layer II) is a 20-cm layer of clay and fine sand mixture with a ratio of 3:1. The third layer (Layer III) is a 10-cm layer of clay and the bottom layer (Layer IV) is 3 cm of thick coarse sand to prevent the sand and clay from blocking the outlet (Figure 1). The basic physical properties of the fine sand and clay are listed in Table 1. In the model, graphite rods with diameter 3 mm and length 20 mm were used as the electrodes to measure the changes in potential and electric current. Four measurement lines with seven electrodes in the four layers are labeled from A1 to A7, B1 to B7, C1 to C7, and D1 to D7. The distance between the electrodes and edges of the container is 15 cm and the distance between the adjacent electrodes is 5 cm. The location of the power supply anode B and reference electrode N are shown in Figure 1.

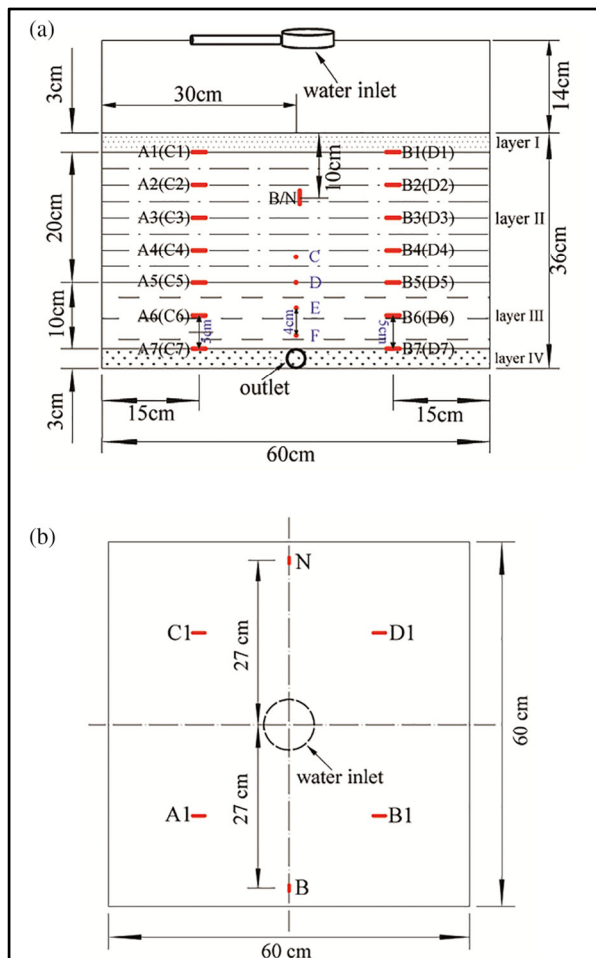


Figure 1: Layers of material and electrode placement in experimental container: (a) longitudinal section (C, D, E and F representing the locations of the resistivity inversion in Figure 2) and (b) transverse section.

The experimental equipment used was an electrical resistivity meter, which can simultaneously measure the temporal and spatial changes through SP, primary field potentials and excitation currents with an emission voltage of 96 V and excitation time of 0.1 s for each voltage. The pole-pole arrangement was adopted. The first electrode was used to inject a current into the soil and all of the other electrodes were used as the potential electrodes at the same time, and then the second electrode was used to inject a current and all of the other electrodes were used as the potential electrodes (Geng et al. 2017). The process was repeated until the last electrode injected an electrical current into the soil. When power is manually supplied for the model, the apparent resistivity is calculated by using the measured voltage and excitation current of each electrode as given by

$$\rho_s = K \frac{\Delta U}{I} \quad (1)$$

where ρ_s is the apparent resistivity, K is a coefficient determined by the electrode arrangement, ΔU is the measured potential difference among the electrodes and I is the excitation current. In the experiment, the apparent resistivity is obtained by the inversion of the test data (voltage and excitation current).

A rain shower nozzle was used to supply tap water at a rate of 0.1 L/min during the infiltration experiment. The electrical resistivity of tap water was 66.7 Ωm at room temperature. The experiment was initiated by flowing tap water into the nozzle and stopped when water started to flow out from the bottom of

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

the model. The second round of water injection started at about 175 min for approximately 15 min before the injection was stopped. The measurement interval was 1 minute in clay layer and 30 s in sand layer. The length of the experiment was 190 min and a total of 200 sets of experimental data were obtained.

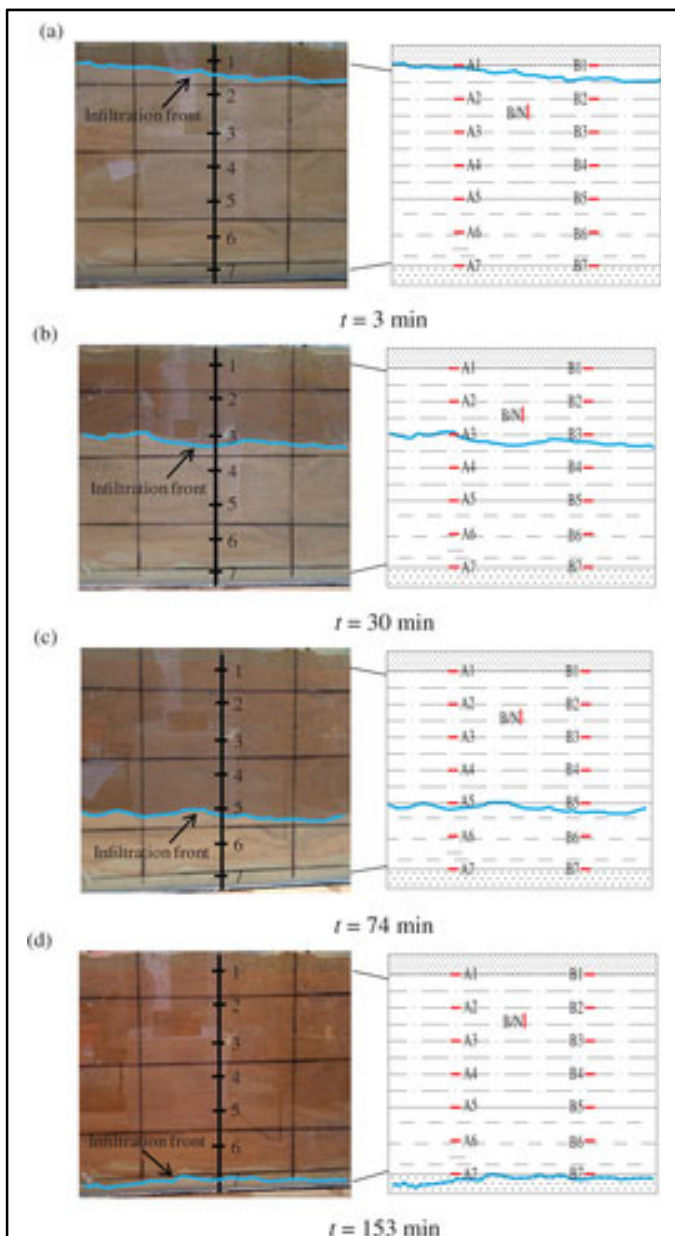


Figure 2: Location of the infiltration front at different times: (a) $t = 3$ min, (b) $t = 30$ min, (c) $t = 74$ min, and (d) $t = 153$ min.

Table 1

Physical Properties of Clay and Fine Sand

Material	Dry Density (g/cm ³)	Plasticity Index (%)	Liquid Index (%)	Hydraulic Conductivity (cm/s)
Clay	1.67	17.6	42	8×10^{-8}
Fine sand	2.13	—	—	5×10^{-3}

Experimental Results

Infiltration Front

A digital camera was used to capture the infiltration front during the infiltration process. The infiltration front continues to descend to lower levels of the box due to gravity (Figure 2). At $t = 3$ min, the infiltration front reaches A1 or B1. At $t = 153$ min, the infiltration front reaches A7 or B7. Finally, the water penetrates throughout the entire model.

SP Response

The natural potential difference between an electrode and the reference electrode N is the SP. Figure 3 shows the variations in the measured SP of each electrode with time. At the beginning of the infiltration process, the SP of each electrode reaches its highest absolute value, which ranges from -200 mV to -550 mV. For example, the SP of A1 to A7 is reduced from -20 to -505 mV, -50 to -508 mV, -34 to -404 mV, -3 to -347 mV, -8 to -368 mV, 73 to -203 mV, 48 to -210 mV and 45 to -306 mV, respectively (see Figure 3a). Then, the measured SP of each electrode, in general, steadily increases with time. With the continued infiltration of water, the SP is reduced by 100 to 200 mV when the water reaches a certain electrode. The time at which the SP changes from a steady increase to decrease of A3 to A7 is 30 , 45 , 74 , 110 , and 153 min, respectively. This time is referred to as the “point of change”. Since water passes through

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

the first two electrodes in just a few minutes, it is difficult to determine the exact time of the “point of change” of the first two electrodes. During the second round of water injection, the measured SP of all the electrodes recovers to the previous extreme values that range from -200 to -550 mV, thus indicating a “memory” effect.

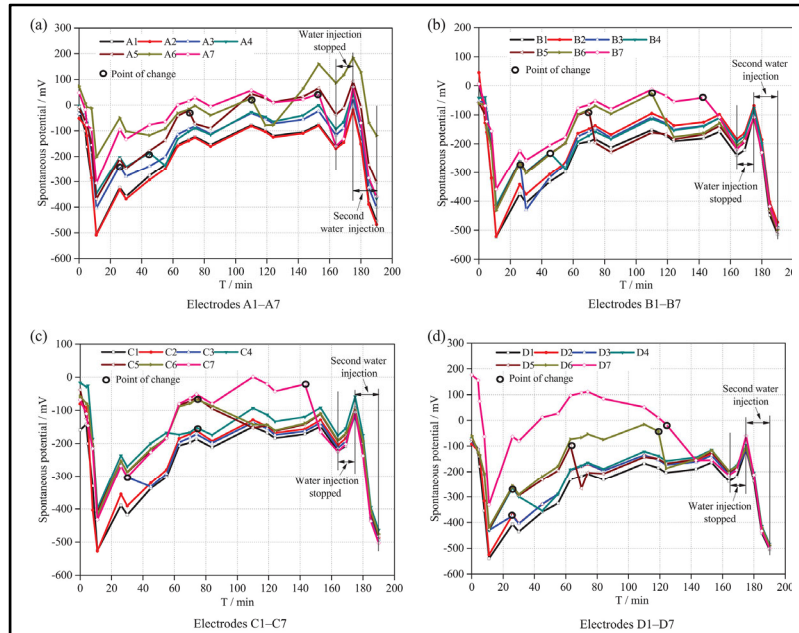


Figure 3: Variations in spontaneous potential with time: (a) Electrodes A1 to A7, (b) Electrodes B1 to B7, (c) Electrodes C1 to C7, and (d) Electrodes D1 to D7.

Excitation Current Response

An excitation voltage of 96 V was supplied to power supply anode B and A7. The potential difference among A1 to A6, B1 to B7, C1 to C7, D1 to D7 and the reference electrode N was measured simultaneously. As shown in Figure 4, the measured excitation current of each electrode increases to the peak value (140 to 220 mA) when water reaches the electrode and subsequently tends to stabilize. The time required for the water to move to each electrode can be observed in Figure 4. Table 2 summarizes the average time for the water to reach the electrodes 1 to 7. Figure 5 shows the infiltration distance H (the distance between the infiltration front and A1 or B1, C1 and D1) with time (t) during the water infiltration process. It is obvious that the variation in infiltration distance can be categorized into two linear stages with time. The slope of the fitting curve is reduced during infiltration. Since fine sand has higher

permeability than clay, the seepage velocity is higher in Layer II than that in Layer III. The calculated seepage velocity in Layers II and III is 4.94×10^{-3} and 2.54×10^{-3} cm/s, respectively. In addition, the time ($t_1, t_2 \dots, t_7$) required for an excitation current jump is consistent with the time at which the SP changes from a steady increase to decrease. The excitation current is immediately reduced by 40 to 60 mA after the water injection and is suddenly increased again at the beginning of the second water injection.

Table 2

Average Time for Excitation Current Jump at Each Electrode

Electrode	Average Time	Average Time Required for Excitation Current Jump (min)
1	t_1	3
2	t_2	5
3	t_3	26
4	t_4	43
5	t_5	71
6	t_6	105
7	t_7	136

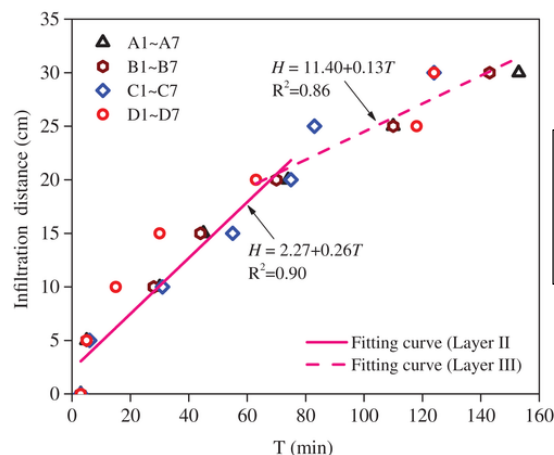


Figure 5: Infiltration distance of water with time.

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

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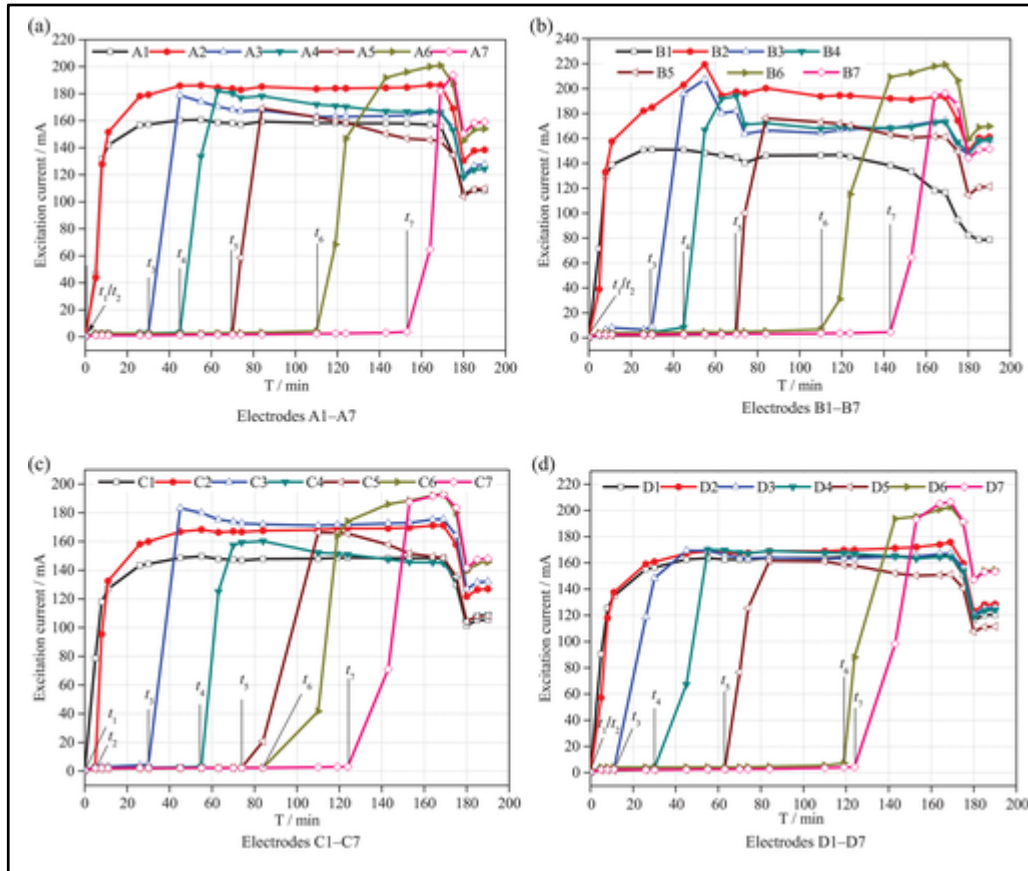


Figure 4: Variations in excitation current with time: (a) Electrodes A1 to A7, (b) Electrodes B1 to B7, (c) Electrodes C1 to C7, and (d) Electrodes D1 to D7.

Apparent Resistivity Response

AGI EarthImager3D is a software that is used to model resistivity in three-dimensional space and induced polarization inversion. The inversion is the mapping of data back to model space and reconstruction of the subsurface resistivity distribution from the measurement data. During the infiltration process, the apparent resistivity profiles between A1 (B1) and A7 (B7) at different times are shown in Figure 6. The origin is in the upper left corner of the model with the horizontal axis increasing to the right, and the vertical axis, z , positive upward. The infiltration reaches A1, or B1, C1 and D1, which are parallel electrodes, 3 min after water injection and the apparent resistivity is reduced to less than $400 \Omega\text{m}$ accordingly (Figure 6). The infiltration reaches A3 at 30 min. Similarly, the infiltration reaches A5 at 74 min and A7 at 153 min after the injection of water. After that, the model is saturated, and the apparent resistivity is less than $400 \Omega\text{m}$.

To quantitatively examine the pattern of changes in the apparent resistivity at different locations, four different locations (labeled as C, D, E, and F in Figure 1) are examined on the apparent resistivity profile. Figure 7 shows the changes in apparent resistivity with time, which can be observed as three stages: the stage where there is a gradual reduction in the apparent resistivity followed by a rapid reduction and finally, the apparent resistivity is constant.

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

Discussion

In the natural environment, groundwater moves throughout the pores of rock and soil, which results in transfers of charges and changes in spontaneous and electrochemical potentials. Therefore, the infiltration can be detected by using the SP, excitation current and electrical resistivity (Yang et al. 2018). The SPs in soil, clay, or rock can be examined through diffusion, electro-filtration and oxidation-reduction potentials (Sun et al. 2014). Diffusion potential is the measure of the ion exchange of two solutions with different ion concentrations. Electro-filtration potential is caused by the coupling of fluid flow and electrical conduction. Finally, during a chemical reaction, oxidation-reduction potential is the ability of water to accept or release electrons and measured by using electrodes (Yang et al. 2018). In this study, the SP is predominantly examined by measuring the electro-filtration potential. When water infiltrates the layer of sand and clay mixture in the experiment, the concentration of positive and negative ions is redistributed along the flowpath due to their selective adsorption characteristics, which depend on the rock and soil particles, thereby creating a potential difference in the direction of flow (Yang et al. 2018). The potential difference is gradually reduced with adsorption until an equilibrium state is reached (Li 2005). The potential difference can be calculated by using (Kong 1999):

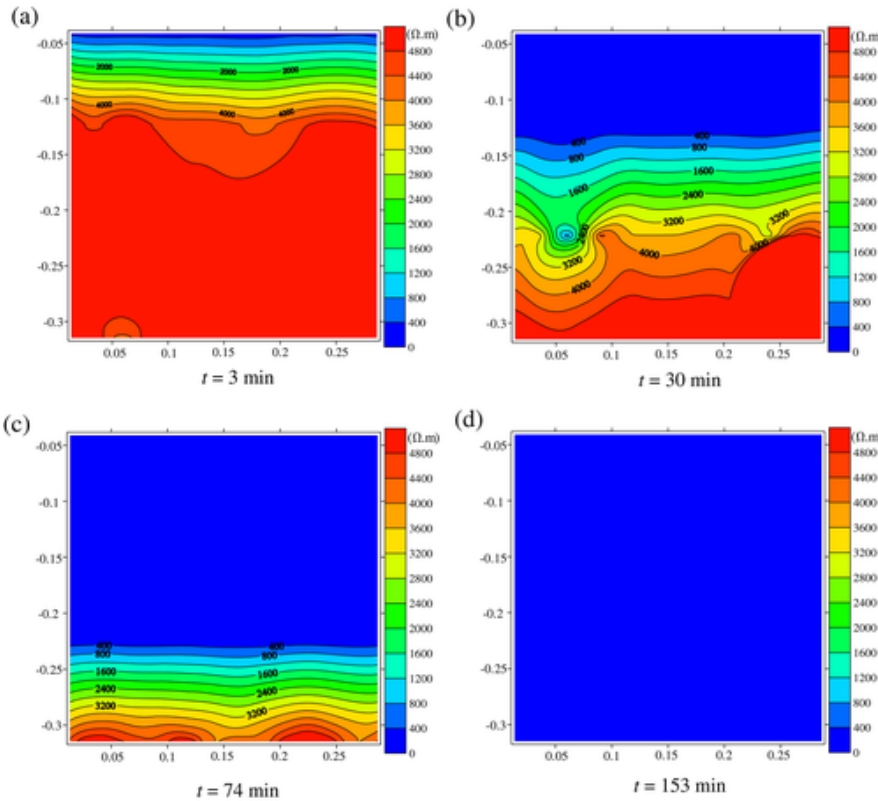


Figure 6: Apparent resistivity profiles at different times during water infiltration (model of length: m, model of apparent resistivity: $\Omega \cdot m$): (a) $t = 3$ min, (b) $t = 30$ min, (c) $t = 74$ min, and (d) $t = 153$ min.

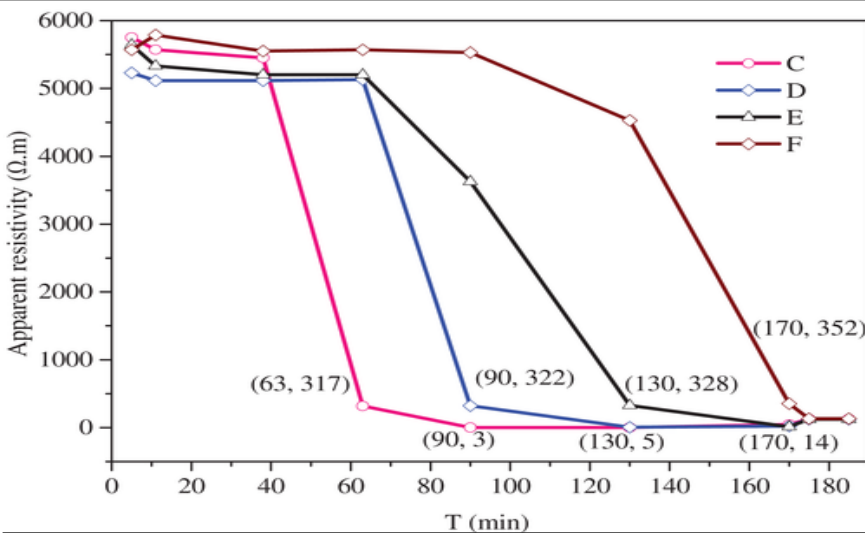


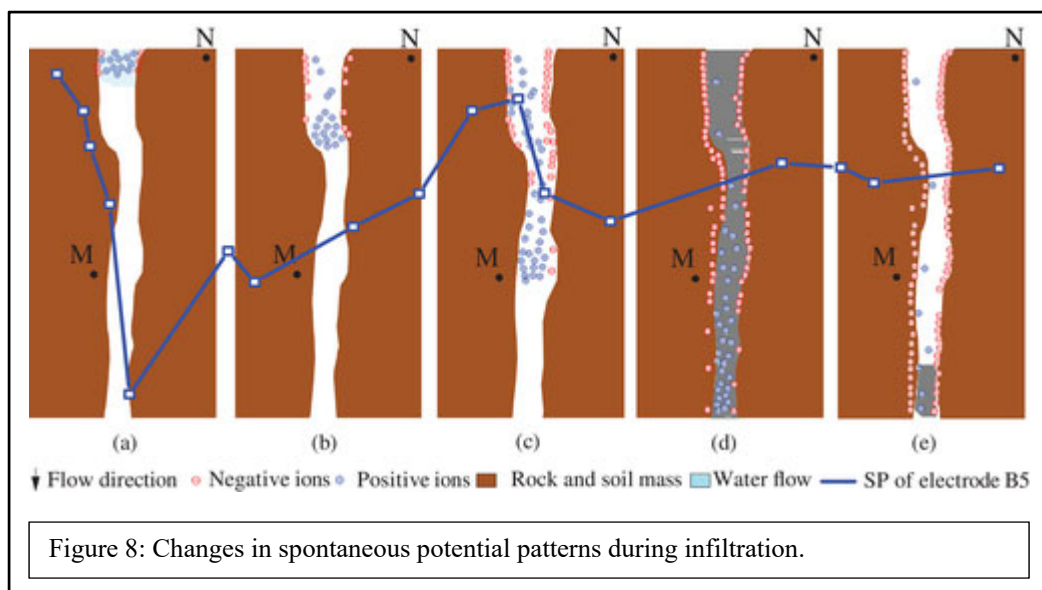
Figure 7: Changes in apparent resistivity with time at different locations.

Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

$$E_f = K_f \frac{\Delta P \cdot R_{mf}}{\mu} \quad (2)$$

where R_{mf} and μ are the resistivity ($\Omega \cdot \text{m}$) and viscosity ($\text{Pa} \cdot \text{s}$) of the filtrate, respectively; ΔP is the pressure difference between the filtrate and porous medium (10^5 Pa); and K_f is the coefficient of the electromotive force (mV) from the electrofiltration, which is related to the chemical composition and concentration of the filtrate. Figure 8 conceptually shows the changes in the SP patterns during water infiltration, in which M is the measurement point of SP and N is the reference point. When water begins to infiltrate into the porous medium, the negative ions in the water are absorbed by the porous medium and the positive ions accumulate in the flowpath. Therefore, the potential difference between M and N is decreased rapidly as shown in Figure 8a. With continued infiltration of water, the U_{MN} is increased. By the time the water reaches M , the number of positive ions accumulated at M is the highest and U_{MN} also reaches its peak, as shown in Figure 8c. When the water passes M , the positive ions move with the water while the negative ions are absorbed by the soil particles at the same time. As a result, the U_{MN} is gradually reduced. Therefore, the SP (absolute value) is also gradually reduced (see Figure 3). Finally, the positive and negative ions in this area reach a dynamic equilibrium state and the U_{MN} is almost constant as water flows away from M or N (Figure 8e). In the experiment, the SP is almost constant at -200 mV after 150 min. In the early stages of infiltration, the number of positive ions at the reference electrode N rapidly increases when the water reaches N . Therefore, the measured SP of each electrode reaches extreme values (from -200 to -550 mV). Similar values of the SP were found during the second round of water injection. When the water reached the electrode, the SP is reduced by 100 to 200 mV due to the increase in the number of negative ions that are adsorbed by the soil particles.



Real-Time Geoelectric Monitoring of Seepage into Sand and Clay Layer

(Continued)

The excitation current increases from 0 to 140 to 220 mA when the water reaches the electrodes. The intensity of the excitation current is related to the measured resistivity of the porous medium and the distance between the electrode and power supply anode B. The excitation current is immediately reduced with the termination of the water injection but suddenly increases again with the commencement of the second round of water injection due to the sensitivity of the excitation current to the water. The excitation current could be used to accurately determine the time for the water to reach the electrodes and the seepage velocity in the porous medium. A comparison of the hydraulic conductivity in Table 1 shows that the seepage velocity is relatively high due to the effect of the artificial electric field (Fang et al. 2012). In the experiment, the dry clay and sand mixture have a much higher apparent resistivity than water. It can be therefore concluded that the apparent resistivity is gradually reduced initially and then rapidly reduced to less than $400 \Omega \cdot \text{m}$ when the infiltration front reaches the electrodes. When the electrodes are completely immersed in water, the apparent resistivity at different locations is reduced to less than $100 \Omega \cdot \text{m}$. Subsequently, the apparent resistivity is almost constant until saturation. In general, the apparent resistivity can be used to determine the position of the infiltration front. Therefore, the geoelectric method in this study monitors real-time flow. The method has the potential for application to a system that can signal water infiltration because it effectively increases the accuracy and timeliness of predicting water infiltration incidents.

Conclusions

This study aimed to investigate the pattern of changes in groundwater seepage using both natural and artificial electric fields in a physical model. The infiltration front and seepage velocity of groundwater can be determined based on the response of the SP, excitation currents and apparent resistivity during the water infiltration process. The SP is reduced by 100 to 200 mV when the water reaches an electrode. During the second round of water injection, the measured SP of all the electrodes recovers to the previous extreme values that range from -200 to -550 mV, thus indicating a “memory” effect. The excitation current increases to the peak value of 140 to 220 mA when the water reaches the electrodes and is stable afterward. The seepage velocity can be calculated by using the time lapsed between the increases in excitation current of the adjacent electrodes. The apparent resistivity is reduced to less than $400 \Omega \cdot \text{m}$ when the water reaches the electrodes. The geoelectric method proposed in this study can be applied to monitor groundwater seepage in rock and soil slopes, building foundations, tunnel engineering, and dam foundation engineering to detect potential water infiltration incidents. Future work will need to focus on the long-term monitoring of water flow in engineering projects with more data.

Read more about this article at: <https://ngwa.onlinelibrary.wiley.com/doi/10.1111/gwmr.12352>

A New Method for Dating Ancient Earthquakes

Constraining the history of earthquakes produced by bedrock fracturing is important for predicting seismic activity and plate tectonic evolution. In a new study published in the Nature journal *Scientific Reports* Jan 17, 2020, a team of researchers presents a new microscale technique to determine the age of crystals grown during repeated activation of natural rock fractures over a time range of billions of years.

The dramatic energy release of an earthquake forms as bedrock segments move in relative opposite directions to each other due to the collision or spreading of the tectonic plates that makes up the Earth's crust. The movement occurs along fault planes where new mineral crystals grow simultaneously.

A New Method for Dating Ancient Earthquakes (Continued)

The bedrock of Scandinavia, up to two billion years old, displays an extensive network of fractures formed at different episodes stretching from the early history of the Scandinavian crust to modern times. In rock samples retrieved from deep boreholes in Sweden, new microscale radioisotopic dating of individual fault crystals reveals the dominant fracturing episodes affecting Scandinavia. Mikael Tillberg, a doctoral student at the Linnaeus University, Sweden, and first author of the paper, explains, "The ages of our analysed crystals matches several distinct periods of extensive mountain range formation when plate boundaries were directly neighboring Scandinavia. These temporal constraints demonstrate that our newly developed approach is suitable to untangle complex fracturing histories."

Thomas Zack, of Gothenburg University, Sweden, and a co-author of the study, describes how the dating method works. "Specific minerals contain radiogenic elements where certain isotopes decay over time. The abundances of these isotopes in tiny crystals formed on fracture surfaces are measured with high precision and detailed spatial resolution." "The link between crystal growth and the frictional movement of earthquakes is ensured by identifying striation lines formed on fracture surface crystals by the movement. This microscopic investigation precedes age analysis to enable a simple and robust procedure for dating of faulting," Henrik Drake at Linnaeus University, also a co-author, adds. Mikael Tillberg summarizes on the significance and possible future applications of this technique: "Repeated earthquake episodes produce a chaotic array of broken rock and mineral growth even in a single crystal or on a particular fracture surface. Our methodology can resolve these sequences and connect the microscale mechanisms involved in fracturing to continent-wide plate tectonic forces. This allows reconstruction of geological models for diverse applications such as seismicity and infrastructure engineering."

Read more about this article at: phys.org/news/2020-01-method-dating-ancient-earthquakes.html

A New Method for Dating Ancient Earthquakes (Continued)



January 9th, the Fernbank Museum hosted Fernbank After Dark with the evening's theme - On the Rocks. Doug John, our Fernbank liaison, asked if we would have a booth/display as part of the evening's entertainment. Given our long relationship with Fernbank, this was a good activity for the AGS; talk a bit about geology and the AGS and help the Museum. But what about a topic given the likely non-geologic background of the event's attendees? What about geologic time and a display of fossils through time? Geologic time is a very basic concept that most folks don't perceive how long it is. And having a hands-on display of fossils with some informed presenters would help folks connect to the concepts. Fortunately for the AGS, we have a fine young paleontologist as a member; Cam Muskelly. As it turns out, between Cam and Ben Bentkowski, they had fossils and rocks from every geologic age, from recent shell hash from Dade County, FL, to a piece of the Acosta gneiss from Australia coming in a 4.03 billion years old. The display consisted of a layout of fossils in chronological order, a little shrine to James Hutton and a small handout of a geologic timescale with the AGS logo and information. Over the course of three hours, we were busy nonstop, talking to an estimated 250 people. It turned out great. Cam and Ben got to talk about geology for 3 hours. Many curious people got to ask their questions and relate their own experiences. And Fernbank had another great event.

January 2020 Atlanta Geological Society PG Cansisate Workshop

Date: Saturday, **January, 2020**
Time: 10:00 am to 12:00 pm
Venue: Fernbank Science Center
(check-in with the receptionist for the specific classroom location)
156 Heaton Park Drive, N.E.
Atlanta, GA 30307
678-874-7102
<http://fsc.fernbank.edu/>

Speaker: Jim Kennedy, PhD, PG

Subject: **Geohydro & Contaminate Transport plus Engineering Geology**

Jim is the Georgia State Geologist and holds B.S. and M.S. degrees in physics and geophysical sciences from the Georgia Tech and a Ph.D. in geology from Texas A&M where he did research on reclaimed lignite mines. As State Geologist, he has worked on the Coastal Sound Science Initiative to manage salt-water intrusion into the Upper Floridan aquifer, permitting of coastal groundwater supply wells, and the State Water Plan. He also has provided expert testimony at the Office of State Administrative Hearings in support of landfill, quarry, and water withdrawal permits issued by the Georgia Environmental Protection Division. Prior to joining the EPD, Dr. Kennedy worked as a consultant and conducted engineering geology, groundwater supply, and environmental remediation projects in various areas of the United States and Europe.

Please join us and forward this message to anyone interested in becoming a Georgia Registered Professional Geologist, or anyone who might be interested in the topic. Two Professional Development Hours are available for attendees of the class. The classes are open to all, membership in the AGS is not required; if you would like to join an application form is attached.

For more information on becoming a member, visit www.atlantageologicalsociety.org. Contact us at the addresses below if you have questions about the workshop or the exams.

Atlanta Geological Society
Professional Registration/Career Development Committee
Ken Simonton, P. G., kws876@gmail.com
Ginny Mauldin-Kenney, ginny.mauldin@gmail.com

Calendar of Events

In addition to a wide array of permanent and special exhibitions and movies on the 4-story giant screen, Fernbank Museum also hosts a variety of special events and programs, as well as drop-in programs offered on weekends and school holidays.

Pirate Day

Saturday, January 25, 2020 10:00 AM

Prepare to weigh anchor and set sail on a swashbuckling adventure with pirate-themed crafts and games.

Tadpole Tales: Pirate Dinosaurs

Saturday, January 25, 2020 11:00 AM

Wee mateys will enjoy a story with a Fernbank crew member, who shall also lead a special activity or song, featuring Captain Flinn and the Pirate Dinosaurs by Giles Andreae and Russell Ayto. Recommended for ages 1–6

Live Animal Encounter

Saturday, January 25, 2020 2:00 PM

Get up close and personal with a member of Fernbank's animal collection. Recommended for ages 4+

Cosmic Quest

Saturday, January 25, 2020 3:00 PM

Blast off for an exploration of space that will take you from Earth through the universe. Recommended for ages 8+

Tadpole Tales

Sunday, January 26, 2020 11:00 AM

Preschoolers will enjoy a story and special activity with a Fernbank educator. Recommended for ages 1–6

Live Animal Encounter

Sunday, January 26, 2020 12:00 PM

Get up close and personal with a member of Fernbank's animal collection. Recommended for ages 4+

Ranger Explorations

Sunday, January 26, 2020 1:30 PM

Join a Fernbank Ranger for fun and informative outdoor activities in WildWoods. Recommended for ages 6+

Cosmic Quest

Sunday, January 26, 2020 3:00 PM

Blast off for an exploration of space that will take you from Earth through the universe. Recommended for ages 8+

Little Lessons

Monday, January 27, 2020 11:30 AM

Our littlest visitors will enjoy a special story time with a Fernbank storyteller. Recommended for ages 1–4

Geology Rocks

Monday, January 27, 2020 2:30 PM

Discover how the rocks of Georgia formed with hands-on experiments and touchable specimens. Recommended for ages 8+

Folk Tales

Tuesday, January 28, 2020 11:30 AM

Hear a story for young visitors from cultures around the world, with a fun activity. Recommended for ages 1–12

Discover Zoology

Tuesday, January 28, 2020 2:30 PM

Explore the different ways animals survive and thrive as you solve a science mystery. Recommended for ages 8+

Calendar of Events

Little Lessons

Wednesday, January 29, 2020 11:30 AM

Our littlest visitors will enjoy a special story time with a Fernbank storyteller. Recommended for ages 1 – 4

Live Animal Encounter

Wednesday, January 29, 2020 1:00 PM

Get up close and personal with a member of Fernbank's animal collection. Recommended for ages 4+

Incredible Weather

Wednesday, January 29, 2020 2:30 PM

Delve into the dynamic world of weather, climate, and atmospheric science with live experiments and demonstrations. Recommended for ages 8 – 15

Tadpole Tales

Thursday, January 30, 2020 11:30 AM

Preschoolers will enjoy a story and special activity with a Fernbank educator. Recommended for ages 1 – 6

Geology Rocks

Thursday, January 30, 2020 2:30 PM

Discover how the rocks of Georgia formed with hands-on experiments and touchable specimens. Recommended for ages 8+

Folk Tales

Friday, January 31, 2020 11:30 AM

Hear a story for young visitors from cultures around the world, with a fun activity. Recommended for ages 1 – 12

Discover Zoology

Friday, January 31, 2020 2:30 PM

Explore the different ways animals survive and thrive as you solve a science mystery. Recommended for ages 8+

Our Senses Opening Day

Saturday, February 8, 2020 10:00 AM

Enjoy hands-on activities and games that'll engage your senses.

Fernbank After Dark: It's Electric

Friday, February 14, 2020 7:00 PM

Sparks will fly during an electrifying evening of hands-on science.

Fernbank Forest Bird Walk

Sunday, February 16, 2020 8:00 AM

Learn more about the feathered inhabitants of Fernbank Forest during a guided walk.

Latin Dance Night

Friday, February 28, 2020 7:00 PM

Spice things up with sassy Latin rhythms as you dance the night away to Salsa, Bachata, Cumbia and Merengue.

Fernbank Forest Winter Tree ID Tour

Saturday, February 29, 2020 9:00 AM

Learn how to identify trees even when their leaves are fallen.



FERNBANK MUSEUM

of NATURAL HISTORY



WILDWOODS AND FERNBANK FOREST

NOW OPEN
Members Free!



A New Way to Museum

An elevated walkway takes you from the museum's terrace into the treetops where you can enjoy stunning views from one of 2 tree pods. From there, guests can tailor their experiences:

- You'll discover something new each time you visit. You don't have to drive north to enjoy the fall colors. [Fernbank Forest](#) and WildWoods come alive with dramatic yellows and oranges as the leaves change.
- Explore immersive and interactive exhibits for children (Nature Stories) and tweens (Adventure Outpost).
- See adventure turn to art in the [Nature Gallery](#), featuring permanent installations and rotating special exhibits.
- Get a closer look at nature through special trailside experiences, including a sensory wall, animal tracks and tree molds.



Our Senses: An Immersive Experience

On view February 8 through May 3, 2020

Explore this immersive exhibit that challenges and engages the senses and discover how our brains shape the way we perceive the world. Delve into advancements in engineering and scientific research that allow humans to extend our senses with technology.



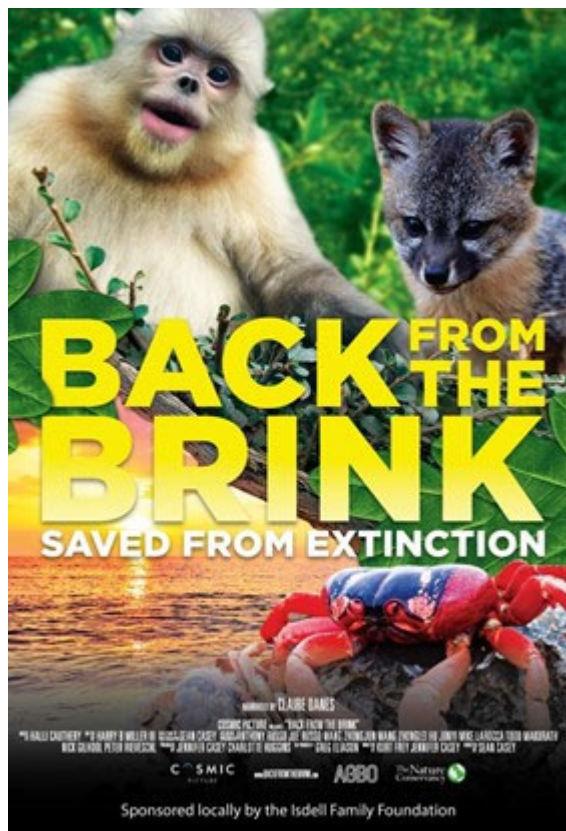
FERNBANK MUSEUM

of NATURAL HISTORY

Fernbank Museum of Natural History

(All programs require reservations, including free programs)

Now showing in the Fernbank movie theater:



Back from the Brink: Saved from Extinction 3D

Showing January 17 through April 30, 2020

Experience the remarkable true story of three animal species rescued from the brink of extinction. Join this amazing adventure alongside scientists, researchers, students and park rangers to save California's enchanting Channel Island Fox, China's fabled Golden Monkey and Christmas Island's wondrous migrating crabs. *Run time: 40 minutes | Assisted listening devices and closed captions are available upon request.*

Pandas 3D

Showing November 1 through February 13, 2020

Journey across the globe to the Chengdu Panda Base China where researchers are working to introduce giant panda cub Qian Qian into the wild for the first time. Kristen Bell narrates Qian Qian's journey through the Sichuan mountains as the cub takes her first steps outside her protected habitat, discovering her true animal nature—even as she faces the challenges of the unknown. *Run time: 40 minutes | Assisted listening devices and closed captions are available upon request.*

AGS Officers

President: Ben Bentkowski

bbentkow@gmail.com

Phone (770) 296-2529

Vice-President: Steven Stokowski

sstokowski@yahoo.com

Secretary: Rob White

Phone (770) 891-0519

robeth@bellsouth.net

Treasurer: John Salvino, P.G.

Phone: 678-237-7329

johnsalvino@bellsouth.net

Past President

Shannon Star George

shannonsgeorge@gmail.com

AGS 2020 Meeting Dates

Listed below are the planned meeting dates for 2020. Please mark your calendar and make plans to attend.

2020 Meeting Schedule

February	25
March	31
April	28
May	26
June	30

2020 PG Study Group Meetings

February	29
March	28
April	25
May	30
June	27

AGS Committees

AGS Publications: Open

Career Networking/Advertising: Todd Roach

Phone (770) 242-9040, Fax (770) 242-8388

tdr@piedmontdrilling.com

Continuing Education: Open

Fernbank Liaison: Miranda Gore Shealy

Phone (404) 929-6341

Miranda.Shealy@FernbankMuseum.org

Doug John

Phone (404) 929-6342

Doug.John@FernbankMuseum.org

Georgia PG Registration: Ken Simonton

Phone: 404-825-3439

kws876@gmail.com

Ginny Mauldin-Kenney,

ginny.mauldin@gmail.com

Teacher Grants: Bill Waggener

Phone (404) 354-8752

waggener80@yahoo.com

Hospitality: John Salvino, P.G.

johnsalvino@bellsouth.net

Membership: Burton Dixon

Dixonburton1@gmail.com

Social Media Coordinator: Carina O'Bara

carina_do@yahoo.com

Newsletter Editor: James Ferreira

Phone 508-878-0980

Ferreira.02767@gmail.com

Web Master: Ken Simonton

kws876@gmail.com

www.atlantageologicalsociety.org

ATLANTA GEOLOGICAL SOCIETY

www.atlantageologicalsociety.org

ANNUAL MEMBERSHIP FORM

Please print the required details and check the appropriate membership box.

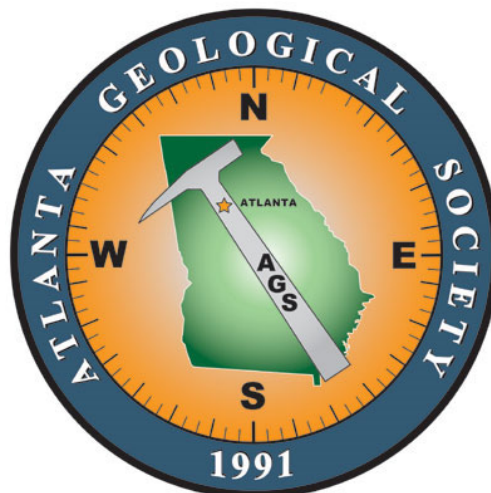
DATE: _____

NAME: _____

ORGANIZATION: _____

TELEPHONE (1): _____ TELEPHONE (2): _____

EMAIL (1): _____ EMAIL (2): _____



STUDENT \$10 ☐

PROFESSIONAL MEMBERSHIP \$25 ☐

CORPORATE MEMBERSHIP \$100 ☐

(Includes 4 professional members, please list names and emails below)

NAME: _____ EMAIL: _____

NAME: _____ EMAIL: _____

NAME: _____ EMAIL: _____

NAME: _____ EMAIL: _____

For further details, contact the AGS Treasurer:
John Salvino johnsalvino@bellsouth.net

Please make checks payable to the “Atlanta Geological Society” and bring them to the next meeting or remit with the completed form to:

Atlanta Geological Society, Attn: John Salvino
3073 Lexington Avenue
Woodstock, Georgia 30189

To pay electronically; click
<https://squareup.com/store/atlanta-geological-society>

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