New Jersey Department of Environmental Protection

Site Remediation Program

Ground Water Technical Guidance:
Site Investigation
Remedial Investigation
Remedial Action Performance Monitoring

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Ground Water Technical Guidance

Intended Use of Guidance Document

This guidance is designed to help the person responsible for conducting remediation comply with the New Jersey Department of Environmental Protection (Department) requirements established by the Technical Requirements for Site Remediation (Technical Rules), N.J.A.C. 7:26E. This guidance will be used by many different people involved in the remediation of a contaminated site; such as Licensed Site Remediation Professionals (LSRP), Non-LSRP environmental consultants and other environmental professionals. Therefore, the generic term "investigator" will be used to refer to any person that uses this guidance to remediate a contaminated site on behalf of a remediating party, including the remediating party itself.

The procedures for a person to vary from the technical requirements in regulation are outlined in the Technical Rules at N.J.A.C. 7:26E-1.7. Variances from a technical requirement or departure from guidance must be documented and adequately supported with data or other information. In applying technical guidance, the Department recognizes that professional judgment may result in a range of interpretations on the application of the guidance to site conditions.

This guidance supersedes previous Department's guidance issued on this topic at N.J.S.A. 26:10C-16.

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1.0 Purpose

The Site Remediation Reform Act (SRRA), N.J.S.A. 58:10C, which was enacted in May 2009, required the Department to develop new regulations and guidance that provide direction on a number of issues involving the investigation and remediation of contaminated sites.

1.1 Document Overview

This document provides guidance on:

- performing and achieving compliance with the requirements of the Department's Technical Rules with regard to the remediation of ground water
- characterizing hydrogeology and ground water contamination during the site investigation and remedial investigation phases
- evaluating the effectiveness of ground water remedial actions

Some wording of the document is borrowed from concepts from other sources of information. These sources are provided as references. While direct citations are noted, the references should be consulted for the exact language attributed to these publications. Additional information on characterizing, delineating, and remediating ground water can be obtained from a variety of publications, including those listed in the attached bibliography. These publications should be referenced for more detailed information regarding the characterization, delineation and remediation of ground water.

2.0 Site Investigation of Ground Water

2.1 Overview

The purpose of a site investigation is to determine if remediation is necessary because contaminants are present at the site or area of concern (AOC), or because contaminants have emanated or are emanating from the site or AOC, above any of the applicable remediation standards or any criterion.

This section provides guidance on:

- determining when a ground water site investigation is necessary
- biasing of initial ground water sample locations
- background ground water investigations

2.2 Ground Water Remediation Standards

The Remediation Standards (N.J.A.C. 7:26D) establish the Ground Water Quality Standards (GWQS) as the minimum standards for the remediation of contaminated ground water. The GWQS (N.J.A.C. 7:9C), establish the designated uses of the State's ground waters, classify ground waters based on those uses, and specify the water quality criteria and other policies and provisions necessary to attain those designated uses. The Ground Water Quality Criteria (GWQC) are numerical values assigned to each constituent (pollutant) discharged to ground waters of the State. The GWQS also contain technical and general policies to ensure that the designated uses can be adequately protected.

Appendix Table 1 (http://www.nj.gov/dep/wms/bwqsa/Appendix_Table_1.htm) in the GWQS (N.J.A.C. 7:9C) lists specific GWQC and Appendix Table 2 (http://www.nj.gov/dep/wms/bwqsa/Appendix_Table_2.htm) provides the interim generic GWQC for carcinogenic and non-carcinogenic synthetic organic compounds. A table of interim specific criteria developed in accordance with the GWQS is available at http://www.nj.gov/dep/wms/bwqsa/gwqs_interim_criteria_table.htm

2.3 Considerations for When a Site Investigation of Ground Water is Necessary

A site investigation of ground water is required at all AOCs where there is the potential that ground water has been contaminated. When determining if there is a potential that ground water has been contaminated the investigator should consider the following:

- potential receptors may have been impacted by contaminants at and/or emanating from the site (i.e., potable wells, occupied structures, or surface water);

- free or residual product is detected;

- the potential discharge is close to or beneath the water table. For example, when investigating an underground storage tank (UST), leach field, seepage pit or where the AOC is classified as an Underground Injection Control (UIC) unit regulated under N.J.A.C. 7:14A-8;

- the potentially discharged contaminants are mobile. For example, individual contaminants with a solubility that is greater than 100 mg/L or mixtures of contaminants that consist of mobile constituents, such as gasoline;

- contamination is detected in soil within two feet of the water table or bedrock;

- the soil at an AOC has a relatively high permeability, or little sorptive capacity; or

- contamination has had time to migrate through the unsaturated zone to the water table based on contaminant and soil transport properties and the estimated date of discharge.

Where ground water is not sampled during the investigation of a potentially contaminated AOC, and a ground water investigation may be warranted based on the above considerations or other site specific data, the technical justification for not performing a ground water site investigation should be provided in the applicable remedial phase report.

2.4 Cross Contamination of Ground Water

During all phases of the remediation, it is important to take steps to limit the potential for cross contamination. The August 2005, Field Sampling Procedures Manual (http://www.nj.gov/dep/srp/guidance/fspm/pdf/fsmp2005.pdf) discusses methods for limiting cross contamination during sample collection. Boring through confining units should be avoided if the presence of dense, non-aqueous phase liquid (DNAPL) or a downward vertical head between upper and lower aquifer units are suspected. Where it is necessary to drill through a confining unit to complete vertical delineation of ground water contamination, wells should be doubled cased and completed outside of areas where DNAPL is suspected. Initial sets of borings and wells should be installed outside of a known or suspected DNAPL areas to postulate an initial hydrostratigraphic model and impacts of the DNAPL areas on the dissolved plume, before proceeding with characterizing the DNAPL source area itself.

2.5 Ground Water Site Investigation Sampling Locations

During the site investigation, ground water samples must be biased to the suspected location of greatest contamination, both horizontally and vertically to determine if ground water has been contaminated by the AOC.

2.5.1 Horizontal Biasing of Ground Water Samples

Ground water samples should be biased based on soil sample analytical results, contaminant type, AOC history, location of potential discharges (i.e. piping joints, dispenser pans, spill buckets, etc.), field instrument readings, visual observation or other field indicators. The number of ground water samples collected at each AOC should be contingent upon the size of the AOC, contaminant and aquifer properties, soil heterogeneity and ground water flow direction.

Where ground water samples cannot be collected at the location of greatest suspected contamination, they should be collected as close to the AOC as practical and in a location that is hydraulically downgradient. Alternatively, multiple ground water samples may be collected surrounding the AOC and as close to it as practical.

Ground water flow direction can often be predicted based on topographic relief, the location of surface water bodies, structural controls in the bedrock or soils, location of pumping wells and subsurface conduits at or below the water table. Ground water flow direction may also be predicted based on data from adjacent sites. If ground water flow direction cannot be determined as stated above, it may be determined by collecting site specific potentiometric surface data from surveyed temporary wells, piezometers or monitoring wells prior to collecting ground water samples.

2.5.2 Vertical Biasing of Ground Water Sample Locations for LNAPL Contaminants

When investigating contaminants that are less dense than water, continuous soil cores should be completed and screened through the water table to a sufficient depth to account for water table fluctuation and the possibility of contamination trapped beneath the water table. Initial ground water samples should be collected at the depth exhibiting the greatest contamination as identified during field screening. For AOCs located within bedrock, it is recommended to screen the upper water bearing bedrock zone.

2.5.3 Vertical Biasing of Ground Water Sample Locations for DNAPL Contaminants

When investigating contaminants that have a density greater than water, vertical soil and ground water contaminant concentration profiles should be completed to determine if free or residual DNAPL is present. Continuous soil cores should be evaluated to the depth at which any of the following are first encountered:

- ground water contamination;
- the first low permeability soil layer located beneath the water table; or
- the top of bedrock if located beneath the water table.

Ground water samples should be collected at the depth at which any of the following encountered:

- the greatest reading on field instrumentation;
- the top of the first low permeability layer;
- the top of bedrock; or
- the water table within bedrock, if the water table is not present in overburden.

For dense non-volatile contaminants, ground water samples should be collected at any indication of contamination (such as staining) or the top of the first low permeability layer or top of bedrock. Where there is no indication of free or residual product, ground water samples should be collected along a vertical profile so that dissolved contamination may be detected.

2.5.3 AOC Specific Ground Water Sampling

For discharges that originate at an underground storage tank, leach field, seepage pit or Underground Injection Control (UIC) unit located below the water table, ground water samples should be collected beneath the water table at the depth of the discharge or the depth of greatest contamination as determined by field screening.

For AOCs where point source discharges may have occurred (i.e., below grade piping or floor drains) or where preferential contaminant migration pathways exist (i.e. utility trenches), installing monitoring points closer together in the direction of assumed ground water flow may be warranted.

2.6 Soil Logging and Field Screening

All site related stratigraphic logs should use the same soil classification system. The Department's Field Sampling Procedures Manual (FSPM) lists several acceptable soil classification systems. All borings should be screened for the presence of contamination, including the presence of free and residual product. Logging and screening of soil from borings is necessary to bias initial ground water samples and to assist in developing a conceptual hydrostratigraphic model.

2.7 Ground Water Sampling Methods and Analyses

Ground water samples may be collected as grab samples using temporary wells or other direct push methods using any accepted method in the latest version of the Department's FSPM (http://www.nj.gov/dep/srp/guidance/fspm/pdf/fsmp2005.pdf).

Monitoring wells and temporary points shall be installed, constructed and abandoned in accordance with N.J.A.C. 7:9D (http://www.state.nj.us/dep/watersupply/NJAC7_9D.pdf). Additionally, Appendices 6.1 and 6.2, in Chapter 6.9 of the FSPM detail monitoring well installation and construction and Department specific well specifications for bedrock, unconsolidated and confined aquifers.

Ground water samples should be analyzed for the contaminants that may be present as determined during the preliminary assessment and from other information obtained during the remediation.

2.8 Determination of Off-Site Contribution of Ground Water Contamination

A background ground water quality investigation is necessary to prove that contamination identified in on-site ground water samples is the result of contamination originating from an off-site source or is natural background. This requires a comprehensive understanding of ground water flow relative to the site as a whole and to each AOC. The extent of the background ground water quality investigation needed is determined by the extent of the contamination that is impacting the site, as well as the size and number of AOCs associated with the site. Depending on the circumstances, it may be possible to support a claim of off-site groundwater contamination with existing off-site data, temporary well points or other ground water investigative techniques rather than permanent background wells.

2.8.1 Ground Water Investigation to Demonstrate Off-site Contribution

A sufficient number of background ground water samples should be collected to evaluate ground water quality upgradient from the AOC. In addition, ground water samples should be collected from each water-bearing zone believed to contain contamination originating from an off-site source.

The samples should be collected:

- upgradient and beyond the area of influence of all site related AOCs (off-site, if necessary).
- along the same ground water flow paths that intersect the AOC.

Background and on-site ground water samples should be collected simultaneously for all on-site contaminants believed to be originating from off-site sources. A sufficient number of background samples should be collected to identify seasonal water quality variations and long-term trends. Statistical methods that may be employed to establish background ground water quality are discussed in the March 2009 “Statistical Analysis of Groundwater Monitoring Data at RCRA Facilities” (USEPA, 2009).

2.8.2 Ground Water Remediation Standard Based on Background Contamination

The remediating party is not responsible to clean up background ground water contamination. Several scenarios are outlined below to assist the investigator in determining if remedial action is necessary and, if so, the appropriate ground water remediation standard.

No further ground water remediation is necessary for an AOC if it is documented that:

- the contaminant in question has never been used on the site,
- the contaminant’s parent compound has never been used on the site;
- there is no evidence of any other on-site discharge of the contaminant in question; and
- the contaminant in question is present in the background samples.

Additional remediation may be necessary when ground water contamination is present at an AOC and in the upgradient background ground water. The need for additional remediation should be based upon the extent and magnitude of the off-site contaminant contribution. To evaluate the off-site contaminant contribution, contaminant attenuation rates, contamination degradation rates, contaminant type, and ground water flow velocity, etc., should be considered.

Additional ground water remediation is required when:

- there has been site related discharge of the contaminant; and
- on-site ground water contaminant concentrations are greater than contaminant concentrations migrating from the off-site source.

Regardless of the up-gradient background ground water concentration(s), at a minimum, the following are required:

- The soil contamination resulting from the site related discharge must be remediated to the site specific impact to ground water soil remediation standard;
- any free and residual product related to the discharge from the AOC must be remediated
- all of the on-site contaminant contribution must be remediated.

When a ground water remediation is conducted at an AOC impacted by contamination from an up-gradient off-site source(s), the ground water remediation standard at the impacted AOC is the higher of:

- the GWQS for the contaminant; or
- the background concentration of the contaminant.

When a ground water remedial action is ongoing and background ground water concentrations are the ground water remediation goal, then the background contaminant concentrations entering the site should be re-evaluated at some frequency, such as annually. Statistical methods may be employed to determine if there are trends in background contaminant concentrations.

2.9 Determination of Natural Background

Natural background ground water contamination may be demonstrated through an evaluation of the distribution of contaminants. Contamination should be ubiquitous across the site with concentrations no greater at potential source areas than at locations up or side gradient of potential source areas. A sufficient number of ground water samples should be collected across the site and upgradient from the site to demonstrate natural background contamination.

3.0 Remedial Investigation of Ground Water

The ground water remedial investigation entails characterizing site hydrostratigraphy, delineation of ground water contamination and the sources of ground water contamination. The first step in completing delineation should be the establishment of a conceptual hydrostratigraphic model, which should be updated with each phase of investigation. The conceptual hydrostratigraphic model is a tool to assist in understanding the site hydrology and stratigraphy and is a portion of the conceptual site model. Please refer to the conceptual site model guidance document for additional direction on developing a conceptual site model.

Where a source of ground water contamination has been identified, rapid delineation and mitigation of this source should be completed. The investigator must identify the need for any interim remedial measures (IRM) necessary to remove, contain, or stabilize a source of ground water contamination to prevent contaminant migration and exposure to receptors in accordance with the Department’s Technical Requirements for Site Remediation.

Delineation of ground water contamination should proceed in a rapid fashion relying on a dynamic workplan, real time screening analytical data and decisions made in the field. When delineation is completed, a ground water performance monitoring network must be designed and installed and a **Classification Exception Area (**CEA) must be established.

3.1. Overview

The Technical Rules require a remedial investigation of ground water when the concentration of any contaminant exceeds the Ground Water Remediation Standards. The purpose of the ground water remedial investigation is to:

- Identify and characterize the migration pathways of contamination in ground water at a contaminated site.
- Delineate the horizontal and vertical extent of ground water contamination to the Ground Water Remediation Standard.
- Update the receptor evaluation and determine if an immediate environmental concern or vapor concern exists.
- Identify the need for any IRM necessary to remove, contain, or stabilize a source of contamination to prevent contaminant migration and exposure to receptors.

A remedial investigation of ground water shall be conducted by:

- Characterizing the hydrogeology of the site. The guidance on characterizing hydrogeology in unconsolidated and bedrock formations is presented in sections 3.3 and 3.4, respectively and provides guidance on:

  - determining regional geology, hydrogeology, and related stratigraphic controls, such as low and high hydraulic conductivity strata that may influence contaminant migration; determining the regional direction of ground water flow in each potentially impacted aquifer or water bearing unit; identify ground water recharge and discharge zones

  - determining the site-specific depth to ground water and local flow direction; characterize perched ground water if present; determining if the water table may have been historically depressed or lowered due to pumping influences or natural conditions (see API, 2006).

  - determining hydraulic properties of all impacted water bearing units as well as intervening aquitard units as components of a conceptual site hydrostratigraphic model

  - determining contaminant migration pathways, taking into consideration the location of any anthropogenic or natural preferential pathways above and below the water table that may influence contaminant migration

- Determining and characterizing ground water contaminant source zone(s) and their extent above and below the water table. The guidance for identifying and delineating free and residual product is included in 3.2 and includes guidance on:

  - delineating the extent of free product and residual product

  - delineating the extent of soil contamination in the unsaturated zone that exceeds the IGWSRS (spell out)

  - delineating the extent of other sources of ground water contamination below the water table

- Determining if receptors may be impacted by performing a receptor evaluation. A form for completing a receptor evaluation is available from the Department’s website at [http://www.nj.gov/dep/srp/srra/forms/.](http://www.nj.gov/dep/srp/srra/forms/)

- Determining and characterizing contaminant type and distribution by completing the vertical and horizontal delineation of ground water contamination to the Ground Water Remediation Standards.

- Determining impacts to and effects of local surface water features.

- Assessing background ground water quality, if impact from an off-site source is claimed. Guidance on determining background ground water quality is provided in section 2.7 of this document.

- Developing and implementing a ground water monitoring program that will effectively monitor the ground water contaminant plume. Section 4.0 of this document provides guidance on establishing a ground water monitoring program.

- Determining contaminant fate and transport. Guidance on determining fate and transport is provided in section 3.3 and 3.4 of this document.

- Proposing a ground water CEA as part of the remedial investigation report. Guidance on establishing a CEA can be found on the Department's website at http://www.state.nj.us/dep/srp/guidance/cea/cea_guide.htm.

3.1.2. The Triad Approach

To minimize costs and the time necessary to conduct a remedial investigation, the LSRP may implement the Triad approach. The Triad approach (http://www.triadcentral.org) is a process that integrates systematic planning, dynamic work plans, and real-time measurements to achieve more timely and cost effective site characterization and cleanup. The Triad approach seeks to recognize and manage the uncertainties involved in generating representative data from heterogeneous environmental matrices. A key output of the Triad approach is the development of an accurate conceptual site model.

3.1.3. Conceptual Hydrostratigraphic Model

The first step in the investigative process is to gather data to develop an initial conceptual hydrostratigraphic model. The conceptual hydrostratigraphic model is a portion of the overall conceptual site model which the investigator should develop for the site. The Department has prepared guidance on the use of conceptual site models for New Jersey contaminated sites, which can be found at http://www.nj.gov/dep/srp/guidance/srra/. The conceptual hydrostratigraphic model provides an early working basis for the implementation of the remedial investigation. Information may be collected prior to performing any invasive fieldwork for the remedial investigation. Data for the initial conceptual hydrostratigraphic model can be obtained from various resources, including by performing a literature review, field reconnaissance and surface geophysics. The model should be updated and modified as new data are acquired and the understanding of contaminant distribution and hydrostratigraphy are refined. Additional investigation needs to be conducted where the model indicates that characterization or delineation is not complete or other data gaps are present.

An initial conceptual hydrostratigraphic model should consist of the following:

- topographic map showing the locations of surface water bodies and any pumping wells

- geologic map

- generalized geologic cross section based on geologic maps and data from nearby sites that have undergone environmental evaluations and local well logs

- an estimated depth to ground water
- an initial estimate of ground water flow direction and gradient based on topography, data from nearby sites, pumping wells, and surface water features
- an estimate of the hydraulic conductivity and effective porosity of the formation based on literature values
- an estimate of the average ground water and contaminant flow velocities
- an estimate of the distance that the plume may have migrated using average ground water and contaminant flow velocities and an estimate of the date of the initial discharge

3.1.3.1 Generic Conceptual Hydrostratigraphic Models for the Unconsolidated Formations and Sediments of NJ

In a majority of contaminated sites in New Jersey, first ground water is found in unconsolidated sediments. Ground water in these unconsolidated sediments frequently acts as a recharge reservoir for both the crystalline and sedimentary bedrock aquifers of New Jersey. Even in an area where it is believed that the water table is deep in competent bedrock, a perched water table, whether present throughout the year, seasonally, or existing for only short periods of time after precipitation events, may be present on low permeability sediments or at the bedrock interface above the regional water table. These unconsolidated sediments may contain significant contaminant mass, therefore, a detailed understanding of the hydrogeology is critical.

The ground water flow direction within these unconsolidated sediments always migrates from areas of high hydraulic head to areas of low hydraulic head or from areas of recharge to areas of discharge. The velocity at which ground water will flow is directly proportional to its hydraulic gradient and the hydraulic conductivity, and indirectly proportional to the effective porosity of the media. However, due to a number of physical and chemical factors, dissolved contamination present in ground water will typically migrate at a slower speed than ground water.

Generic hydrostratigraphic models are provided below for the coastal plain sediments, which cover roughly half of New Jersey; glacial sediments, which cover most of northern New Jersey; and unconsolidated sediments overlying bedrock.

3.1.3.1.1 Generic Hydrostratigraphic Model for Coastal Plain Sediments

The New Jersey Coastal Plain covers roughly the southern half of the state. It consists of unconsolidated deposits ranging in age from the Early Cretaceous into the Miocene. The northern boundary of the coastal plain extends to a line drawn roughly between Trenton and Carteret. The boundary is often referred to as the “fall line” due to the presence of waterfalls and rapids between the older sedimentary and igneous rocks of the Newark Basin in the Piedmont physiographic province to the north and the Coastal Plain physiographic province to the south.

Deposition of coastal plain sediments began at a time of major sea level rise during the Cretaceous Period. The majority of sediments of the coastal plain consist of sand, silt, and clay deposited in deltaic and marine environments, based in large part by fluctuations of sea level. These deposits dip towards the coast. They have a feather edge towards the northwest and thicken towards the southeast. Near Atlantic City, the coastal plain sediments are roughly 4,500 feet thick. These deltaic and marine deposits are overlain in areas by a thin blanket of younger river deposited sands and gravels.

The deposits of the coastal plain have been divided into numerous formations, some of which are very prolific water producers and others, with lower conductivity values, act as confining or semi-confining units. Because of the dip of the coastal plain deposits, confined aquifer units located in the southeast are typically recharged in outcrop areas of the aquifer to the northwest. Due to heavy aquifer utilization in urban areas, the natural ground water flow direction is often impacted by pumping wells. For example, the Delaware River is a natural ground water discharge area, but in some areas, such as locations near Camden, ground water flow is away from the river toward pumping centers.

3.1.3.1.2 Generic Hydrostratigraphic Model for Glacial Sediments

Glacial sediments in New Jersey are the result of at least three glaciations that occurred during the Pleistocene Epoch and overlie roughly 30 percent of the state. Most of the glacial sediments lie north of an area from Phillipsburg, to Buttzville, to Denville, to Perth Amboy. Generally, the only place where glacial sediments are not present north of this line is where bedrock is exposed. South of this line, scattered deposits exist in flat upland areas and in valleys. The area between Plainfield, New Jersey and Riegelsville, Pennsylvania, roughly marks the maximum southern extent of the glacial deposits.

The sediment grain size associated with the glacial deposits range from clay to boulder. Some of the deposits are unsorted and rich in fine-grained sediments and do not produce much water, while sand and gravel deposits in buried valleys can be extremely productive and used as a water supply. Till and lacustrine lake bottom deposits may act as confining or semi-confining units in some areas for productive fluvial and lacustrine fan deposits below. The thickness, type, and extent of the glacial sediments vary from place to place and with sediment size up to boulders, investigation within these units can be challenging.

3.1.3.1.3 Generic Hydrostratigraphic Model for Unconsolidated Sediments Overlying Bedrock

Unconsolidated sediments are frequently found overlying bedrock as overburden. These deposits consist of alluvium, colluvium, artificial fill, and regolith (Herman 2010). Depending on the depth of the regional water table, ground water may be present in these unconsolidated sediments year round or depending on water table fluctuations, ground water in the overburden may be a seasonal occurrence. However, even if the water table is found deep in competent bedrock, a perched water table will likely be present. This is because the hydraulic conductivity of the overburden is frequently greater than the underlying bedrock. As noted by Nielsen (2006) and Sara (2003), if the hydraulic conductivity in the underlying unit is three or more orders of magnitude greater than the overlying unit, perched water will almost always have the potential to be present.

Additionally, a transitional weathered bedrock zone frequently overlies competent bedrock. This occurs because physical and chemical weathering of the competent bedrock will cause development of additional porosity within the rock, to where the rock is no longer competent and may be referred to as weathered bedrock. As stated by Herman (2001), conductive features within weathered bedrock include partially dissolved tectonic fractures, stratigraphic zones of mineral dissolution, bed-parallel mechanical layering, and fractures resulting from erosion and weathering. As noted by Herman (2010), because of the abundance of connected pathways, ground water in weathered bedrock commonly behaves as an unconfined aquifer, similar to overburden.

While ground water flow in the overburden and weathered bedrock may generally follow topography, it is also affected by aquifer heterogeneities, including competent bedrock topography. This is caused by differences in hydraulic conductivities in and between the overburden, weathered bedrock, and competent rock. If the competent bedrock is an undulating or irregular surface, it may have a channeling or steering effect on the ground water flow direction. For example, if there is a subsurface bedrock pinnacle that extends above the overburden water table, the dominant ground water flow paths may migrate around the bedrock pinnacle.

Because of their typical higher porosity values, the overburden and weathered bedrock zones typically have higher storage compared to competent bedrock. As a result of its high storage combined with its large lateral extent, the overburden and weathered bedrock zones can act as a dominant source of ground water recharge for competent bedrock. Because most contaminant discharges occur at or near the surface, the overburden and weathered bedrock zones are typically the first to be impacted by contamination. Because of their high storativity values, the overburden and weathered bedrock have the potential to retain the bulk of contaminant mass following a contaminant release and, if uncontrolled, may act as a continuing source for bedrock ground water contamination. Therefore, it is critical that the overburden and weathered bedrock zones be adequately characterized.

3.1.3.2. Generic Conceptual Hydrostratigraphic Models for the Bedrock Regions of NJ

Investigations of contaminated bedrock sites usually cover areas ranging from less than one acre to tens of acres. At this scale, the primary importance is to identify those hydraulically conductive fractures, or fracture zones, that provide primary or preferential pathways for advective migration of contaminants. In bedrock, fracture networks confine flow to discrete pathways that may be strongly heterogeneous, discontinuous and anisotropic. Conductive fractures can be viewed as aquifer units that subdivide the bedrock into blocks. These blocks function as aquitard units in the first approximation, but often include minor conductive fractures within the bedrock matrix.

A generic conceptual hydrogeologic model offers clues to the ground water flow pathways typical of a given bedrock region, including which type of fractures are extensive and conductive to control the bedrock flow, the impact of bedrock structure on the flow, recharge, discharge and contaminant pathways, as well as the relative importance of sorption and diffusion interactions between conductive fractures and the matrix. The generic model synthesizes past investigative experiences from the region to offer a starting framework and selection of proper investigative tools for a contaminated site in the same region.

In general, two categories of contaminated bedrock sites are distinguished in bedrock regions of New Jersey: sites without a saturated overburden and sites with the saturated overburden. For the latter category, the investigative sequence overburden>bedrock should be followed, and the flow relationship between the overburden and the bedrock becomes an additional critical investigative issue.

3.1.3.2.1 Generic Hydrostratigraphic Model for the Newark Basin

The Newark basin is a half graben filled with non-marine sedimentary and igneous rocks of Triassic and Early Jurassic age. The Newark basin forms the largest physiographic province (Piedmont province) in the northern half of New Jersey. The beds are typically tilted at angles of 5°-15° and strike northeast but are locally faulted and folded (Olsen 1980; Schlische 1992; Herman 2001). The sedimentary formations consist of repeated sequences of sandstone, siltstone and mudstone beds, in varying proportions, which reflect climate-driven oscillation, from fluvial deposition during the Stockton Formation to the lacustrine cycles in Lockatong, Passaic and Feltville formations. A high-resolution stratigraphy of the Newark basin was developed based on analyses of 6,770 m of continuous cores obtained as part of the Newark Basin Coring Project (Olsen et al. 1996). This publication, and composite cores also available at http://www.ldeo.columbia.edu/~polsen/nbcp/cores.html, provides a valuable resource for determining the prevalent rock color (red, gray, black) and type, and the stratigraphic position of a given site. Intrusive and extrusive volcanism related to plate margin rift tectonics occurred episodically and contemporaneously within the New Jersey portion of the Newark basin. See http://3dparks.wr.usgs.gov/nyc/mesozoic/newarkbasin.htm for additional discussion.

Experience from numerous contaminated sites shows that a leaky, multi-unit aquifer model (Michalski 1990; 2010; Michalski and Britton 1997; Dougherty et al. 2004; Herman 2010) provides an appropriate generalization of ground water flow and contaminant migration in the Newark basin. In this model, certain bedding-plane partings act as discrete aquifer units within the dipping, layered bedrock, owing to their relatively high transmissivity and large areal extent. Near-vertical jointing in adjacent beds provides for leakage between the adjacent aquifer units, but vertical extent of such cross-bed leakage is commonly inhibited by the termination of the majority of the joints at bed boundaries. The major bedding fractures exhibit transmissivity values exceeding 1,000 gpd/ft, and occur at uneven stratigraphic intervals of tens to hundreds of feet (Michalski 1990). There are indications associating such fractures with thinly bedded gray beds (Lacombe and Burton 2010).

Some major conductive bedding fractures are traceable for more than 1,500 feet along their strike (Michalski and Britton 1997). Ground water flow in such conductive fractures is typically along strike direction, providing preferential ground water flow pathways with velocities that exceed 10 ft/d at low ambient horizontal hydraulic gradient values (0.001-0.0001). High transmissivity combined with low storativity values (10^{-5} to 10^{-6}) for such conductive bedding fractures leads to quick propagation of hydraulic stresses (Michalski 2010).

The parallelism of the most prominent set of joints in the basin with the strike of bedding (Herman 2005) reinforces the bedding-parallel flow. Herman (NJGS 2010) noted that in some parts of the Passaic formation, relict soil horizons previously rich in calcite and gypsum have been partially dissolved out from the bedrock, leaving vuggy, tabular beds of siltstone and mudstone with relatively high transmissivity. These paleosols parallel the bedding, and are observed most frequently in the middle part of the Passaic formation. In the Lockatong Formation, Lacombe (2010) found that the black, carbon- rich laminated mudstone fractures easily, has relatively high hydraulic conductivity and is associated with high natural gamma-ray count rates. Dark-gray laminated mudstone is less fractured and has a lower bulk hydraulic conductivity than the black carbon-rich laminated mudstone. He also noted that the light-gray massive mudstone and red massive mudstone are highly indurated and tend to have the least fractures and a low hydraulic conductivity.

At many small contaminated sites, bedding fractures of much lower transmissivity may provide major preferential contaminant migration pathways. Such minor transmissive fractures are commonly found in low-permeability bedrock sections that would be considered aquitards in water-supply applications. Because these bedding fractures facilitate horizontal flow and contaminant migration within the low-permeability bedrock section, they function as preferential local flow pathways within the layered aquitard section, which are much slower relative to the flow in the previously described major conductive bedding fractures. If NAPL is discharged into such an aquitard unit, it tends to act as an extended source area, from which contaminants may slowly diffuse into more transmissive fractures (Kueper and Davies 2009).

A more realistic representation of this generic model includes a weathered bedrock zone over the multi-unit bedrock. The weathering of mudstone and shale beds tends to produce numerous poorly integrated fractures, which impart a much lower permeability but greater storage to the weathered zone relative to the underlying bedrock. While the weathered zone resembles a porous medium, extensions of the major or minor bedding fractures into this zone provide the principal pathways for down dip flow and drainage across the weathered zone (Michalski 2010). Up-dip flow prevails in ground water discharge zones within the discrete aquifer units. Discharge can also occur through upward leakage along joints. Contaminant migration may also be down-dip contaminant migration may also occur if pumping induced flow is occurring (Lewis-Brown et al. 2005).

Faults generally act as barriers to ground water flow, as they tend to disrupt the continuity of the discrete aquifer units. However, the occurrence of flow parallel to the trace of a fault needs to be considered.

Matrix diffusion effects can be significant in sandstone beds of the Stockton formation and portions of the Passaic formation due to their relatively high primary porosity, but less important for mudstone lithofacies (Goode et al. 2010).

The igneous rocks in the Newark Basin (basin is capitalized and lowercase in the document – need to be consistent) include basalt flows and diabase intrusions. Both rock types exhibit low matrix permeability, but the columnar jointing and the contacts between individual lava flows that make the Orange Mountain, Preakness and Hook Mountain Basalts provide secondary pathways for vertical and horizontal migration pathways (Herman 2010). The latter pathway can be enhanced where sedimentary or volcanoclastic rocks are included between the individual lava flows. Results of permeability studies of intrusive diabase bodies are inconsistent. The hydraulic tightness of diabase intrusions and the surrounding baked argillitic rocks is well documented from domestic well data and observations in quarries in the central and western New Jersey. On the other hand, a permeable character of the intrusive contact between the Palisades sill and the underlying sedimentary rocks, attributed to thermal fracturing and cracking of both formations, was determined just north of the New Jersey border (Matter et al. 2005). Herman (2010) has documented the presence of fractures along breaks in compositional layering as well as tectonically controlled steeply dipping fractures within boreholes advanced in diabase in Mercer and Hunterdon counties.

There exists a significant body of published studies on-sites in the Newark basin of New Jersey, eastern Pennsylvania and New York State near the border with New Jersey. A partial bibliography of publications is compiled in Appendix A. Many of these publications, especially those performed or sponsored by government entities, are available online.

3.1.3.2.2 Generic Hydrostratigraphic Model for Paleozoic Sedimentary Bedrock

The Paleozoic sedimentary bedrock includes sandstone, shales, conglomerate, limestone and dolomite of the Ridge and Valley province of the Appalachian Mountains. These folded rocks form ridges made of resistant beds and valleys made of more erodible material. Faulting and jointing resulting from tectonic stress, erosional unloading and deglaciation are common. Glacial drift covers large portions of the region. Structural features, mainly longitudinal fold axis and transverse faults, exert strong control over the surface and subsurface drainage patterns. Limestone and dolomitic rocks of this region are prone to karst development.

Evidence from this and other Paleozoic regions in Minnesota, Wisconsin and Ontario indicates that certain bedding fractures provide principal flow and contaminant migration pathways. Thus, the main features of the generic conceptual flow model used for the sedimentary formations of the Newark basin are also applicable to non-karstic members of the Ridge and Valley province.

3.1.3.2.3 Generic Hydrostratigraphic Model for Crystalline Bedrock

The crystalline bedrock of New Jersey includes Cambrian and Precambrian metamorphic and igneous rock of the New England province. These rocks have very little, if any, primary porosity. Ground water flow is controlled by fractures developed by tectonic forces, thermal contraction due to rock mass cooling, unloading due to erosion and deglaciation, and weathering. Complex patterns of folding, faulting and fracturing resulted from superposition of these processes over the long geologic history of this region. Strong topographic relief and the presence of various glacial deposits, including permeable drift in the valleys, are typical characteristics of this region.

Because lithologic and structural changes are common in the region, few regional hydrogeologic generalizations can be made. Geologic maps with the largest scale available should be consulted to determine the lithology, joint patterns, foliation trends, lineaments, stream patterns, or other features relevant to the small scale of the investigation. Herman (2006a) conducted borehole geophysical surveys in Middle Proterozoic granite and gneiss at two Morris County sites. He noted the presence of layering fractures and tectonic fractures in the boreholes.

While no single predictive generic hydrostratigraphic model is recognized for all types of crystalline rock masses, some helpful generalizations can be made. Discrete fracture models are still appropriate for contaminant investigations at sites in crystalline rocks. Preferential fracture flow pathways often follow lineaments and foliation trends. As a result, ground water flow in metasediments is more predictable than in cases of igneous rock masses. The amount of fracturing attributed to deglaciation and weathering tends to decrease with depth. A conceptual flow model revealed by studies in granitic gneiss at the Mirror Lake, New Hampshire. (http://toxics.usgs.gov/sites/mirror_page.html) study site features the presence of pods of high-transmissivity fractures that are connected by low-transmissivity network. The latter controls the regional transmissivity of the rock mass, while the former controls the local flow and contaminant transport.

Whereas crystalline bedrock sites present a greater challenge for determining fracture connectivity between wellbores than sites in sedimentary bedrock, the borehole-scale investigation methods described in the following section apply to both categories of bedrock sites.

3.1.3.3 Literature Review

Prior to doing field work, a literature review should be performed to gain an understanding of the regional and local hydrogeology. These data can be easily obtained and are less expensive to collect than invasive field work. The order of preference for sources of information to use in the conceptual hydrostratigraphic model is site-specific data, data from nearby sites, regional studies, and literature values.

Following are some of the available resources that may be used to develop an initial conceptual hydrostratigraphic model:

- A United States Geological Survey (USGS) topographic map and an aerial photograph that include the site to evaluate the topography and the drainage pattern in the vicinity of the site. The topographic map for the New Jersey quadrangles can be viewed or downloaded at http://elibrary.rutgers.edu/quadpage/index.html.

- Published geologic and hydrogeologic reports and maps are available for purchase through the Department’s Maps and Publications Office (http://www.state.nj.us/dep/njgs/pricelst/pubsinfo.htm). The New Jersey Geological Survey (NJGS) has made numerous geologic maps and reports available for download online at http://www.state.nj.us/dep/njgs/pricelst/njgsmaps.htm . These include the surficial and bedrock geology of New Jersey at a scale of 1:100,000. The data are provided in Environmental Systems Research Institute's Geographic Information Systems (GIS) shapefiles. Many USGS 7.5-minute Geologic Quadrangle Maps are also available for downloading online or purchase from the Department's Maps and Publications Sales Office. A list of available quadrangle geologic maps can be found at http://www.state.nj.us/dep/njgs/pricelst/njgsmaps.htm . Where available, information from the quadrangle map should be used instead of the 1:100,000 map.

- Hydraulic conductivity values for New Jersey geologic formations are contained in Table 3 of the document "Guidelines for Delineating Well Head Protection Areas in New Jersey" (http://www.state.nj.us/dep/njgs/whpaguide.pdf). In addition, figure 4-11 of the United States Environmental Protection Agency (USEPA) document "Handbook Ground Water and Contamination" provides range of hydraulic conductivity values for a variety of media. It is available at http://www.epa.gov/nscep/index.html, searching for 625690016A.

- Effective porosity values may be found in "Guidelines for Delineating Well Head Protection Areas in New Jersey" (http://www.state.nj.us/dep/njgs/whpaguide.pdf). This document indicates that the effective porosity for sand and gravel is approximately 25% and approximately 2% for clay, silt, and bedrock. In addition, table C.3.2 of the document "Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water" (http://www.epa.gov/osw/hazard/correctiveaction/resources/guidance/rem_eval/protocol.pdf) provides a range of effective porosity values for common aquifer materials.

- An estimate of ground water and contaminant flow velocities and distance traveled may be obtained using ground water flow equation and factoring in the type of contaminant to estimate the extent of the contaminant plume. A variety of computer software is also available for calculating contaminant fate and transport at http://www.epa.gov/nrmrl/gwerd/csmos/index.html.

- The Department's i-MapNJ Geology is another valuable resource accessible at: http://www.state.nj.us/dep/gis/imapnj_geolsplash.htm. Available data layers include the bedrock geology of the state, faults, folds, dikes, bedrock-surface topography, surface topography, bedrock outcrops, ground water recharge areas, bedrock and surficial aquifers, quarries, aerial photography, and quadrangle boundaries.

- United States Department of Agriculture's soil surveys provide insight into the subsurface at a particular location, including drainage of the soil and information on perched and/or shallow water tables. These data are available at http://websoilsurvey.nrcs.usda.gov/app/.

- Determine if the site had undergone any previous environmental or geotechnical investigations and collect any usable data. If any monitoring wells exist, determine their integrity, obtain their logs, and any monitoring data. Determine if any production wells operated at the site. Obtain information concerning the production well's pumping history, water quality, depth, boring log and construction details.

- If nearby sites are undergoing environmental investigations, logs of wells, hydraulic testing results, ground water sampling data or other pertinent information may be available. The Department's online interactive mapping website at http://www.state.nj.us/dep/gis/newmapping.htm may be used to determine the status of any environmental investigations at nearby sites, and a file review can be conducted to obtain specific information about nearby contaminated sites via the Open Public Records Act (OPRA) process. The OPRA website is www.nj.gov/dep/opra/. The Department's OPRA website includes the Department's on-line well search tool to check for recently installed wells, both monitoring and supply wells http://datamine2.state.nj.us/DEP_OPRA/OpraMain/categories?category=WS+Well+Permits .

- Determine if any high capacity production wells existed in the proximity to the site, which may have historically depressed the local water table. A well search may be conducted through the Department's Bureau of Water Systems and Well Permitting at http://www.nj.gov/dep/watersupply/well/well_search.htm. Old water supply reports that were prepared for most counties may also provide valuable historic data on production and supply wells, including their logs and performance data. These reports are available through the Department's Maps and Publications Office http://www.state.nj.us/dep/njgs/enviroed/freepubs.htm#ic. Additionally, local and/or county health departments may have data on current use of residential supply wells and ground water quality data in the study area.

3.1.3.4 Perform Field Reconnaissance of the Site and Surrounding Area

Following the literature review, a field reconnaissance should be performed. Items of interest identified during the literature search should be evaluated. In addition, note any of the following features:

- surface water features
- topography
- areas of impermeable and permeable surfaces
- locations of AOCs
- stressed vegetation
- evidence of staining and spillage
- subsurface utilities (i.e., manholes)

If the site is located in an area where a bedrock aquifer may have been impacted, identify bedrock outcrops near the site and document bedrock type and any structural features observed. Measure strike and dip of bedding and joint sets and foliation trend, if applicable. Make observations on the frequency of jointing, and the spacing of bedding plane partings (Compton 1985) (Barnes 2004). Note any manifestation of water seeps and their association with specific fractures or joint sets. Note primary and secondary sedimentological features such as mudcracks, evaporate replacement features, bioturbation, root penetrations, paleosols and microbreccias as these features have been associated with high porosity intervals (Herman GANJ 2001).

3.1.3.3 Perform surface geophysics to ascertain buried structures and subsurface stratigraphy

Surface geophysical methods can be useful to reveal various properties of the subsurface, which as stated by the Department's Field Sampling Procedure's Manual "can be used to determine hydrostratigraphic framework, depth to bedrock, extent of high concentration ground water contaminant plumes, the location of voids, faults or fractures…". For a detailed discussion of the various surface geophysical techniques, please refer to chapter 8 of the Department's August 2005 "Field Sampling Procedures Manual". The manual is available online at www.state.nj.us/dep/srp/guidance/fspm/.

The USGS has compiled information concerning geophysical methods pertaining to ground water investigations at their online portal: http://water.usgs.gov/ogw/bgas/g2t.html.

3.1.3.4 Updating and Maintaining the Conceptual Hydrostratigraphic Model

As additional data become available, the hydrostratigraphic model should be refined and used to guide the remainder of the remediation. Maps and cross sections should be updated with each phase of the investigation to support the hydrostratigraphic model. The updated model should contain the following:

- A site map and a stratigraphic cross section that identifies:
    - each AOC at the site
    - ground water contour maps
    - sample contaminant concentrations
    - contaminant isoconcentration maps
    - horizontal and vertical extent of the source area and plume including the extent of free and residual product, if present
    - all wells and borings used to construct the cross section

3.2 Delineation of NAPL and Sources of Ground Water Contamination

The assessment and delineation of sources of ground water contamination is critical for determining appropriate remedial actions at a site and for the success of the remedial action. Sources of ground water contamination should be delineated and remediated rapidly to limit continued migration of contamination from the source area. The person responsible for the remediation must identify the need for any interim remedial measure necessary to remove, contain, or stabilize a source of contamination to prevent contaminant migration and to protect the public health and safety and the environment.

Sources of ground water contamination discussed in this document include:

- Unsaturated Zone Soil Contamination
- Saturated Zone Contamination
- Free and Residual Product

3.2.1 Development of a Conceptual Hydrostratigraphic Model of Source Areas

The investigator should develop a conceptual model of the contaminant source area. Guidance for developing a conceptual model for a LNAPL release is available in the Department's LNAPL technical guidance document. Guidance for developing a conceptual hydrostratigraphic model for a DNAPL release is available from the USEPA. The 1992 USEPA guidance entitled "Estimating Potential for Occurrence of DNAPL at Superfund Sites" http://www.epa.gov/superfund/health/conmedia/gwdocs/pdfs/estdnapl.pdf presents several initial conceptual models for evaluating DNAPL releases. In addition the USEPA issued guidance in September 2009 entitled "Assessment and Delineation of DNAPL Source Zones at Hazardous Waste Sites". This guidance is available at http://www.epa.gov/nrmrl/pubs/600r09119/600r09119.pdf

3.2.2 Unsaturated Zone Sources of Ground Water Contamination

Potential sources of ground water contamination within the unsaturated zone include any soil contamination that exceeds the site specific IGWSRS. The Department's Remediation Standards at N.J.A.C. 7:26D-1.1(b) establish the requirement to develop site specific IGWSRS. The site specific standard must be developed on a site-by-site basis, pursuant to the Department's authority under N.J.S.A. 58:10B-12a. The Department outlines several methods that may be used to develop site specific IGWSRS in the Soil Remediation Standards Guidance available at http://www.nj.gov/dep/srp/guidance/rs/.

3.2.3 Saturated Zone Sources of Ground Water Contamination

Sources of ground water contamination within the saturated zone include free and residual product (NAPL) and back diffusion of contaminants from fine grained sediments or the bedrock matrix. NAPL is a source of ground water contamination when contaminants dissolve from the

NAPL into the ground water. In addition, contaminants that have diffused into relatively low permeability zones from higher permeability zones when ground water contaminant concentrations were high, will back diffuse into the high permeability zones from the lower permeability zones when ground water contaminant concentrations are lower in these high flow zones.

3.2.3.1 Back Diffusion of Contaminants from Low Permeability Zones

Non-NAPL sources can be a significant source of ground water contamination (Air Force 2007). As a plume advances, contaminants are lost from high permeability zones (sands) by diffusion into low permeability zones (fine sands, silts and clays). After a remedial action depletes free and residual NAPL, the low permeability layers or lenses act as a "non-NAPL source". In general, the older a release the more likely it is that back diffusion is the source for contaminant concentrations at the head of a persistent plume (Air Force 2007).

The release of contaminants stored in low permeability zones can sustain contaminant discharge from source zones even after free and residual NAPL is removed. This can explain persistent releases of contaminant from plume heads where little, if any, DNAPL can be found. Therefore, remedial actions that treat both the high permeability and low permeability zones should be selected to reduce the potential for contaminant rebound after treatment is completed.

3.2.3.2 Free and Residual Product

Where free and/or residual product is encountered, it shall must be delineated. Hydrogeology, as well as the type of contaminants present should be understood before the source area investigation is undertaken. Information on the hydrogeology (e.g., thickness of confining layers, if any) should first be obtained in areas outside of the suspected source areas. Once the hydrogeology is defined, the subsurface investigation is performed in the source area in such a way as to prevent the downward or lateral migration of free or residual product into uncontaminated areas. Techniques that may be used to delineate free and residual product include:

- field screening
- geophysical techniques
- vertical profiling techniques
- Equilibrium Partitioning Evaluation of Soil Sample Analytical Results

The Department's Technical Requirements establish timeframes for the remediation of LNAPL for Site Remediation. Additional guidance concerning LNAPL remediation is available from the Department's LNAPL guidance document.

3.2.3.2.1 Field Methods for Detecting Free and Residual Product

Field screening of subsurface soil samples can be used to identify, characterize and delineate the presence of free and residual NAPL at a site. Many field methods are available and are outlined below.

For sites with LNAPL, a cone penetrometer (CPT) equipped with laser induced fluorescence (LIF) can detect gasoline, diesel fuel, jet fuels, fuel oil, motor oil, grease, and coal tar in the subsurface. This screening data can be used to guide an investigation or removal action or to delineate the boundaries of a subsurface contamination plume prior to installing monitoring wells or taking soil samples. Information concerning LIF can be found at http://www.clu-in.org/characterization/technologies/lif.cfm.

For sites impacted with DNAPL, it is very important to limit invasive investigation in the DNAPL zone to avoid opening further pathways for DNAPL migration during the environmental investigation. An outside-in investigative approach should be used in these instances (USEPA 1994). The outside-in approach includes drilling in areas known to be free of DNAPL before drilling in DNAPL zones. Data collected from borings outside of the source zone should be used to form a conceptual model of site hydrogeology, stratigraphy, and potential DNAPL pathways. After the conceptual model is developed, use this information to determine the safe vertical limit of penetration into the DNAPL source area. The USEPA issued guidance on methods for delineating DNAPL entitled "Site Characterization Technologies for DNAPL Investigations" (USEPA 2004) which is available at http://clu-in.org/download/char/542r04017.pdf .

An iterative process for delineating the extent of free and residual DNAPL is outlined in "A Comparison of Field Techniques for Confirming DNAPL" (Griffin and Watson). Their strategy for identifying and delineating DNAPL in unconsolidated subsurface material is as follows:

- Evaluate existing ground water quality data to establish the 1% solubility or effective solubility isopleth boundary for the contaminant(s) in question.
- Confirm the presence of free or residual product and delineate using a combination of the membrane interface probe (MIP) system and laboratory analysis of soil samples. The MIP study would be completed and the results evaluated prior to selecting soil core sample locations.
- Collect soil core samples from high the concentration areas identified during the MIP survey and evaluate the soil data with respect to phase equilibrium partitioning algorithms.
- Determine the analytical concentrations in soil that would be indicative of free product at the given study area. The soils and MIP data can be combined and an interpretation made of the DNAPL area.

In addition, in "Assessment and Delineation of DNAPL Source Zones at Hazardous Waste Sites" http://www.epa.gov/nrmrl/pubs/600r09119/600r09119.pdf the USEPA outlines a method whereby converging lines of evidence are used to assess if DNAPL is present at a site (USEPA 2009). When the technique infers that DNAPL is present, multiple lines of evidence should be used as outlined in the USEPA paper.

Papers that evaluate the costs and performance of DNAPL characterization technologies include; "DNAPL Characterization Methods and Approaches, Part 1: Performance Comparisons" (Kram 2001), and "DNAPL Characterization Methods and Approaches, Part 2: Cost Comparisons" (Kram 2002).

3.2.3.2.1.1 Soil Field Screening Techniques

Soil cores must be screened for the presence of contamination during the investigation. Field screening techniques that should be used to determine if free or residual product is present include:

- Visual Identification
- Organic Vapor Monitors (e.g., FID, PID)
- Soil Agitation Tests
- Ultra Violet Fluorescence
- Hydrophobic Indicator Dyes

A study by Cohen et al. (Cohen 1992) found the following:

- Hydrophobic dye methods, followed by ultra-violet (UV) fluorescence were the simplest, most practical, and effective means for direct visual identification of clear, colorless NAPL in soil samples.

- For volatile NAPL, analysis of organic vapors in soil sample headspace can be used to screen samples for further examination.

- NAPL may be detected in water samples by adding a very small amount of hydrophobic dye.

Unaided visual identification of free and residual product is unlikely where the product is clear and colorless. However, this is a useful method where the product has color such as coal tar.

3.2.3.2.1.2 Geophysical Techniques

Surface geophysics may provide evidence of NAPL source areas. Acoustic, electrical resistance, electromagnetic, gamma or neutron contrasts may, under the appropriate conditions, provide indirect evidence based upon sufficient contrasts between source area, background, porosity and moisture content.

3.2.3.2.1.3 Vertical Profiling Techniques

Direct push technologies (DPT) are a category of equipment that push or drive steel rods into the ground. They allow cost-effective, rapid sampling and data collection from unconsolidated soils and sediments. A variety of equipment is available, particularly in the type of attachments used at the end of rods to collect samples and data. These attachments may collect soil, soil gas, or ground water samples; may conduct in situ analysis of contaminants; or may collect geophysical data that are continuously logged as the DPT rods are advanced. Continuous logs of subsurface conditions are particularly valuable because they help to develop a three-dimensional conceptual site model.

Direct push investigative methods such as CPT or Geoprobe® can be used to provide direct and inferred evidence of NAPL presence as well as identify soil types. Various sensor technologies may be employed. Permeable membrane sensors and a MIP provide inferred evidence based on VOC partitioning across metal-polymer membranes. These down-hole sensors can be combined with FID, PID, halogen specific detector (XSD), and ion trap mass spectrometer (ITMS) detectors to provide semi-quantitative data on NAPL composition and concentration. A table of methods for detecting and delineating free and residual product contamination source zones and a cost comparison of the methods are provided in Kram et al. (Kram 2001). The investigator should choose an appropriate technology based on lithology, degree of consolidation, chemical composition of NAPL, requirements for confirmation sampling, and maturation of the sensing technology.

3.2.3.2.1.4 Soil Gas Surveys

Soil gas surveys can be used to screen for LNAPL source accumulations and may be useful in determining the DNAPL entry location. Soil gas surveys consist of insertion of soil vapor collection devices into the subsurface and passive or active (via application of a slight vacuum) collection of samples and measurement of VOCs using a gas chromatograph or other vapor measurement instrumentation. Soil gas surveys may not be appropriate for very low permeability or saturated soils.

3.2.3.2.2 Residual Product Presence Inferred from Soil and Ground Water Analytical Data

Contaminant concentrations in soil samples or ground water samples may be used to determine if free and or residual product are present at a site. The USEPA guidance document entitled "Assessment and Delineation of DNAPL Source Zones at Hazardous Waste Sites" http://www.epa.gov/nrmrl/pubs/600r09119/600r09119.pdf outlines several methods whereby chemical concentrations in soil and ground water samples are used to assess if DNAPL is present at a site (USEPA 2009). This guidance is included as Appendix A of this guidance.

Ground water sample analytical data may be used to determine if residual product is present. For DNAPL chemicals, residual product shall be considered to be present if the contaminant is detected in ground water at concentrations equal to or greater than 1% of its effective water solubility in accordance with the Technical Requirements for Site Remediation.

3.2.4 Interim Remedial Measure

An interim remedial measure should be implemented to remove, contain, or stabilize any source of ground water contamination. The Department's Technical Rules and LNAPL guidance document establish timeframes for the remediation of free phase LNAPL. For other cases, free and residual product must be removed or treated where practicable or contained where removal or treatment is not practicable. Effective remediation technologies will depend on the NAPL location as well as the site-specific geologic and hydrogeologic constraints. Descriptions and evaluation of remediation technologies as well as training opportunities can be found at the following websites:

- USEPA Contaminated Site Clean-Up Information (Clu-in) http://www.clu-in.org/
- Interstate Technology and Regulatory Council (ITRC) http://www.itrcweb.org/

3.3 Characterization of Unconsolidated Aquifers

This section of the guidance concerns the characterization and delineation of contamination in unconsolidated aquifers. It has two major parts:

- an approach for conducting remedial investigation of ground water contamination in unconsolidated aquifers

- a bibliography of useful guidance materials for the characterization and remediation of ground water contamination (see Section 6.0).

3.3.1 Remedial Investigation Approach in Unconsolidated Aquifers

It is important to characterize the hydrogeology, determine the direction of ground water flow and delineate the horizontal and vertical extent of contamination in each contaminated hydrostratigraphic unit at the site. Characterization must be sufficient to enable the selection of appropriate and effective remedial actions to satisfactorily address the contamination.

With an initial hydrostratigraphic model in place the investigator can begin delineation of the contamination and revision of the model. Site specific data must be collected to perform the following:

- determine ground water flow direction
- delineate the contaminant plume
- determine hydraulic properties of the impacted water bearing zones
- design and install a contaminant plume monitoring network

It is important that the investigator incorporate the site-specific data into the hydrostratigraphic model so that future investigations are designed to fill in data gaps identified during each phase of the investigation.

3.3.1.1 Determination of ground water flow direction

Ground water flow direction must be determined at the site. The ground water flow direction is determined by collecting ground water elevation data at a minimum of three surveyed permanent or temporary monitoring wells. The use of surveyed temporary wells to determine ground water flow direction is discussed further in the Department's Alternate Ground Water Sampling Techniques Guide, which may be found at http://www.nj.gov/dep/srp/guidance/agws/ . When constructing the ground water contour map, ensure that wells are screened within the same hydrostratigraphic unit and, where possible, are placed equidistant from each other to provide optimal triangulation.

If LNAPL is observed in the any of the wells, the depth to water measurement can be corrected for the LNAPL using the formula:

$$\text{Corrected Depth to Water} = D_w - (P_t \times G)$$

where:

D_w = Depth to Ground Water
P_t = Product Thickness
G = Specific Gravity (density of free product / density of water)

(USEPA, 1998)

If a surface water feature is on or adjacent to a contaminated site, it is important to collect surface water elevation measurements concurrent with ground water elevation measurements to aid in construction of the contour map. If the site is located in an area that is tidally influenced, synoptic ground water and surface water levels should be collected using a pressure transducer recording hourly for a minimum of 24 hours.

If it is suspected that the site is influenced by pumping wells, synoptic ground water elevation data should be collected using a pressure transducer recording at sufficient intervals for a sufficient period of time (e.g., hourly for 7 days) to account for weekend and week day pumping schedules.

Computer software may be used to develop the ground water contour map. However, ground water contours constructed using software must be checked to ensure that they are reflective of actual field conditions.

The following factors may affect ground water flow direction and preparation of the contour map:

- excavations with high permeability backfill
- pumping wells and sumps
- subsurface utilities
- perched water
- lateral heterogeneities resulting in preferential flow paths
- surface water

3.3.1.2 Delineation of Contaminant Plume using Transects, Vertical Profiling and Real Time Analysis

An effective approach to delineate the horizontal and vertical extent of contamination is to use direct push technologies and real time analysis followed by the installation of monitoring well network. A properly designed monitoring well network is difficult without an adequate understanding of contaminant distribution, soil heterogeneity and contaminant migration pathways. An example of a poorly designed monitoring well network can occur when the contaminant plume is diving which is described in “Downward Solute Plume Migration:

Assessment, Significance, and Implications for Characterization and Monitoring of "Diving Plumes**"** (API, 2006) which can be found at http://www.api.org/ehs/groundwater/upload/bull24-2.pdf . In this example, downgradient wells installed at the same depth as the source area wells will be clean leading the investigator to believe that the plume is delineated, when in reality the plume is migrating beneath the downgradient wells. Likewise as a contaminant plume approaches an area where there is an upward hydraulic gradient, such as a surface water body, it will rise. The forces that induce a dissolved contaminant plume to dive or rise are independent of a dissolved contaminant's specific gravity.

To delineate the contaminant plume and to characterize the hydrogeology at the site, the Department recommends the installation of transects of temporary well points, using a direct push method or similar technology, perpendicular to the ground water flow direction. An initial transect is made across the plume source area to determine the width of the plume and source area, then additional transects are located downgradient. Transect construction should continue downgradient until the tip of the contaminant plume is reached. The initial transect should be made in an "outside in" fashion, starting from clean side gradient locations and advance towards the source area.

Downgradient of the source area, transects can be installed using an inside out approach, with the initial ground water sample obtained at the estimated centerline of the plume and advancing out until a clean zone is reached, laterally and vertically. A sufficient number of ground water samples should be collected to adequately characterize the plume centerline and the side gradient and downgradient edges of the plume.

Vertical profiling should be conducted at each boring location within the transect. Contaminant and stratigraphic data should be collected vertically at each location until the vertical extent of contamination is defined.

The placement of each downgradient transect should be based upon the working hydrostratigraphic model. The investigator should have an idea of the extent of the plume, as based upon a hydraulic gradient, an estimated date of release, the hydraulic conductivity value of the aquifer, and its estimated effective porosity value.

If earlier work identified ground water contamination with no apparent source, ground water samples should be collected up and side gradient of the known contamination in a step-wise fashion until sufficient data points exist for the investigator to determine the source area for the contamination.

When data gaps are identified in the working hydrostratigraphic model, additional investigation should be performed to fill them. For example, when the horizontal and vertical extent of the plume does not fit with the working hydrostratigraphic model, the investigator should evaluate if additional investigation is required to update contaminant delineation and the site hydrostratigraphic model.

3.3.1.3 Installation of Monitoring Well Network

A ground water monitoring network and monitoring program should be developed and installed. Using the updated hydrostratigraphic model, permanent wells should be installed to monitor the contaminant plume over time, to document ground water flow direction in each impacted water-bearing unit, and to document vertical gradients. The monitoring well network is also used to aid in evaluating the effectiveness of the remedial action. Section 4.0 of this document discusses the design of the ground water monitoring network, the ground water sampling plan, ground water sampling program, and data evaluation to assess the effectiveness of the remedial action.

3.3.1.4 Determination of Site Specific Hydraulic Properties

Site-specific hydraulic properties such as hydraulic conductivity, effective porosity and hydraulic gradient should be determined at the contaminated site. Hydraulic conductivity may be obtained by performing pumping or slug tests.

Slug tests may be conducted to determine site-specific hydraulic conductivity values. Slug test values are not necessarily representative of the aquifer as a whole. An adequate number of tests need to be performed to determine hydraulic characteristics throughout the contaminant plume to support the conceptual model. The intensity and level of effort to achieve the most accurate data possible should be guided by the size and type of discharge, presence of receptors, and the type of remediation to be utilized.

For many contaminated sites, slug testing may be sufficient; however, by performing single well pumping tests or multi-well pumping tests the investigator obtains a more realistic assessment of aquifer characteristics. Pumping tests need to be long enough to estimate hydraulic properties based on well and aquifer characteristics. The bibliography provides several guidance documents and references concerning the performance of pumping tests. To improve the accuracy of the aquifer property data, tracer tests may be conducted to determine the ground water flow velocity and effective porosity. The USGS maintains a web site on the performance of tracer tests at http://toxics.usgs.gov/topics/tracer_tests.html .

3.3.1.5 Update the Hydrostratigraphic Model

After the additional permanent ground water monitoring wells are installed based on the temporary well information, the conceptual hydrostratigraphic model should again be refined. A ground water contour map should be constructed for each impacted hydrostratigraphic unit and each ground water sampling event. The contour maps and isopleths maps should be compared over time to determine if there are any changes related to seasonal variation, pumping influence or contaminant migration or degradation.

It may be beneficial to construct vertical flow nets as part of the site hydrostratigraphic model to predict or explain contaminant distribution. Flow nets aid in determining the extent of capture at the depth of a pumping well and in determining where to focus surface water investigations.

Construction of vertical flow nets requires ground water potentiometric data from multiple well clusters.

Based upon the contaminant distribution, receptors, and anticipated remedial technology, the hydrostratigraphic model is evaluated for data gaps, as well as any discrepancies between field data and the model.

3.3.2 Ground Water Classification Exception Area

After characterizing the contaminant plume, determine its fate and transport for development of the CEA. The Department's CEA guidance document at http://www.nj.gov/dep/srp/guidance/cea/cea_guide.htm should be used when developing the CEA.

3.4 Characterization of Bedrock Aquifers

This guidance consists of two major sections:

- A structured approach and resources needed for designing and conducting remedial ground water investigations at new and existing contaminated sites is presented in the second section.

- A "toolbox" in Appendices that provides descriptions of the various bedrock investigative tools. Appendices B through D include procedures for vertical flow tracing using salt tracer, depth-discrete sampling, packer testing, a bibliography of articles on New Jersey geology (see Section 6.0), and a list of useful website links.

3.4.1 Remedial Investigation Approach for Bedrock Aquifers

This section discusses investigative strategy and approaches to complete hydrogeologic characterization of contaminated bedrock sites at new sites and then at sites with existing wells. The level of the characterization needs to be sufficient to select appropriate and effective remedial actions for the site. It should consist of the following goals:

- Characterize the regional and local geology, including structural features influencing ground water flow in the bedrock (bedding plane fractures, joint sets, foliation and lineaments, etc).

- Locate conductive fractures that control ground water flow at the site, define ground water flow, recharge and discharge zones for these fractures (viewed as aquifer units), and characterize their hydraulic properties as well as intervening aquitard units. Synthesize the data into a conceptual site hydrogeologic model.

- Characterize contaminant type and distribution within the site model and complete delineation of the vertical and horizontal extent of contaminant plumes.

- Characterize the contaminant source zone and quantify its impact on the dissolved plume(s).

A sequential and comprehensive approach to achieving these goals is outlined below. The investigator may choose a narrower or broader approach to investigate a particular site, if the investigative goals are realized.

3.4.2 Investigating New Contaminated Bedrock Sites

3.4.2.1 Implement initial test drilling program

The initial test bore hole drilling program must use an outside-in approach. An outside-in approach (USEPA 1992) requires that no borehole is drilled into the known or suspected source area until the site-specific hydrostratigraphy is developed and source impacts on ground water are well understood. This approach prevents inadvertent product migration induced by drilling into an unexplored source zone.

It is understood that the contaminant distribution information collected using this approach will provide a general indication of the contamination in bedrock. However, because the initial boreholes are somewhat distal from the source area by design, additional monitoring wells will likely be necessary to fill gaps in the understanding of contaminant plume configuration. Even in the scenario that none of the fractures sampled in the test boreholes detect contamination, the hydrostratigraphic information obtained will be used to site wells intended to intercept fractures connected to the suspected source area.

Install a minimum of three deep open-hole temporary test boreholes that will be converted to monitoring wells upon completion of their testing. The boreholes must be placed outside the suspected source area/NAPL entry point: two of them to be sited along strike of bedding (or foliation) on either side of the suspected source and the third located down-dip of bedding from the source area. If there is an existing on-site production well, this well should be included in the testing program, as historic use of this production well likely impacted the flow pattern and the well may still provide a conduit for cross-well flow. The production well could replace a third deep borehole, if it satisfies the well placement criteria.

The temporary test boreholes need to be deeper than the maximum suspected depth of the contaminant plume to ensure the vertical contaminant delineation. Consider the depths of proximate domestic supply wells when deciding how deep to advance the boreholes. If there are no other indicators to suggest the borehole depth, advance the two strike-parallel boreholes to 150 feet. The third (downdip) borehole should be deeper to terminate at the same stratigraphic depth as the other two holes. At suspected DNAPL sites, an updip extension of the plane originating at the terminus of the third (downdip) borehole should pass below a suspected DNAPL entry point into the bedrock to insure that possible down-dip DNAPL migration along a conductive bedding fracture would be detected at the downdip borehole. Six-inch diameter of the open-holes in the test boreholes is preferred, as it permits conversion of the open-holes to two-inch diameter short-screen wells at the completion of the test boring program.

If the temporary test boreholes will be left open for an extended period of time (7 to 10 days generally), the risk of cross contamination through vertical cross-flows needs to be considered and mitigated. This risk can be quantified based on the measured cross-flows and concentrations of contaminants obtained from depth-discrete sampling (Appendix C). A packer can be inflated at a proper location in the hole to isolate the fractures involved in the cross-flow and mitigate the problem until the time of borehole conversion to a monitor well. Alternatively, a flexible borehole liner can be used to mitigate the cross-flow.

3.4.2.2 Conduct borehole geophysical logging and other downhole characterization tests

Geophysical logging of the test boreholes should be completed to characterize the lithology and fracturing, including identification of potentially conductive fractures. Many borehole logging methods exist. A firm specializing in geophysical investigations should be utilized to perform the logging. A recommended set of geophysical logs should include the following:

Table 3-1 Borehole Geophysical Methods Bedrock Investigations.

LOG TYPE	APPLICATION	REMARKS
Caliper	Casing depth and diameter, borehole diameter and fracture zones	All arms open as a unit, well alignment can limit some calipers.
Borehole Image Processing System (BIPS) or Acoustic Televiewer (AT)	Fracture orientation, lithology, structural features and solution opening; Casing depth, borehole orientation/deviation.	Turbidity can interfere with BIPS images; well alignment can limit the type of viewer and ability to advance. AT cannot detect oxidation or precipitation of minerals or fracture infilling.
Gamma	Identify lithologic change (e.g., sandstone vs. shale), gamma marker beds are used for stratigraphic correlation to determine dip and strike of bedding.	Can be used in the cased/screened section of boreholes but the signal is attenuated.
Fluid Temperature and Electrical Conductivity (EC) or Resistivity	Deflection in temp. and EC logs can indicate potential inflow fractures/zones	Quality of temperature probes vary
Heat Pulse Flow Meter	Rate and direction of vertical flow. Location of in-flow and out-flow zones may be inferred.	Quality of probes vary. Only high-resolution meters can measure typical ambient borehole flow rates.
Resistivity	Identify lithologic changes	Alignment can limit the type of viewer and ability to advance

For a detailed discussion of borehole geophysical techniques, please refer to chapter 8 of the Department's August 2005 "Field Sampling Procedures Manual". The manual is available online at: www.state.nj.us/dep/srp/guidance/fspm/. See also Keys 1990; Williams and Johnson 2004; Johnson et al. 2005; http://toxics.usgs.gov/sites/mirror_page.html and http://toxics.usgs.gov/investigations/fracrock_aquifers.html.

3.4.2.3 Conduct vertical flow measurements and sampling

The direction and amount of ambient vertical flows (cross-flow) within the open-hole of each of the test holes should be measured to identify locations of conductive inflow and outflow fractures. Current tools available to measure vertical flow in the borehole include a high-resolution heat pulse flowmeter http://water.usgs.gov/ogw/bgas/flowmeter/) or a salt tracing method (Appendix B). The USGS indicates that a HPFM with a fully fitted diverter can usually measure flow of 0.01 to 1.5 gallons per minute. Be sure to follow the manufacturer's operational instructions to obtain the best possible resolution of the instrument to resolve very low flow rates. The ambient flow measurement results are used to identify conductive fracture locations and to determine hydraulic head in such fractures relative to a composite water level in the open hole based on this simple rule: The head in an inflow producing fracture is greater than the composite head, and lower in a water exit/outflow fracture.

Once the saline tracer (Appendix B), or high-resolution heat pulse flowmeter testing, has been completed and interpreted, the next step is to obtain screening-level contaminant concentrations in individual inflow fractures via depth-discrete sampling of hole segments above and below inflow fractures for a steady-state flow condition (Michalski 2001; 2010). Further discussion of this method and its interpretation can be found in Appendix B. This method not only allows identifying which of the conductive inflow fracture of the open hole is most contaminated but also quantifying cross-flow impacts on specific exit/outflow fractures. The need and urgency of cross-flow mitigation measures is then determined. The depth-discrete sampling provides a screening-level information of vertical contaminant distribution among inflow fractures. In the context of the initial conceptual flow model, this information may provide valuable clues as to the pathway to, and location of the source of this contamination. Since water quality in outflow fractures within the borehole is impacted by mixture of waters originating from inflow fractures, the depth-discrete sampling at outflow fractures cannot provide information on native water quality in such fracture outside the borehole. The limitations may also be extended to packer sampling of outflow fractures (Michalski 2001; 2010). To make up for this limitation, outflow fractures/zones should be targeted for monitoring when converting the temporary test holes to monitoring wells (see Section 4.2.3.2.10).

To further characterize the distribution of transmissivity and hydraulic heads after completion of the depth-discrete sampling, another set of vertical flow measurements can be made while water is pumped from the top of the water column at a steady rate of 1 gpm or so. The velocity profiles measured under the pumping and the ambient conditions can provide the basis for determining, for major conductive fracture identified, the transmissivity and heads using the method developed by Paillet (1998) (see also Day-Lewis, et. al., 2011). This would accomplish a hydrogeologic characterization of the bedrock at each test hole when packer testing is not feasible.

3.4.2.4 Conduct packer tests

In addition to sampling of ground water inflow fractures/zones, straddle packer testing may be utilized to measure the hydraulic head and transmissivity values where inflow or outflow fractures were identified. Packer testing may also provide information on the hydraulic connection (vertical leakage) between transmissive fractures. Pumping from packer-isolated intervals can be used as short-term pumping tests to verify hydraulic connections along conductive fractures (aquifer units) across multiple borehole/well locations, thus providing further information regarding the hydrostratigraphic conceptual model. Appendix D provides a detailed discussion of packer testing and recommended test procedures.

When straddle packer testing is used to obtain ground water quality information from discrete fractures, the limitations of the method should be accounted for. Specifically, if an outflow fracture is packer-isolated, it may not be possible to collect a representative ground water sample, as the volume removed during routine purging may not eliminate the effects of prior crossflows into the outflow fracture.

3.4.2.5 Synthesize the testing results into the initial site-specific hydrogeologic model

First, determine a site specific orientation of the strike and dip of bedding, as an initially selected value may not be accurate. If a distinct gamma marker is identifiable on gamma logs of each of the three boreholes, determine the elevation of the marker and obtain the orientation of the marker by solving a three-point problem on a site map. Lacombe (2010) describes the use of gamma logs not only for the dip-and-strike determination but also for development of a gamma-based lithostratigraphic correlation for a large site in the Lockatong formation. Where no distinct gamma marker is apparent, as is often the case with red-color only portions of the Passaic formation, look for a good match between the lower-to-higher segments of the three gamma profiles, starting with an offset corresponding to the initial/generic orientation of the bedding. The gamma matching can be supplemented by matching of the optical or acoustic images of the three holes. An accurate determination of the strike-and-dip of bedding is an important task of the investigation, given the prominent role of bedding fractures in flow and contaminant transport at sedimentary bedrock sites.

Cross-sections running perpendicular to the strike of the bedding should be prepared to depict site-specific hydrogeologic bedrock framework. Project onto these cross-sections the open-hole segments of the test boreholes, the locations of conductive fractures identified and their type (inflow or outflow), bulk transmissivity values, and concentrations of principal contaminants of concern. Draw bedding-parallel lines through the locations of the conductive fractures identified to determine if they extend to, or correlate with, any of the other boreholes. If so, such conductive fractures should be viewed as major or minor aquifer units, depending on their transmissivity. A preliminary site-specific hydrogeologic model thus obtained provides the basis for locating additional monitoring wells to refine the model and delineate contaminant plume(s).

3.4.2.6 Convert the test holes into monitoring wells

Upon completion of the testing program, each temporary test borehole should either be grouted or converted into a monitor well with a short screened interval monitoring a single zone of interest. Three potential monitoring targets are usually recognized: a) the most contaminated fracture, b) the most transmissive fracture and c) an exit fracture. In some cases, two of these targets become one, so that only two targets remain. A deep exit fracture/zone makes a good monitoring target to delineate the vertical extent of ground water contamination.

Multilevel monitoring systems can be deployed in the test boreholes if more than three target fracture/zones are to be monitored. Major multilevel commercial systems include Solinst’s Waterloo, Westbay, and FLUTe systems. Refer to manufacturers’ brochures regarding the specifics of each system. Consider possible limitations of each system on purging/sampling, water level measurements, and long-term dependability over expected monitoring duration at the site.

Since the initial test holes may not delineate the extent of contamination, additional monitoring wells will be necessary to fill data gaps in the understanding of the contaminant source area and plume configuration.

3.4.2.7 Update Conceptual Hydrostratigraphic Model and Complete Plume Delineation

Based on the site cross-sections, potentiometric maps and analytical results obtained from the sets of well clusters installed at these locations, additional wells are sited to complete horizontal delineation of the plume(s) for the most transmissive and the most contaminated fractures/ aquifer units, whichever is applicable. This may involve several mobilizations of a drilling rig.

As target intervals for the additional wells can be determined by projections, additional testing during drilling of new wells may be limited to resolving specific data gaps, or interpretive alternatives, posed by the current understanding of the site-specific conceptual model. The site-specific bedrock conceptual model can then be tested and verified.

A minimum of three monitor wells will be necessary to determine ground water flow direction within each such transmissive fracture, as each fracture can be considered a discrete aquifer. Determination of ground water flow direction is a pre-requisite for contaminant plume delineation in each unit.

Short-duration tests lasting less than an hour should be used to determine aquifer hydraulic properties (hydraulic conductivity and transmissivity). Longer term aquifer tests, lasting 48 to 72 hours, will be necessary to determine aquifer boundary and recharge conditions and, ultimately, a capture zone.

Consider installing pressure transducers in the existing monitoring well network when a new activity that may stress the aquifer, such as drilling, well development, purging, packer or slug tests, is going to be performed. Quick hydraulic responses to the applied hydraulic stress can often be seen at relatively distant wells completed in the same transmissive fracture, owing to the large fracture transmissivity and its very low storage (Michalski 2010). These data can be used to verify hydraulic connections and determine aquifer parameters for the major transmissive fractures.

After aquifer hydraulic properties have been determined, perform fate and transport calculations to estimate the future migration potential for the contaminant plume. This information is necessary to establish a ground water CEA and should factor into the remedy selection.

3.4.3 Investigating Contaminated Bedrock Sites with Existing Monitoring Wells

Existing bedrock monitor wells with excessive open-hole intervals can be problematic, as they may provide conduits for cross-contamination of the bedrock aquifer. Of special concern in this regard are existing wells drilled into the known or suspected source/DNAPL area. Drilling regulations set the maximum length of open/screened interval at 25 ft, but shorter lengths (5 ft to 15 ft) are recommended because the objective is to monitor the specific transmissive fracture identified during the investigation. The site-specific conceptual bedrock flow and transport model may be lacking, or a generic flow model inappropriate for the site was used, which resulted in mis-delineation of contaminant plume(s) and failed remediation efforts.

The following steps can be taken to correct these bedrock characterization problems:

1. Perform literature/office studies and select a generic flow model appropriate to the site location.

2. Conduct supplemental logging and testing in existing wells (if not available), including gamma, electrical conductivity and temperature logs, slug and salt tracing tests to determine bedrock structure (e.g., dip and strike of bedding), likely locations of conductive fractures, the bulk transmissivity of open/screened intervals of each well, and possible vertical transport of contaminant along some wells.

3. Prepare a site cross-section perpendicular to the strike of the bedding. Project onto the cross-section elevations of open/screened intervals of all existing wells, locations of conductive

fractures (interpreted at Step 2), bulk transmissivity values, concentrations of a principal contaminant of concern, and the water level elevations. Identify any conductive fractures and contaminated zones that follow a common projected bedding plane for two or more existing wells. Prepare ground water flow maps with contaminant concentration data for a contaminated zone that extends through three or more wells. A preliminary site-specific bedrock flow model is thus formulated at this step.

4. Identify critical data gaps and design additional well(s) to bridge such gaps and correct construction of any existing problem wells with excessive length of open hole.

5. When installing such additional wells, at least one well should be installed in a test hole in which a full set of geophysical tests is conducted and vertical cross-flows are measured and sampled. In addition, hydraulic responses of selected existing wells to drilling and development of the test hole, or other new wells, can be monitored through pressure transducers installed in such wells. The observed responses (direct versus indirect) provide means of testing the hydraulic connection between the wells, as well as improving and verifying of the site-specific bedrock flow model. An iterative approach works well to achieve an adequate level of flow and contaminant characterization in the bedrock.

3.4.4 Characterize the Ground Water Contamination Source Zone

In general, proceed with source zone characterization after the site-specific hydrogeologic framework has been developed and impact of the source on plume generation is understood. From this outside-in approach, a safe penetration depth for drilling into the source zone can be determined, which is an important consideration for possible DNAPL sites. (Contaminant concentrations detected in near-source wells will indicate if DNAPL may be present at the source).

The hydraulic structure and properties of source zone should be determined using appropriate hydraulic tests considering the permeability, the depth of source zone, the type of NAPL if present, and the remedial alternatives being considered for the site.

Analyses of rock cores collected from selected wellbores can provide information on contaminant concentrations and their likely physical forms (capillary residuals, sorbed, or diffused into the matrix), and on contaminant mass distribution between the rock solids and aqueous phases within the source zone. This information is vital for selecting effective remedial actions to mitigate the source zone contamination (O’Brien 2011). Many times a significant mass of contamination is trapped in the highly weathered bedrock zone and follows the more competent bedrock topography. Rock coring can be useful in defining this highly weathered zone and top of competent bedrock.

For LNAPL cases, where weathered bedrock is present in the source area, it may be possible to remove the bulk of the contamination by excavating the weathered bedrock to the competent stratum. Source delineation and remediation can be conducted simultaneously using field methods in the open excavation as screening tools to guide the source area removal. After source remediation is completed, ground water downgradient from the removed source area must be monitored to determine the effectiveness of the remediation.

3.4.5 Ground Water Classification Exception Area

With the hydraulic properties known and the contaminant plume characterized, the fate and transport of the plume is determined for development of the CEA and placement of sentinel monitoring wells. The Department's CEA guidance document at http://www.nj.gov/dep/srp/guidance/cea/cea_guide.htm should be referenced when developing the CEA.

3.4.6 Fractured Rock Remediation Technologies

The remediation of contaminated bedrock aquifers is a challenging endeavor. In many instances, hydraulic containment may be necessary in addition to in-situ treatment technologies. Many technologies are available to remediate ground water. The USEPA maintains a remediation technologies web site, clu-in http://www.clu-in.org/, which lists technologies and project profiles.

Descriptions of remediation technologies are available at http://www.clu-in.org/remediation/.

Fractured rock case project profiles are available at http://www.clu-in.org/products/fracrock/.

4.0 Performance Monitoring of Active Ground Water Remedial Actions

This section of the document provides technical guidance on:

- establishing an effective ground water monitoring network
- implementing a ground water monitoring program
- data evaluation methods for determining the effectiveness of ground water remedial actions
- technology re-evaluation

It is recommended that this section of the technical guidance be used in conjunction with related Department guidance for Monitored Natural Attenuation (MNA) and Remedial Action Permits for Ground Water.

The Remedial Action Permits for Ground Water Guidance requires that a ground water remediation system is operational and functional and that the source of ground water contamination has been effectively removed, treated or contained prior to issuance of the permit. This document provides guidance on how to monitor a ground water remediation system to make these determinations. Once issued, the Ground Water Remedial Action Permit will establish the ground water monitoring frequency.

4.1 Ground Water Monitoring Network

A ground water monitoring program is required that will effectively monitor the performance of the ground water remedial action. The number and type of wells to be included in the performance monitoring program will be dependent on the Conceptual Site Model (CSM) including consideration of the size and stability of the plume, relative levels of contamination, and presence of potential receptors. Performance monitoring wells should be positioned to evaluate the long-term performance of the remedy and ensure protection of receptors. Performance monitoring wells will typically include source area wells, plume fringe area wells, and sentinel wells. Performance monitoring may also include wells perpendicular to the groundwater flow direction to monitor lateral components of the plume.
An effective ground water monitoring network should consist of the following components:

- Source Area Monitoring

  Monitoring well(s) should be located within the source area, where possible, or as close downgradient as feasible to monitor the effectiveness of source area remediation.

  Well clusters should be installed downgradient from, and as close to, the source area to monitor the vertical extent of the source area and to assure that the source remedial action is effective.

- Contaminant Plume and Plume Fringe Monitoring

  Plume monitoring well(s) should be located downgradient of the source area within the contaminant plume. Plume monitoring wells should be located along the predominant contaminant flow path(s). In cases where there may be significant variations in contaminant flow, the location of the plume monitoring wells should account for any such changes.

  Plume fringe monitoring wells should be located to monitor the horizontal and vertical contaminant concentrations at the leading, lateral and vertical edge(s) of the contaminant front.

- Sentinel Wells

  Downgradient Sentinel Wells: The location of the downgradient sentinel wells should focus on detecting migration of the contaminant plume towards the nearest unimpacted downgradient receptor and allow sufficient time for a remedial response to be implemented on the contaminant plume to prevent impact to the potential receptor.

  The downgradient sentinel well location is determined based on the distance to potential receptors and contaminant velocity. The sentinel well should be located based on the behavior of the contaminant plume (i.e., stable, shrinking, advancing, or diving) and on ground water monitoring data obtained from the plume monitoring wells.

  If no receptors are identified within the model-predicted maximum extent of the contaminant plume, sentinel wells may not be necessary provided that monitoring data confirm modeling results, the leading edge of the plume is effectively monitored, the remedial action is effective and ground water contaminant concentrations are decreasing.

  Lateral Sentinel Wells: Where there is a potential for the ground water contaminant plume to impact receptors located adjacent to the contaminant plume, sentinel wells should be located laterally adjacent to the plume. Sentinel wells should be located adjacent to and outside of the lateral extent of ground water contamination.

  Vertical Sentinel Wells: Sentinel wells should be located below the contaminant plume where there is the potential to impact a receptor at depth. Sentinel wells should be positioned at the water table above the contaminant plume to evaluate potential for vapor intrusion where receptors are present.

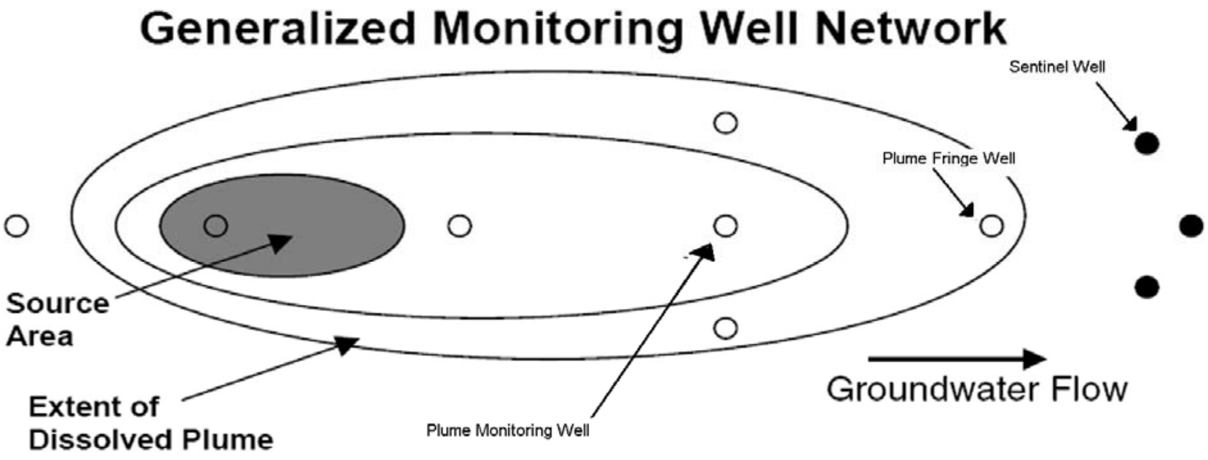
  Ground water quality data obtained from sentinel wells must be below the applicable Ground Water Remediation Standards. As the plume shrinks, wells within the contaminant plume may become sentinel wells.

- Background Monitoring Wells

  Where background contamination is impacting the site and background ground water monitoring is being conducted, monitoring well(s) should be located and sampling data evaluated as described in section 2.8 of this document.

Figure 4-1 below depicts the components of a ground water monitoring network as described above.

Figure 4-1 Generalized Monitoring Well Network



4.2 Ground Water Monitoring Program

A ground water monitoring program is required to assess the effectiveness of any ground water remediation. This section discusses recommendations for all ground water monitoring programs excluding MNA.

Consistent ground water sampling should be completed on a recurring schedule, as it will provide data for evaluation of trends using trend analysis or statistical methods. In general, it is recommended that ground water sampling be conducted on a quarterly basis during the first two years of the remedial action implementation to determine if data are affected by seasonal variations. Less frequent sampling after the initial two years may be appropriate if the remediation system is meeting cleanup objectives. Additional guidance on the frequency of performance monitoring can be found in the Department's Remedial Action Permits for Groundwater Guidance. Where remediation objectives are not being met during the initial two years, the ground water remediation system must be re-evaluated and upgraded or changed.

Analytical parameters should be selected based on contaminants of concern and their degradation products. In addition, indicator parameters which provide data to determine the effectiveness of in-situ treatment systems should be monitored.

The ground water monitoring program should include collection of synoptic ground water levels to monitor ground water flow velocity, and to determine if there is a relationship between ground water elevation and contaminant concentrations.

4.3 Data Evaluation

Ground water monitoring data must be evaluated to determine if the remediation system is meeting cleanup objectives. Statistical tests can be useful to evaluate trends in contaminant concentrations in source and contaminant plume monitoring wells. Information concerning statistical methods can be found in the USEPA guidance document entitled "Statistical Analysis of Groundwater Monitoring Data at RCRA Facilities Unified Guidance March 2009" which may be found at: http://www.epa.gov/osw/hazard/correctiveaction/resources/guidance/sitechar/gwstats/unified-guid.pdf . The following statistical tests are recommended by USEPA for contaminant trend analysis (USEPA 2009(b)):

Wilcoxon Rank-Sum (Mann Whitney-U): Non-parametric test to compare medians of two populations; data need not be normal; some non-detects OK; should have no significant spatial variability; useful at very small sites in interwell comparisons and for certain intrawell comparisons; also useful for updating background.

Linear Regression: Parametric estimate of linear trend; trend residuals must be normal or normalized; useful for testing trends in background or at already contaminated wells; can be used to estimate linear association between two random variables.

Mann-Kendall Trend Test: Non-parametric test for linear trend; non-detects OK; useful for documenting upward trend at already contaminated wells or where trend already exists in background.

Theil-Sen Trend Line: Non-parametric estimate of linear trend; non-detects OK; useful for estimating magnitude of an increasing trend in conjunction with Mann-Kendall test.

In addition, the United States Air Force developed a Microsoft Access based program for the optimization of ground water monitoring programs. The program includes a trend analysis package. The program is available for download at http://www.gsi-net.com/software/free-software/maros.html#MAROS_Installation_Instructions.

Data from the monitoring network should be evaluated as follows:

- Source Monitoring Wells: Data must show either:
    - Where treatment or removal of the source of ground water contamination is practicable, monitoring data must show a decreasing trend in contaminant concentrations and that the Ground Water Remediation Standards will be attained.
    - Where treatment or removal of the source of ground water contamination is not practicable, monitoring data must show the source area is contained and that the contribution of contaminant mass to the dissolved plume is reduced to the extent practicable.
- Contaminant Plume Monitoring Wells: Data must show that there is a decreasing trend in contaminant concentrations and that the Ground Water Remediation Standards will be attained.
- Sentinel Wells: Data must show that sentinel wells remain below the Ground Water Remediation Standards.

4.4 Technology Specific Monitoring

Monitoring Hydraulic Containment Systems

Hydraulic containment is often performed to prevent, or significantly reduce the migration of contaminants and would be used when treatment or removal of the source area contamination is technically impracticable. Hydraulic containment can either be passive (i.e., impermeable barrier) or active (i.e., pump and treat).

Guidance on monitoring hydraulic containment can be found at the following links:

USEPA has established a website entitled "Methods for Monitoring Pump-and-Treat Performance," part 1 http://www.epa.gov/superfund/health/conmedia/gwdocs/pdfs/ptmethods.pdf and part 2 http://www.epa.gov/superfund/health/conmedia/gwdocs/pdfs/ptmethods2.pdf

USEPA “A Systematic Approach for Evaluation of Capture Zones at Pump and Treat Systems” which may be found at http://www.epa.gov/nrmrl/pubs/600R08003/600R08003.pdf provides guidance on pump and treat capture zone analysis.

Monitoring In-Situ Treatment Systems

There are numerous in-situ treatment technologies available for the remediation of contaminated ground water. Monitoring needs for in-situ treatment systems are based on the remedial technology selected. ITRC (http://www.itrcweb.org/gd.asp) and CLU-IN (http://cluin.org/about/) offer in-depth descriptions of remedial technologies and the monitoring parameters for those technologies.

4.5 Remediation Technology Re-Evaluation

Pursuant to proposed N.J.A.C. 7:26E-5.6(c), the person responsible for conducting the remediation shall submit a revised remedial action workplan or remedial action workplan addendum when a remedial action does not perform as designed. Contaminant trend data and hydraulic data should be used to make this evaluation. Modification or a change of a ground water remediation technology should be considered when:

- Contaminant levels are not decreasing in source monitoring wells or data indicate that a source containment system is not achieving remediation objectives.

- The contaminant levels detected in any of the contaminant plume fringe monitoring wells are not decreasing; or

- Sentinel well(s) become impacted.

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Appendix A: USEPA Assessment and Delineation of DNAPL Source Zones at Hazardous Waste Sites

United States
Environmental Protection
Agency

Ground Water Issue

Assessment and Delineation of DNAPL Source Zones at Hazardous Waste Sites

Bernard H. Kueper* and Kathryn L. Davies**

1.0 - Introduction

Groundwater contamination from classes of chemicals such as chlorinated solvents, polychlorinated biphenyls (PCBs), creosote, and coal tar is frequently encountered at hazardous waste sites (40, 43). These types of contaminants have low solubilities in water and have densities greater than that of water. Therefore, they can exist in the subsurface as Dense, Non-Aqueous Phase Liquids (DNAPLs) and have the potential to migrate as a separate liquid phase to significant distances below the water table in both unconsolidated materials and fractured bedrock. Because of the physicochemical properties associated with DNAPLs, they migrate through the subsurface in a very selective and tortuous manner (13, 27, 29). Thus, the majority of DNAPL present in the subsurface may not be found immediately below the entry location and directly encountering DNAPLs with conventional drilling techniques may be difficult.

Determining the presence or absence of a DNAPL is an important component of the conceptual site model and is critical to the proper selection of the remediation approach. Subsurface DNAPL acts as a long-term source for dissolved-phase contamination and determines the spatial distribution and persistence of contaminant concentrations within the dissolved-phase plume. Once it has been determined that DNAPL exists within the subsurface, subsequent characterization activities are typically conducted to better delineate the boundaries of the DNAPL source zone. The DNAPL source zone is the overall volume of the subsurface containing residual and/or pooled DNAPL. It should be recognized that there will be uncertainty associated with the delineation of the DNAPL source zone. In addition to the DNAPL, there may be significant amounts of contaminant mass that have diffused into low permeability zones. Back diffusion of contaminant mass from these zones may sustain dissolved-phase plumes for significant periods of time, even after DNAPL has been removed. Establishing the presence and locations of such non-DNAPL sources is beyond the scope of this document.

In January 1992, EPA published a Fact Sheet entitled ‘Estimating Potential for Occurrence of DNAPL at Superfund Sites’ (42) with the goal to help site personnel determine if DNAPL-based characterization strategies should be employed at a particular site. In September 1994, EPA issued a subsequent Fact Sheet entitled ‘DNAPL Site Characterization’ (39) discussing direct and indirect methods to assess the presence of DNAPL in the subsurface. Since the publication of the initial fact sheets, there have been advancements in characterization tools, site investigation approaches (14) and knowledge of DNAPL source zone architecture within the subsurface. This document builds on information from the previous fact sheets to provide a framework for not only assessing the presence of DNAPL, but also for delineating the spatial extent of the DNAPL source zone, a priority at many sites due to the more prevalent use of *in-situ* remediation technologies (38). The strategy described in the present document utilizes converging lines of evidence that incorporate the scientific advancements in the field and expands the applicability of the document to include both unconsolidated deposits and fractured bedrock. An iterative, flexible site investigation approach (7) is encouraged.

2.0 - Nature of the DNAPL Source Zone

Upon release to the subsurface, DNAPL will distribute itself in the form of disconnected blobs and ganglia of organic liquid referred to as residual DNAPL, and in connected distributions referred to as pooled DNAPL (Figure 1). Residual DNAPL is found both above and below the water table within the pathways of DNAPL migration, and typically occupies between 5% and 30% of pore space in porous media (6, 27, 44) and in rock fractures (21). Residual DNAPL is trapped by capillary forces, and typically will not enter an adjacent monitoring well, even under the influence of aggressive groundwater pumping (6, 27).

Pooling of DNAPL can occur above capillary barriers, which are typically layers and lenses of slightly less permeable material (Figure 1). Pooling can therefore occur at any elevation in the subsurface, and not just at the base of permeable zones. Absence of pooling above clay aquitards and bedrock may be due to the presence of dipping fractures, bedding planes, joints and faults which may allow the continued downward migration of the DNAPL. Pools represent a continuous distribution of DNAPL, and typically correspond to DNAPL saturations of between 30% and 80% of pore space in both porous media and fractures. The frequency of pool occurrence and the thickness of pools are increased by the presence of horizontal capillary barriers, lower DNAPL density, higher interfacial tension, and an upward component to groundwater flow (17, 22). The thickness of pools typically ranges from fractions of an inch to a few feet, depending on fluid and media properties (36) as well as the volume released. Because pools represent a connected distribution of DNAPL, the pooled DNAPL is susceptible to mobilization through drilling activities and can short-circuit along existing monitoring wells and piezometers. In addition, pools may also be mobilized in response to changes in hydraulic gradient. The gradient required to mobilize a pool is a function of the DNAPL-water interfacial

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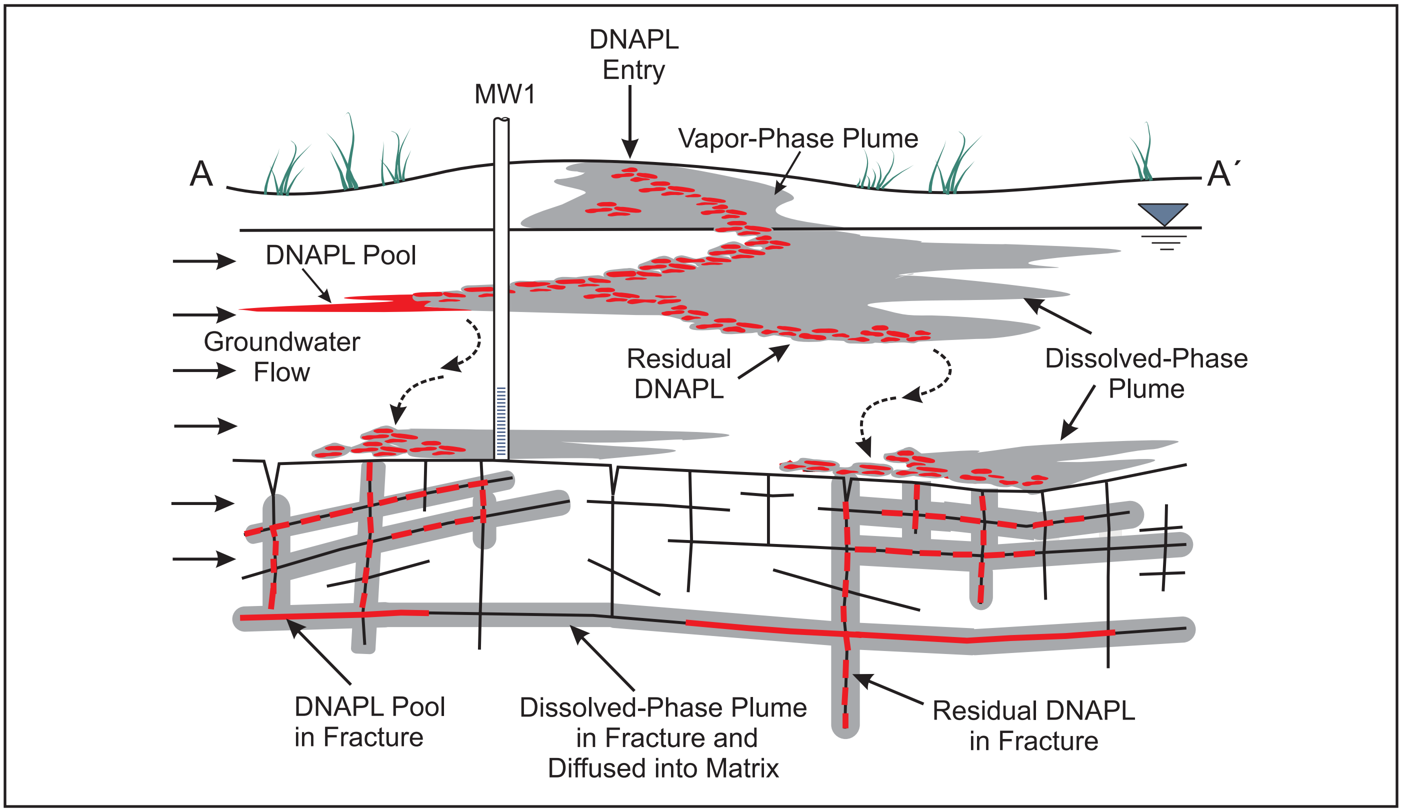

Figure 1 – Schematic illustration of contamination associated with a DNAPL release. Note that DNAPL migrates in three dimensions, and that residual DNAPL accumulated above bedrock is the result of the release at ground surface. The reader is referred to Figure 2 for a depiction of matrix diffusion. Figure is not to scale.

tension, the pool length, and the permeability of the surrounding material (6, 27). Pumping groundwater from beneath DNAPL pools, for example, can lead to an increase in capillary pressure and subsequent downward DNAPL mobilization.

The spatial distribution of residual and pooled DNAPL is strongly influenced by geology, and also by DNAPL properties and release history (frequency, intensity, duration, volume and location). DNAPL migration can occur through lenses and laminations of porous media at the scale of inches or less (17, 29). For DNAPLs that are non-wetting (see wettability in glossary) with respect to water (which is usually the case), migration below the water table is typically through the larger pores (and hence higher permeability regions) in unconsolidated media and larger aperture fractures in bedrock. The orientation of stratigraphic and structural features will largely determine the degree of lateral and vertical DNAPL spreading. DNAPL migration from the release location can occur in any direction, and is typically not greatly influenced by low ambient hydraulic gradients except for creosotes and coal tars which have densities close to that of water.

The overall region of the subsurface containing residual and pooled DNAPL is referred to as the DNAPL source zone. For high density and low viscosity DNAPLs (such as chlorinated solvents), migration in relatively permeable media can cease as soon as a few months to a few years following the time of release (3, 17, 27, 29). Some geological conditions, such as horizontal to sub-horizontal fractures, gently dipping strata and sand seams in low permeability media can give rise to longer time scales for migration of chlorinated solvent DNAPLs, particularly for large volume DNAPL sources. For low density and high viscosity DNAPLs (such as creosote and coal tar), migration has the potential to continue for many decades (12). The overall depth of DNAPL migration is dependent not only on the presence or absence of capillary barriers, but also on the volume released, the interfacial tension, the degree of lateral spreading, and the bulk retention capacity (see glossary) of the medium. Because fractured rock has very low bulk retention capacity, small volumes of DNAPL can migrate greater distances in bedrock in comparison to the same volume released into unconsolidated deposits (18).

Groundwater flowing past residual and pooled DNAPL will result in dissolved-phase plumes of contamination. Complete dissolution of all DNAPL as a result of natural groundwater flow is expected to take from several decades to hundreds of years for most DNAPLs. For multi-component DNAPLs, the presence of more than one component typically suppresses the aqueous solubility of the other components in the DNAPL (6, 27). Exceptions to this can occur, however, when co-solvents such as alcohols are present in the DNAPL. In the absence of co-solvents, the concentration of any particular component dissolving into groundwater can often be approximated using Raoult's Law (2, 6, 27). Early in the dissolution process, the plume chemistry will be dominated by the higher effective solubility components which tend to be those present in the largest mass fraction within the DNAPL, and those with the highest single-component (handbook) solubility values (24). The concentration of any or all components in groundwater downgradient of a multi-component-DNAPL source zone will typically be lower than expected using a single component solubility limit. With time, both the DNAPL composition and the plume composition will change in response to the dissolution process. The dissolved components that comprise the plume will migrate in groundwater subject to advection, dispersion, sorption, volatilization, and degradation processes.

Both residual and pooled DNAPL, and dissolved-phase plumes that are in direct contact with clays, silts, or a porous bedrock matrix, can diffuse into the low permeability media (forward diffusion). If concentrations outside of the low permeability zone become lower than those inside, diffusion will occur back into the higher permeability zone (back diffusion) and can result in plume persistence (5, 33). The forward and back diffusion processes are collectively referred to as matrix diffusion (Figure 2). The persistence of DNAPL in fractures in bedrock, saprolite and clay can be shortened by the matrix diffusion process (19, 28). In addition, the rate of advance of a dissolved-phase plume in fractured rock with a porous matrix can be strongly attenuated by the matrix diffusion process (20, 35). The influence of matrix diffusion on dissolved-phase plume migration in fractured rock and clay relative to other processes such as advection, dispersion, sorption, and possible degradation processes will vary depending on site specific geological conditions and contaminant properties. In general, matrix diffusion has a greater influence on dissolved-phase plume migration in the case of wider fracture spacing, smaller fracture aperture, lower hydraulic gradient, higher matrix porosity, and higher matrix organic carbon.

Above the water table, volatile DNAPL can vaporize into air filled pore spaces (Figure 1). For DNAPLs with significant vapor pressure, this can lead to expanded vapor-phase plumes in the unsaturated zone. The concentration of contaminants in the vapor phase will be governed by the vapor pressure, and for a multi-component DNAPL can often be approximated using Raoult's Law. In relatively warm and dry environments, the persistence of some DNAPLs (e.g., chlorinated solvents) can be relatively short (on the order of months to a few years) in unsaturated media. The absence of residual and pooled DNAPL in the unsaturated zone may not, therefore, be sufficient evidence to conclude that DNAPL has not migrated below the water table at the site of interest.

3.0 - Types of DNAPLs

Coal Tar is a complex mixture of hydrocarbons produced through the gasification of coal that was produced as a by-product of manufactured gas operations as early as 1816 in the United States. It is still produced as a by-product of blast furnace coke production. Coal tar contains hundreds of hydrocarbons, including light oil fractions, middle oil fractions, heavy oil fractions, anthracene oil, and pitch. The low density (typically 1.01 g/cc to 1.10 g/cc

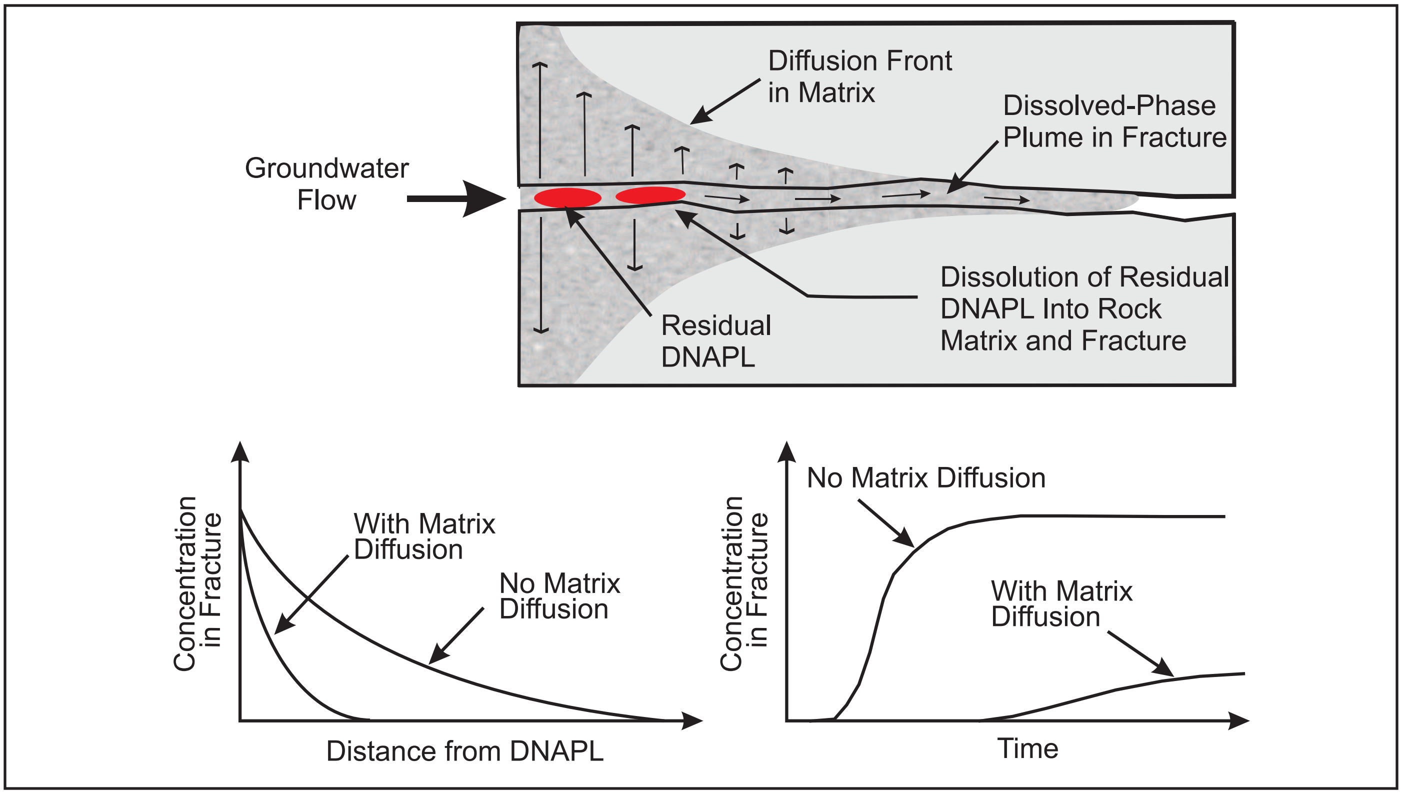

Figure 2 – Matrix diffusion of dissolved-phase contaminants adjacent to DNAPL and along length of plume in fracture. Matrix diffusion can attenuate the rate of plume advance in fractured rock (bottom left concentration vs distance plot), and can result in delayed breakthrough curves (bottom right concentration vs time figure). These factors need to be considered when relying upon groundwater concentration data to assess DNAPL presence.

compared to 1.00 g/cc of water [at 4°C]) and high viscosity (up to 200 to 300 times, or more, than that of water) facilitate long time-scales of migration, with the possibility of movement continuing for many decades following initial release. Due to the lengthy list of compounds present in coal tar, many investigators select a sub-set of coal tar compounds based on mobility and toxicity to assess water quality. These compounds may include benzene, toluene, ethylbenzene, xylenes (BTEX), benzo[a]pyrene, naphthalene, and phenanthrene. Depending on the age of the DNAPL and groundwater velocity, some of the lower molecular weight and more soluble compounds of the coal tar may have been leached out of the DNAPL by the time a site investigation is initiated. Naphthalene is often the dominant compound in present day coal tar (9). In addition, the various components in the plume will migrate at different velocities because of varying degrees of sorption and degradation (often aerobic conditions). The lower molecular weight, less sorbing compounds (e.g., BTEX) can migrate significantly further in groundwater than the higher molecular weight, more sorbing compounds (e.g., PAHs).

Creosote is composed of various coal tar fractions and was commonly used to treat wood products. It is still used today in certain wood treating operations and as a component of roofing and road tars. Creosote is a multi-component DNAPL that contains many hydrocarbons, primarily polycyclic aromatic hydrocarbons (PAHs), phenolic compounds, and carrier fluids such as diesel. The low density (typically 1.01 g/cc to 1.13 g/cc) and high viscosity (typically 20 to 50 times that of water) of creosote facilitate long time-scales of migration, with the possibility of movement continuing for many decades following initial release. Most investigators select a sub-set of creosote compounds, based on mobility and toxicity to characterize water quality, such as naphthalene, benzo(a)pyrene, and phenanthrene.

Polychlorinated Biphenyls (PCBs) are a class of 209 chemical compounds referred to as congeners, in which between one and ten chlorine atoms are attached to a biphenyl molecule. The majority of PCBs were manufactured between 1930 and 1977 under the trade-name Aroclor for use in capacitors, transformers, printing inks, paints, pesticides, and other applications. Aroclors differ based on the amount and types of congeners present. PCBs by themselves are DNAPLs, and were often blended with carrier fluids such as chlorobenzenes and mineral oil prior to distribution. The density of most PCB oils ranges from 1.10 g/cc to 1.50 g/cc, while the viscosity ranges from 10 to 50 times that of water. Most congeners are very hydrophobic and their transport can be retarded strongly relative to the rate of groundwater migration. In some cases, however, PCB transport in groundwater can be facilitated through the formation of emulsions or the presence of colloids.

Chlorinated Solvents such as trichloroethene (TCE), tetrachloroethene (PCE) and carbon tetrachloride (CT) have been produced in large quantities since the mid 1900's. Some chlorinated solvents contain trace amounts of stabilizers, preservatives and impurities. Typical uses vary widely and include dry cleaning, metal degreasing, pharmaceutical production, pesticide formulation, and chemical intermediates. Chlorinated solvents can be encountered as single component DNAPLs (e.g., as primarily PCE at a dry cleaning facility, or as primarily TCE at a vapor degreasing facility), or as part of a multi-component DNAPL containing other organic compounds. The relatively high density (typically 1.10 g/cc to 2.20 g/cc) and low viscosity (typically ranging from half to twice that of water) of chlorinated solvents can result in a relatively short time-scale of migration following release compared to coal tar and creosote. In a dissolved-phase plume, most chlorinated solvents are not retarded strongly relative to the rate of groundwater flow.

Mixed DNAPLs A DNAPL that contains two or more compounds is referred to as a multi-component DNAPL (e.g., creosote). A mixed DNAPL is a multi-component DNAPL that contains a wide variety of organic compounds as a result of blending and mixing prior to disposal operations, or as a result of cotemporaneous disposal. Examples include DNAPLs encountered at former solvent recycling facilities and industrial disposal sites. Such DNAPLs can contain aromatic compounds normally associated with LNAPLs (e.g., toluene) along with chlorinated solvents, PCBs, alcohols, ketones, and tetrahydrofuran. The density of mixed DNAPLs typically ranges from 1.01 g/cc to 1.60 g/cc, and the dissolved-phase plumes associated with mixed DNAPLs usually contain a wide variety of compounds with varying mobility.

4.0 – DNAPL Source Zone Investigation Methods

This section presents various site investigation methods and related interpretation techniques that can be useful when characterizing a DNAPL source zone. These methods and techniques will be relied upon in Sections 5 (Assessing DNAPL Presence) and 6 (Delineation of the DNAPL Source Zone). Additional information is provided in (6, 26, 37).

A Visual Observation

DNAPL obtained from the bottom of a monitoring well or as an emulsion from a pumped water sample is conclusive evidence of DNAPL presence (pooled DNAPL). Monitoring wells can be sampled for DNAPL using bottom loading bailers lowered to the bottom of the well or pumping from the bottom of the well. If an interface probe indicates DNAPL presence, then the sample should be retrieved and it should be confirmed (visually, or through laboratory analysis) that the substance is DNAPL. If DNAPL is visually observed in drill cuttings or in a soil sample for the first time, then a sample should be sent to the laboratory for confirmatory evidence. This line of evidence is applicable in both unconsolidated deposits and fractured rock, but it should be noted that visual observation of DNAPL in rock core is rare because of the aggressive flushing nature of the drilling process. Because of the typically sparse and tortuous nature of DNAPL distribution in the subsurface, DNAPL is not encountered and visually observed within many DNAPL source zones.

B Chemical Concentrations in Soil Above Threshold DNAPL Saturation

Chemical concentrations in soil exceeding the value corresponding to a threshold DNAPL saturation are conclusive evidence of DNAPL presence (see Calculation 1). The threshold DNAPL saturation for use in Calculation 1 should be set to be between 5% and 10% of pore space for all DNAPL types. The particular threshold saturation chosen should result in a chemical concentration in soil that is an order of magnitude higher than that determined in line of evidence C. It follows that high organic carbon content soils and highly hydrophobic chemicals may require the use of threshold saturations toward the higher end of the above range. This method is applicable to unconsolidated media both above and below the water table, but is not applicable in fractured rock. The calculation requires knowledge of site-specific parameters and a quantitative chemical analysis of the soil. Care should be taken to sample soil horizons in core exhibiting the highest headspace readings and the strongest visual indication of DNAPL presence. The use of fixed depth intervals or compositing from several depth intervals is discouraged when collecting soil samples to evaluate the presence of DNAPL. Methanol preservation or a similar technique to reduce VOC losses during handling and transport of soil samples should be employed.

C Chemical Concentrations in Soil Above Partitioning Threshold

Chemical concentrations in soil exceeding the value corresponding to equilibrium partitioning relationships (see Calculation 2) are consistent with DNAPL presence (11). The composition of the DNAPL need not be known (see Calculation 4). The calculation is applicable to unconsolidated media both above and below the water table, but is not applicable in fractured rock. The calculation requires knowledge of site-specific parameters and a quantitative chemical analysis of the soil. Measured concentrations that only marginally exceed the calculated partitioning threshold may be false positives primarily because of uncertainty associated with estimating the soil-water partition coefficient.

D Site Use/Site History

Investigations during the past 30 years have shown that the subsurface occurrence of DNAPL is often associated with the industries, practices, and processes outlined in Table 1. Site Use/Site History can be ascertained using methods such as employee interviews, company purchase and sale records, aerial photographs, and building plans. Former lagoons, underground tanks, floor drains and leach fields are sometimes coincident with the location of DNAPL source areas.

E Vapor Concentrations

The location of a vapor-phase plume may be coincident with the current or former presence of DNAPL in the vadose zone. Mapping the vapor-phase plume may be useful in deciding where to collect additional data. Because some DNAPLs can completely vaporize in relatively short time periods (yet the vapors will persist much longer), the presence of vapors and the mapping of a vapor-phase plume should generally not be used in isolation to conclude that DNAPL is present in the vadose zone, or to delineate the spatial extent of the DNAPL source. Care should also be taken to avoid mistaking vapors derived from off-gassing of a groundwater plume with vapors derived from DNAPL sources. In-situ vapor concentrations can be sampled using invasive techniques (soil vapor surveys), and can be monitored during drilling. This line of evidence is not applicable to DNAPLs lacking a significant vapor pressure (e.g., coal tar, creosote, PCBs).

F Hydrophobic Dye Testing

Hydrophobic dyes such as Oil Red O will partition into DNAPL, imparting a red color to the organic liquid. Dye techniques are particularly useful when encountering a colorless DNAPL. Hydrophobic dye techniques include the jar shake test in which a soil or water sample is placed into a jar with a small amount of dye (6), and down-hole samplers that force a dye-impregnated absorbent ribbon against the borehole wall in either fractured rock or a direct push borehole (30). It should also be noted that the absence of staining on a down-hole ribbon sampler is not evidence of the absence of DNAPL, since only pooled DNAPL can migrate towards the sampler (residual DNAPL may be present in the formation adjacent to the sampling interval, and remain undetected).

Table 1 – Industries and Industrial Processes Historically Associated With DNAPL Presence (modified after USEPA, 1992).

Industry	*Industrial Process*
Manufactured gas plant, Wood preservation (creosote), Electronics manufacturing, Solvent production/recycling, Pesticide/Herbicide manufacturing, Dry cleaning, Instrument manufacturing, Metal product manufacturing, Engine manufacturing, Steel industry coking operations (coal tar), Chemical production, Airplane maintenance, Transformer oil production	Storage of solvents in uncontained drum storage areas, Metal cleaning/degreasing, Metal machining, Tool and die operations, Paint stripping, Use of vapor and liquid degreasers, Storage and transfer of solvents in above and below ground tanks and piping, Burning waste liquids, Storage and treatment of waste liquids in lagoons, Use of on-site disposal wells, Loading and unloading of solvents, Transformer reprocessing, Disposal of solvents in unlined pits.

The following lines of evidence G1 through G6 all make use of groundwater quality data and can be evaluated every sampling round.

G1 Magnitude of Groundwater Concentrations

Sampled groundwater concentrations in excess of 1% effective solubility (see Calculation 3) indicate that the sampled groundwater may have come in contact with DNAPL. If the composition of the DNAPL is not known, Calculation 6 can be used. The distance to the possible DNAPL locations cannot be determined from the magnitude of the concentration alone. Sampled groundwater concentrations downgradient of a DNAPL source zone can be significantly less than the effective solubility because of hydrodynamic dispersion, wellbore dilution, non-optimal monitoring well placement, and degradation processes. In cases where significant degradation is occurring in the dissolved-phase plume, daughter product concentrations can be converted to equivalent parent product concentrations before comparing to the 1% effective solubility threshold (see Calculation 8). However, it should be noted that daughter product compounds may also be part of a multi-component DNAPL. Monitoring well points where groundwater concentrations exceed 1% effective solubility can also be useful in locating additional sampling points potentially nearer to the possible DNAPL source zones. The interpretation of groundwater concentrations exceeding 1% effective solubility is discussed further in (27).

G2 Persistent Plume

The presence of a contiguous and persistent plume extending from suspected release locations in the downgradient direction is evidence of a continuing source (e.g., DNAPL). If ‘sufficient time’ has passed since the last possible introduction of contaminant to the subsurface and the plume has not ‘detached’ itself from the suspected release locations, a DNAPL source may be present. The ‘sufficient time’ is dependent on site-specific conditions such as groundwater velocity and the amount of sorption occurring (see Calculation 7). This line of evidence is applicable to both unconsolidated deposits and fractured rock, but can be inconclusive in environments subject to significant amounts of back diffusion (e.g., fractured bedrock with a porous matrix, fractured clay). Significant amounts of back diffusion can be the source of a persistent plume even if DNAPL is not present. This line of evidence is therefore most applicable to high permeability settings.

G3 Presence of Contamination in Apparently Anomalous Locations

The presence of contaminated groundwater in locations that are not downgradient of known or suspected sources may be evidence of DNAPL presence hydraulically upgradient of the monitoring point in question. An example includes the presence of dissolved-phase contamination in groundwater that is older than the potential contaminant release (using age dating) or in groundwater on the other side of a flow divide located between the monitoring location and suspected release locations. In Figure 1, for example, the presence of contamination in the illustrated monitoring well cannot be explained without the upgradient presence of DNAPL. This line of evidence is not contingent on any concentration threshold. Temporal changes in hydraulic heads and groundwater flow directions, as well as changes in historic pumping patterns should be considered at sites where groundwater extraction has, or is, occurring. Consideration should also be given to the presence of unknown or off-site sources that may account for the observed contamination.

G4 Groundwater Concentration Trends with Depth

Abrupt reversals of groundwater contaminant concentration levels with depth or increasing concentrations with depth can be associated with DNAPL presence. Concentration trends can be best detected using small interval sampling techniques [e.g., direct push sampling devices; short well screens; multilevel completions; cone penetrometer equipped with measurement probes (16, 26)]. Multilevel monitoring completions can be incorporated into open holes in bedrock to provide concentration as a function of depth. Other methods in bedrock include the use of temporary straddle-packer assemblies to sample specific depth intervals, and the use of diffusion bag samplers placed at specific depths. Use of these latter methodologies should be made only when intraborehole flow conditions have been adequately characterized.

G5 Groundwater Concentration Trends with Time

Groundwater downgradient of a multi-component DNAPL may exhibit a temporal decline in the concentration of the higher effective solubility compounds and a stable or increasing trend in time of the lower effective solubility compounds. Highly soluble and mobile compounds, such as low molecular weight alcohols, furans, ketones and some solvents such as methylene chloride may show a decreasing concentration versus time signature downgradient of a DNAPL source zone while at the same time higher molecular weight alcohols and semi-volatile compounds may show a stable concentration trend. This line of evidence is primarily applicable to mixed DNAPLs. Consideration should be given to compound specific biodegradation, which may result in the concentration of certain compounds decreasing and others (such as low molecular weight daughter products) increasing within the plume. Dissolved-phase concentrations downgradient of a single component DNAPL may decline due to removal of some of the source mass during dissolution; a declining concentration versus time signature does not preclude the presence of DNAPL.

G6 Detection of Highly Sorbing Compounds in Groundwater

The detection of highly sorbing and low solubility compounds which have low mobility in groundwater may be associated with a nearby DNAPL source. This line of evidence can be useful in delineating the extent of the DNAPL in the downgradient direction. Examples of compounds that have very low mobility in groundwater (absent transport facilitated by colloids, cosolvents, or emulsions) include PCBs and high molecular weight PAHs.

H Other Types of Methods

Partitioning interwell tracer tests (PITTs) [1, 4, 15] involve the injection and withdrawal of a tracer that has the ability to partition into the DNAPL. While the method can be used to detect the presence of DNAPL, given the significant effort involved in conducting tracer tests, PITTs are typically employed after some level of source zone characterization has been completed. Literature sources suggest (for certain sites with appropriate geologic conditions and contaminant properties) measuring a depletion of Radon-222 in groundwater (34). Direct push platforms can be used to deploy a variety of probes to vertically profile contaminant concentrations. These probes include laser induced fluorescence (LIF) measurement devices (6, 31, 32) such as ROST (rapid optical screening tool) and TarGOST (tar-specific green optical screening tool), which is specifically designed for detecting the presence of coal tar and creosote (32); and probes employing Raman methods (31). LIF techniques respond well to the presence of NAPLs containing aromatic hydrocarbons, but may not be suitable for many chlorinated solvent DNAPLs. Direct push platforms can also be used to deploy a membrane interface probe (MIP) or a hydrosparge probe (8), both of which transfer contaminants to a flowing gas stream for analysis at the surface. Another measurement probe is the precision injection/extraction (PIX) device (23). The use of measurement probes with direct push platforms is becoming increasingly popular, but care should be taken in interpreting results with respect to DNAPL presence given that most of these devices provide a relative measure of total concentration. Consideration of the potential for, and consequences of, false positives should be given to each of these methods.

5.0 - Assessing DNAPL Presence

Determining the presence or absence of DNAPL is an important component of the site characterization process and subsequent development of a conceptual site model. The length of time and degree of effort required to determine the presence or absence of DNAPL will vary from site to site. Once it has been determined that DNAPL resides in the subsurface, the objectives for further investigation and potential remediation strategies can be established. This section focuses on methods to assess the presence of DNAPL; Section 6 of this document focuses on methods to delineate the DNAPL source zone.

Converging lines of evidence can be used to determine whether or not DNAPL is present in the subsurface. Figure 3 presents a graphical summary of the converging lines of evidence approach. Example calculation procedures are contained in Appendix A. All lines of evidence are discussed in Section 4, and are applicable to both unconsolidated deposits and fractured rock, unless noted otherwise. As indicated in Figure 3, either line of evidence A or B will lead to the conclusion that DNAPL is present. If A and B are both found to be negative, then the determination of whether DNAPL is present must be made on the basis of a weight of evidence approach, with multiple converging lines of evidence

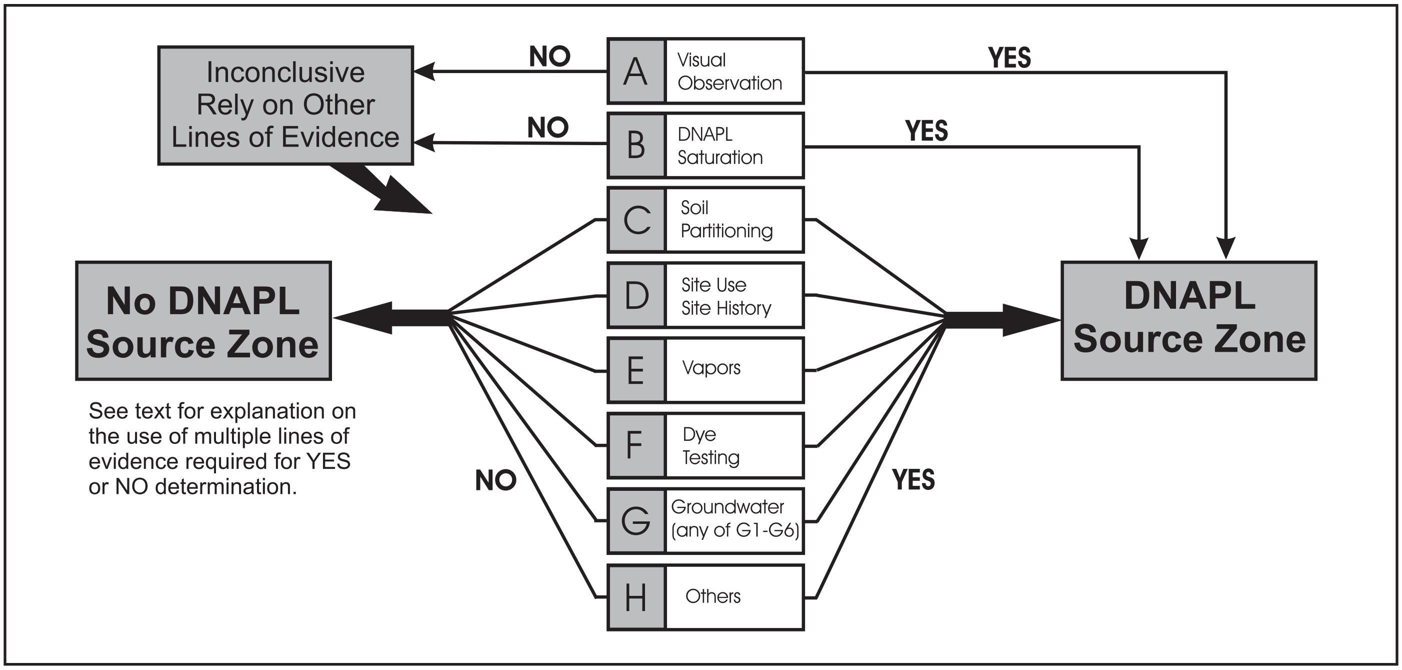

Figure 3 – Converging lines of evidence approach to assessing DNAPL presence. Methods B and C are not applicable to fractured rock.

combining to form either a positive or negative determination. Note that it is not likely that all of C through H will be satisfied at any one particular site, and that neither A nor B are necessary requirements to conclude that DNAPL is present. Most confirmed DNAPL source zones will have some of A through H determined to be negative. Because conditions vary from site to site, this document does not prescribe a specific number of lines of evidence that must be satisfied to arrive at either a positive or negative determination.

If the various lines of evidence contradict each other, it may be necessary to collect more data. It is possible that a minority of positive determinations can outweigh a majority of negative determinations if the positive lines of evidence cannot be explained without the presence of DNAPL. It should also be noted that not all sites lend themselves to collecting all of the types of data outlined here. In fractured rock, for example, soil vapor data and partitioning calculations would not be relied upon.

Evaluating the presence of DNAPL is an iterative process that incorporates new data as they are obtained. It is recognized here that certain types of data are more likely to be collected in the early stages of site investigation, while others (e.g., groundwater concentrations) can be collected on a routine basis throughout the investigation process. The fact that a number of lines of evidence are outlined in Figure 3 does not suggest that they should all be pursued at any one particular site. Site specific conditions will dictate what lines of evidence should be pursued. Care should be taken, however, to ensure that a negative response to the various lines of evidence is not simply attributable to inadequate characterization and an insufficient amount of data.

6.0 - Delineation of the DNAPL Source Zone

Depending on the spatial density of sampling points installed during initial investigation efforts, the general area within which the DNAPL resides may have been identified. Once it has been determined that DNAPL is present in the subsurface, the objectives for delineation of the source zone can be established. These objectives can vary from site to site, but typically involve one or more of the following:

- Delineation of the DNAPL source zone to ensure that the flow paths and quality of the groundwater downgradient of the source zone are monitored for the presence of dissolved-phase contaminants to assess protection of current and potential receptors.
- Delineation of the DNAPL source zone to facilitate proper design of containment systems involving groundwater extraction and/or physical barriers.
- Delineation of the DNAPL source zone to facilitate implementation of DNAPL mass removal technologies.
- Delineation of the DNAPL source zone as part of establishing boundaries for institutional controls.
- Delineation of the DNAPL source zone as part of Technical Impracticability assessments (41).

Given the selective nature of DNAPL migration, it is not feasible to determine the exact location and extent of individual DNAPL migration pathways within the overall confines of the source zone in either unconsolidated deposits, or fractured bedrock. Because data collection efforts typically involve a finite number of local-scale measurements taken at discrete locations (e.g., water quality samples, soil samples, etc.), some uncertainty will exist regarding the delineated spatial extent of the source zone.

To address the issue of uncertainty, it is recommended that both a ‘Confirmed/Probable’ DNAPL source zone be delineated, as well as a ‘Potential’ DNAPL source zone (see Figure 4). The Confirmed/Probable source zone is the volume within which compelling and multiple lines of evidence indicate that DNAPL is present. Note that what may be a compelling line of evidence at one site may not be so at another site (e.g., G2 Persistent Plume, is a stronger line of evidence in a high permeability setting than at a site where back-diffusion may dominate). The Potential source zone is of larger spatial extent, and is defined as that volume of the subsurface within which some lines of evidence indicate that DNAPL may be present, but the lines of evidence are not as numerous, consistent, or compelling as within the Confirmed/Probable source zone. Defining a Potential source zone outside of the Confirmed/Probable source zone addresses the uncertainty associated with finite amounts of data. This can be particularly useful in the hydraulically downgradient direction where it is often difficult to determine the distance to the edge of the DNAPL source zone based on groundwater quality data (e.g., using lines of evidence G1 through G6).

With respect to the various criteria for assessing DNAPL presence outlined in Section 4, lines of evidence A and B will both fall within the Confirmed/Probable source zone. All other lines of evidence (C through H) could fall within either the Confirmed/Probable source zone, or the Potential source zone. Note also that positive determinations for lines of evidence A and B are not necessary to define a Confirmed/Probable source zone. The defining feature of the Confirmed/Probable source zone is that multiple lines of evidence indicate that DNAPL is present. In practice, this will manifest itself as various lines of evidence all plotting within the same general spatial area on plan view and cross-section figures (see Figure 4 for plan view example). Within the Potential source zone, there will be fewer lines of evidence, and their occurrence may not be as contiguous as within the Confirmed/Probable source zone. Consideration should be given to known DNAPL release locations and structural aspects of the geology (e.g., dipping beds, dipping fractures) when delineating both the Confirmed/Probable and Potential source zones.

There is no prescriptive number of lines of evidence that separate the two source zone delineations. The individual lines of evidence cannot be weighted either, as the strength of the uncertainty/certainty determination is dependent on how often more than one line of evidence occurs at a particular location and how many contiguous locations have multiple lines of evidence; assigning a weighting factor to each line would negate this objectivity. Furthermore, many factors influence the transport of the DNAPL and the associated concentration of the dissolved-phase constituents such that a weighting factor could not be fairly assigned for all types of hydrogeologic environments and types of DNAPL contaminants.

The amount of acceptable uncertainty in delineating the source zone boundaries is likely to be dependent on the remedial actions considered. If hydraulic or physical containment of the DNAPL source zone were a component of the remedial actions, for example, an accurate delineation of the Potential source zone would be war-

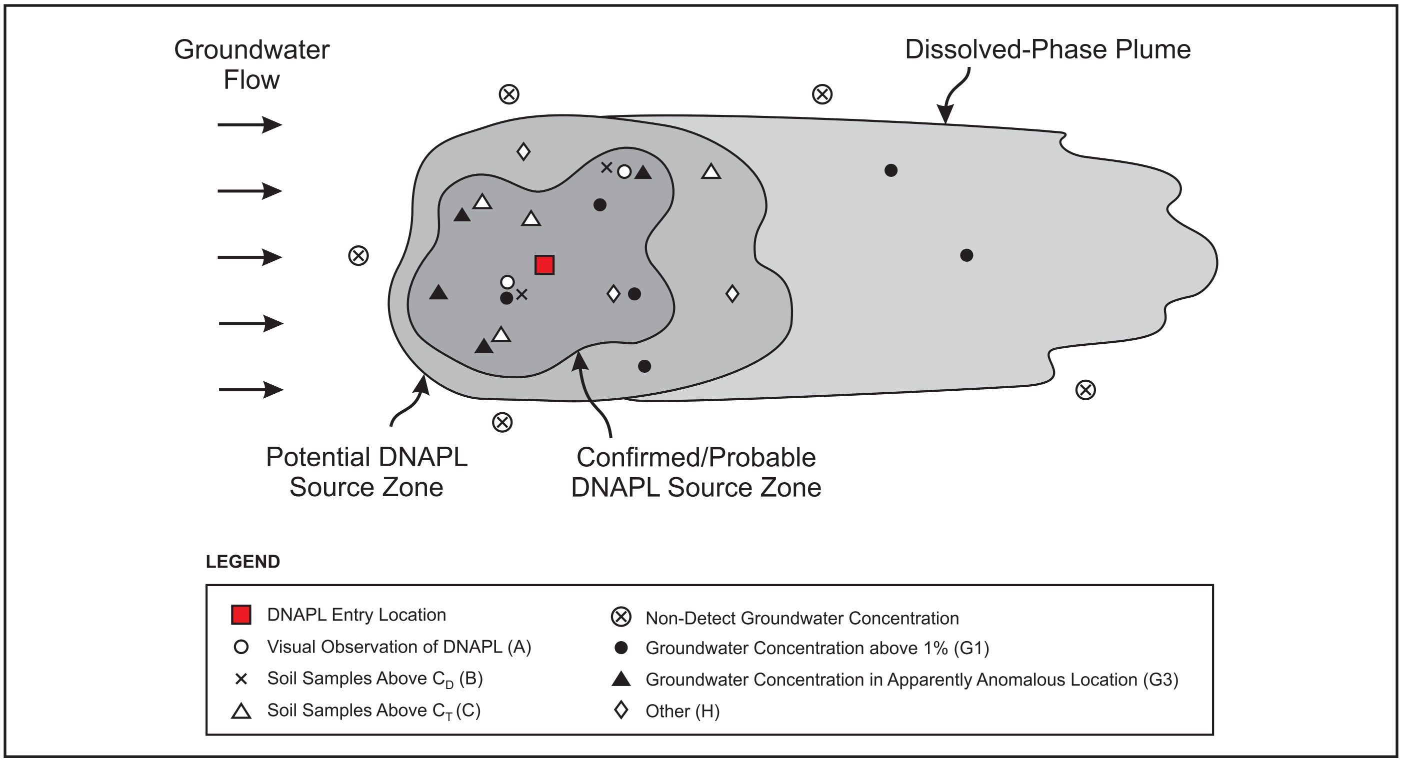

Figure 4 – Example of plan view schematic illustrating confirmed/probable and potential DNAPL source zones. Note that not all lines of evidence are depicted. Types and distribution of lines of evidence will vary from site to site.

ranted (the likely target for hydraulic containment) and accurate delineation of the Confirmed/Probable source zone may not be necessary. If the remedial actions included implementation of a DNAPL mass removal technology, however, then an accurate delineation of the Confirmed/Probable DNAPL source zone (the likely target for mass removal) would be warranted. A similar approach may be appropriate for designating a zone of technical impracticability (TI). Overestimating the size of the Confirmed/Probable source zone could overstate costs for technology application and may result in a particular technology being screened out. Underestimating the size of the Confirmed/Probable source zone, on the other hand, could lead to underestimation of costs and the perception of poor performance following completion of technology application. Monitoring points outside of an underestimated source zone may provide data showing little, if any, benefit resulting from source zone removal or treatment.

Typically, to refine the locations of the boundaries, additional drilling and sampling may be required between the Confirmed/Probable and Potential DNAPL areas. Figure 5 depicts an iterative process of data collection. Usually the degree of uncertainty in delineating these two zones will be greater in a more complex hydrogeologic environment. Although additional sampling points may be easily installed in shallow, unconsolidated materials, the same level of effort may not be feasible or may be cost prohibitive in deep fractured rock. Care must also be taken to ensure that drilling and sampling activities do not mobilize DNAPL deeper in to the subsurface. Strategies in place of extensive drilling to depth within the source zone include drilling adjacent to the suspected source zone and using lines of evidence such as G1 through G6 to infer DNAPL presence in the upgradient direction.

In all environments, the risks of potentially mobilizing the DNAPL and the associated incremental costs of additional sampling points should be compared to the benefits of increased ability to evaluate the spatial extent of the DNAPL. Additionally, site investigators should have a DNAPL Contingency Plan on hand in the field to address actions to be taken if pooled DNAPL is encountered during drilling. At some sites, it may be desirable to adopt an ‘outside in’ approach to reduce the number of invasive borings that need to be placed within the DNAPL source zone.

In addition to delineating the spatial extent of the source zone, investigators may need to assess whether or not DNAPL is still migrating within the subsurface. The assessment of mobility can be carried out using screening calculations (27) and observations such as an expanding area of lines of evidence indicating DNAPL presence. Other features of the source zone that may be of interest include the mass of DNAPL present, the mass flux downgradient of the source zone, and the relative proportions of residual versus pooled DNAPL. Calculation 1 can be used to distinguish between residual and pooled DNAPL in soil samples by selecting a saturated threshold above which DNAPL is considered pooled. Also of note is the fact that residual DNAPL will not enter monitoring wells, implying that the accumulation of DNAPL in a well indicates the presence of pooled DNAPL in the formation. Details regarding how to estimate the mass of DNAPL present in a source zone or the distribution of mass flux downgradient of the source zone, however, are beyond the scope of this document.

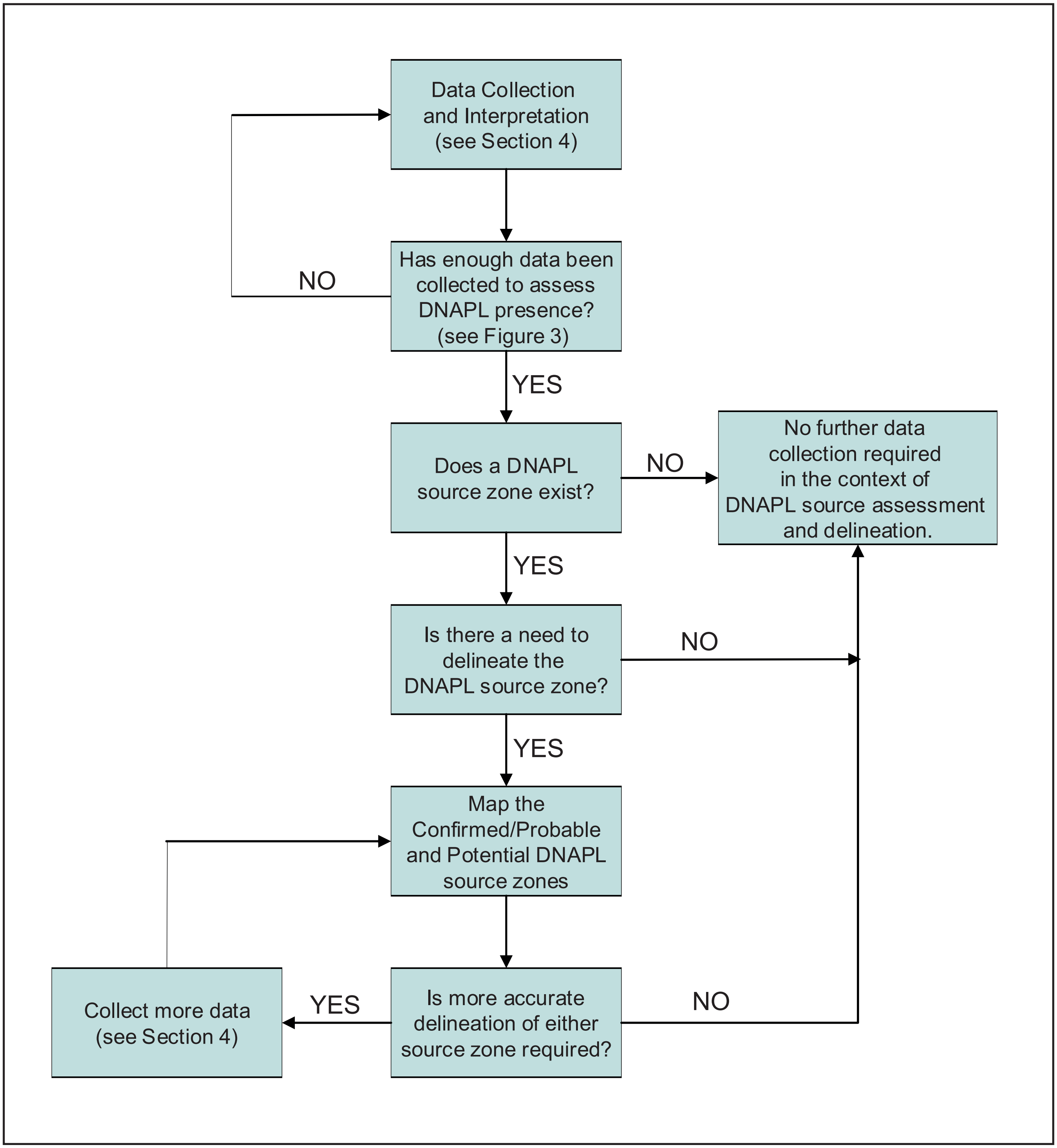

Figure 5 - Flowchart depicting iterative data collection process used in refining the DNAPL source zone boundaries.

7.0 - Glossary

Bulk Retention Capacity is defined as the total volume of DNAPL that has been retained as residual and pooled DNAPL in a unit volume of the subsurface. The bulk retention capacity accounts for the fact that not all lenses, laminations and geological units within a source zone contain DNAPL (27), and it is a function of the release history, geology and DNAPL properties. In unconsolidated media, the bulk retention capacity can be in the range from 0.005 to 0.03 (36). In fractured media, the bulk retention capacity can be in the range of 0.0002 to 0.002 (36). Fractured rock and clay cannot retain as much DNAPL per unit volume as unconsolidated deposits.

Capillary Barriers are fine grained lenses, layers and laminations upon which lateral spreading and pooling of DNAPL can occur. Even if the capillary barrier is penetrated by the DNAPL, it is likely that lateral spreading will have occurred along the top surface of the barrier prior to the capillary pressure having exceeded the entry pressure of the barrier. The finer grained the capillary barrier, the higher the pool height of DNAPL that it can support (17).

Capillary Pressure is the pressure difference between two immiscible liquids and arises because of interfacial tension. It is calculated as the non-wetting phase pressure minus the wetting phase pressure. If the DNAPL is the non-wetting phase and water is the wetting phase, for example, the capillary pressure would be the DNAPL pressure minus the water pressure.

DNAPL (Dense, Non-Aqueous Phase Liquid) is an organic liquid that is more dense than water and does not mix freely with water. A ***single-component DNAPL*** is composed of only one chemical. A ***multi-component DNAPL*** is composed of two or more chemical components.

DNAPL Source Zone The DNAPL source zone is the overall volume of the subsurface containing residual and/or pooled DNAPL. Not all portions (e.g., lenses, laminations, or fractures) of the source zone will contain residual and/or pooled DNAPL. The ***Confirmed/Probable DNAPL Source Zone*** is the part of the source zone within which it is known or highly likely that DNAPL exists. The ***Potential DNAPL Source Zone*** is the part of the source zone within which it is possible that DNAPL exists, but the lines of evidence indicating DNAPL presence are either fewer or are not as strong as those associated with the Confirmed/Probable DNAPL Source Zone.

Dissolved-phase Plume The zone of contamination containing dissolved-phase constituents resulting from groundwater flowing past residual and pooled DNAPL. The contaminants present in the plume are subject to advection, dispersion, and possibly sorption, decay, and matrix diffusion. Dissolved-phase plumes can be sustained by back diffusion from low permeability regions in the absence of DNAPL.

Effective Solubility For a multi-component DNAPL, the equilibrium solubility in water of any component of the DNAPL is referred to as the component's effective solubility. In general, the various components of a DNAPL suppress each other's aqueous solubility implying that effective solubilities are typically less than single-component (handbook) solubilities. For structurally similar compounds, the effective solubility can be estimated using Raoult's Law (2).

Interfacial Tension (IFT) is a tensile force that exists in the interface separating DNAPL and water. Because of interfacial tension, DNAPLs do not mix freely with water and exist in the subsurface as a separate liquid phase. IFT is a site-specific value that can be assessed with a simple laboratory test if a sample of DNAPL can be obtained. Literature values tend to overestimate the IFT encountered at sites. In general, higher IFT leads to more lateral spreading of DNAPL in horizontally bedded deposits, stronger capillary trapping forces, and a greater tendency for DNAPL pooling.

Mole Fraction refers to the proportion of a component, on the basis of moles, in a multi-component DNAPL. The sum of all the mole fractions is unity. Mass fractions, as provided by laboratory analysis, can be converted to mole fractions using the molecular weight of each component (see calculation 5).

1% Rule of Thumb is a generality that sampled groundwater concentrations in excess of 1% effective solubility (see Calculation 3) indicate that DNAPL may be present in the vicinity of (any direction) the monitoring point of interest. The distance between the monitoring point in question and the DNAPL source zone varies from site to site and is generally difficult to quantify with a high degree of accuracy.

Pooled DNAPL refers to local, continuous distributions of DNAPL that accumulate above capillary barriers. The capillary barriers are typically lower permeability horizons, and they can occur at any elevation in the subsurface. Within the pool, the DNAPL saturation is typically between 30% and 80% of pore space in both porous media and fractures (27). Because pools are contiguous through the pore structure they are potentially mobile and can migrate into monitoring wells, and can be mobilized by increases in the hydraulic gradient or lowering of IFT.

Raoult's Law is given by $C_i = m_i S_i$ where C_i is the effective solubility (mg/l) of component i, m_i is the mole fraction (unitless) of component i in the DNAPL, and S_i is the single-component (handbook) solubility of component i (2). This expression assumes ideal partitioning behavior and is used to estimate the maximum concentrations in groundwater immediately adjacent to residual and pooled DNAPL.

Residual DNAPL refers to disconnected blobs and ganglia of the DNAPL, trapped by capillary forces in the pore space of both porous media and fractures (21, 27, 44). The blobs and ganglia are typically from 1 to 10 grain diameters in size in unconsolidated deposits (44), and are left behind in the pathways that DNAPL has migrated through.

Residual Saturation refers to the volume of residual DNAPL present in a unit volume of pore space. Residual DNAPL saturations typically vary between 5% and 30% of pore space in both porous media and fractures (21, 27, 44).

Source Zone Architecture refers to (i) the overall shape and dimensions of the source zone, (ii) the ratio of residual to pooled DNAPL (also referred to as the ganglia to pool ratio), (iii) the lateral continuity of zones of residual DNAPL and DNAPL pools, (iv) the thickness of zones of residual DNAPL and DNAPL pools, and (v) the portion of lenses and layers containing DNAPL versus those void of DNAPL. The source zone architecture influences the downgradient dissolved-phase plume concentrations and mass flux distribution.

Wettability refers to the affinity of the DNAPL for a solid surface in the presence of water (6, 27). Many DNAPLs are non-wetting, implying that they will preferentially occupy the pore spaces within coarser grained lenses and laminations, and larger aperture fractures. Some DNAPLs are wetting with respect to water, however, implying that they will preferentially coat the aquifer materials and thereby occupy the pore spaces of the finer grained media. Coarser grained horizons and larger aperture fractures represent capillary barriers to DNAPLs that are wetting with respect to water.

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Notice

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Appendix A - Example Calculations

Note that the following calculations are generally subject to uncertainty because of input parameter variability. This variability may stem from spatial or temporal variation in site-specific conditions, or variation in textbook parameters such as contaminant chemical properties. The investigator is advised to make conservative choices with respect to input parameters and consider using a range of either measured or estimated values when performing calculations.

Calculation 1 – Chemical Concentration in Soil Corresponding to Threshold DNAPL Saturation

$$\boxed{C_D = \frac{S_r \phi \rho_N 10^6}{\rho_b} + C^T}$$

C_D = soil concentration (mg/kg) corresponding to threshold DNAPL saturation [calculated],

S_r = threshold DNAPL saturation [set between 0.05 and 0.10],

ϕ = effective porosity (unitless) [site specific measurement],

ρ_N = DNAPL density (g/cc) [site specific measurement],

ρ_b = dry soil bulk density (g/cc) [site specific measurement],

C^T = amount of contaminant (mg/kg) present in the soil sample in the aqueous, vapor, and sorbed phases [see Calculation 2 to evaluate C^T].

Example Calculation

PCE DNAPL (ρ_N = 1.62 g/cc) in a soil sample with S_r = 0.05, ϕ = 0.25 and ρ_b = 2.0 g/cc corresponds to (ignoring the C^T fraction) C_D = 10,125 mg/kg. Note that the quantity C^T is typically negligible compared to the DNAPL saturation term. The above equation is applicable to single-component DNAPLs in unconsolidated porous media. See reference (25) for the relationship between C_D and DNAPL saturation for a multi-component DNAPL. It should be noted that $0.05 \leq S_r \leq 0.10$ is suitable for geologic deposits having typical ranges of f_{oc} values (i.e., less than 2%). In general, the value of S_r should be chosen such that the resulting C_D is at least an order of magnitude higher than the C^T in calculation 2 arrived at using the highest f_{oc} value measured at the site.

Calculation 2 – Threshold Chemical Concentration in Soil Based on Partitioning Relationships (see Ref. 11)

$$\boxed{C_i^T = \frac{C_i}{\rho_b}\left(K_d \rho_b + \theta_w + H' \theta_a\right)}$$

C_i^T = soil concentration (mg/kg) threshold for component *i* [calculated],

C_i = effective solubility (mg/l) [see Calculation 3] of component *i* [calculated],

ρ_b = dry soil bulk density (g/cc) [site specific measurement],

K_d = soil-water partition coefficient (ml/g) [calculated using $K_d = K_{oc} f_{oc}$],

θ_w = water-filled porosity (unitless) [calculated from site specific measurement of moisture content],

H' = unitless Henry's constant [handbook],

θ_a = air-filled porosity (unitless) [site specific measurement],

K_{oc} = organic carbon - water partition coefficient (ml/g),

f_{oc} = fraction organic carbon (unitless) [site specific measurement].

C_i^T represents the maximum amount of contaminant *i* that can be present in a porous media sample in the sorbed, aqueous, and vapor phases without a DNAPL phase present. The calculation can be applied below the water table by setting $\theta_a = 0$. Note that the water-filled porosity and the air-filled porosity sum to the total porosity. Note also that the calculation of C_i^T is typically more sensitive to f_{oc} than it is to the porosity values.

Example Calculation

Consider a single-component DNAPL composed of TCE (C_i = 1100 mg/l, K_{oc} = 126 ml/g, H' = 0.31) in a soil sample having θ_w = 0.15, θ_a = 0.10, ρ_b = 2.0 g/cc, and f_{oc} = 0.003. The corresponding value of C^T is 515 mg/kg. For a multi-component DNAPL, a separate value of C_i^T would be calculated using the above equation for each component detected in the soil sample.

Calculation 3 – Effective Solubility Calculated Using Raoult's Law (see Ref. 2)

$$\boxed{C_i = m_i S_i}$$

C_i = effective solubility (mg/l) of component *i* [calculated],

m_i = mole fraction (unitless) of component *i* in the DNAPL [site specific measurement],

S_i = single-component solubility (mg/l) of component *i* [handbook].

Example Calculation

Consider a 3-component DNAPL composed (by mass) of 25% TCE (S_i = 1100 mg/l), 35% PCE (S_i = 200 mg/l), and 40% toluene (S_i = 500 mg/l); the corresponding mole fractions (see Calculation 5) are 0.23, 0.25, and 0.52 respectively, and the corresponding effective solubilities are 250 mg/l, 50 mg/l, and 260 mg/l respectively. Sampled groundwater concentrations in excess of 1% of any of these effective solubilities are evidence of possible DNAPL presence in the vicinity of the monitoring point. The distance to the DNAPL cannot be determined on the basis of the magnitude of the groundwater concentration alone. In cases where some of the components of the DNAPL are not known, the unknown mass fraction can be assigned an estimated molecular weight, or the average of the molecular weights of the known components.

Calculation 4 – Threshold Chemical Concentration in Soil Based on Partitioning Relationships Where Composition of DNAPL is Not Known

$$\boxed{\sum_{i=1}^{n} \frac{C^T_{obs,\,i}}{C^T_{S,\,i}} \geq 1}$$

$C^T_{obs,\,i}$ = reported concentration (mg/kg) of component *i* [site specific measurement],

$C^T_{S,\,i}$ = single component soil partitioning concentration (mg/kg) of component *i* (see C_i^T in Calculation 2),

n = number of components observed in the soil sample [site specific measurement].

For a multi-component DNAPL of unknown composition, the sum of the mole fractions must equal unity. DNAPL will therefore be present in a soil sample if sum of $\frac{C^T_{obs,i}}{C^T_{S,i}}$ exceeds unity.

Note that $C^T_{S,\,i}$ is calculated for each component in the summation using Calculation 2 with the single-component solubility as input. The presented technique can be prone to false negatives in cases where the soil sample was not analyzed for some of the components of the DNAPL. Because of this, it may be prudent in some cases to only use the calculation for demonstrating that DNAPL was present in a soil sample and not rely upon it to demonstrate that DNAPL was absent from a soil sample.

Example Calculation

The table below provides an example calculation for a soil sample in which 5 components have been detected. The sample is characterized by a porosity of 25%, a fraction organic carbon of 0.003, and a dry bulk density of 1.99 g/cc. The last column of the table sums to greater than 1.0, indicating that DNAPL was present in the soil sample.

Compound	$C^T_{obs,\,i}$ (mg/kg)	K_{OC} (l/kg)	Handbook Solubility (mg/l)	$C^T_{S,\,i}$ (mg/kg)	$\frac{C^T_{obs,i}}{C^T_{S,i}}$
Trichloroethylene	145	126	1100	554	0.262
Tetrachloroethylene	155	364	200	244	0.636
Carbon Tetrachloride	200	439	790	1140	0.175
Chlorobenzene	177	330	500	558	0.317
1,1,1-Trichloroethane	213	152	1320	768	0.277
				SUM =	**1.668**

Calculation 5 – Mole Fraction (n-component DNAPL)

$$m_i = \frac{\dfrac{ms_i}{mw_i}}{\dfrac{ms_i}{mw_i} + \dfrac{ms_{i+1}}{mw_{i+1}} + \dots \dfrac{ms_n}{mw_n}}$$

m_i = mole fraction of component *i* (unitless) in the DNAPL [calculated],

ms_i = mass fraction of component *i* (unitless) in the DNAPL [measured],

mw_i = molecular weight (g/mol) of component *i* [handbook].

Example Calculation

Consider a 3-component DNAPL composed by mass of 25% TCE (*mw* = 131.5 g/mol), 35% PCE (*mw* = 165.8 g/mol), and 40% toluene (*mw* = 92.1 g/mol). The corresponding mole fractions are 0.23, 0.25, and 0.52 respectively. In cases where some of the components of the DNAPL are not known, the unknown mass fraction can be assigned an estimated molecular weight, or the average of the molecular weights of the known components.

Calculation 6 – 1% Effective Solubility Threshold Not Knowing DNAPL Composition

$$\sum_{i=1}^{n} \frac{C_i^{obs}}{S_i} = \alpha$$

C_i^{obs} = sampled groundwater concentration (mg/l) of component *i* [site specific measurement],

S_i = single-component solubility (mg/l) of component *i* [handbook],

α = cumulative mole fraction of the sample [set],

n = number of components in groundwater sample.

Calculation assumes that the degree of borehole dilution, dispersion, and degradation is identical for each component of interest in an obtained groundwater sample. If the 1% rule-of-thumb is used, DNAPL may be present in the vicinity of a monitoring well if $\alpha > 0.01$. The procedure can be applied on a sample-by-sample basis without having to make the assumption that the DNAPL composition is spatially uniform in the subsurface. If it is believed that a value other than 1% effective solubility indicates DNAPL presence, α can be set to the corresponding value. The presented technique can be prone to false negatives where the groundwater sample was not analyzed for some of the components of the DNAPL. Because of this, it may be prudent in some cases to only use the calculation for demonstrating that α has been exceeded in a sample and not rely upon it to demonstrate that α was not exceeded in a sample.

Example Calculation

The table below presents an example calculation for 5 components. Although each component has been detected at a concentration less than 1% of S_i, the cumulative mole fractions sum to 3.4%, providing evidence of possible DNAPL presence in the vicinity of the monitoring location. If the groundwater sample is not analyzed for all components present in the DNAPL, or if any compounds are degrading in the aqueous phase, the calculation procedure will underestimate the likelihood of DNAPL presence.

Compound	C_i^{obs} ***(mg/l)***	S_i ***(mg/l)***	$\frac{C_i}{S_i}$
Trichloroethene	4.4	1100	0.004
Tetrachloroethene	1.8	200	0.009
Toluene	3.5	500	0.007
Chlorobenzene	4.0	500	0.008
Trichloromethane	48.0	8000	0.006
$\sum \frac{C_i^{obs}}{S_i}$			**0.034**

Calculation 7 – Plume Detachment Time

$$\boxed{t = \frac{LR}{v}}$$

t = time (yrs) required for contaminants to migrate through source zone of length L in the direction of groundwater flow,

v = average linear groundwater velocity (m/yr) [site specific],

R = retardation factor (unitless) for the contaminant of interest [site specific measurement – see calculation below],

L = length (m) of source zone in direction of flow [site specific measurement].

Calculation assumes unidirectional, steady-state flow conditions subject to advection and sorption only (dispersion and matrix diffusion are ignored). The calculation assumes that contaminant mass is not being added to the saturated flow system from any unsaturated zone sources (e.g., leaching and desorption). Note that R is often approximated in unconsolidated media by

$$R = 1 + \frac{\rho_b}{\phi} K_{oc} f_{oc}$$

where ρ_b is the dry bulk density (g/cc), ϕ is the porosity (unitless), K_{oc} is the organic-carbon partition coefficient (ml/g), and f_{oc} is the fraction organic carbon (unitless). Calculations considering dispersion and degradation can be found in (10).

Example Calculation

Using $L = 50$ m, $v = 25$ m/yr, and $R = 5$, the source zone should be flushed of dissolved and sorbed contaminants in approximately 10 years following the last release of contaminants. Dispersion, which always occurs, will lengthen this time as will back-diffusion, if it is occurring. In cases where complicated flow conditions exist and where it is desired to account for dispersion and back-diffusion, numerical models can be used to perform the assessment.

Calculation 8 – Conversion to Parent Compound

Daughter product concentrations can be converted to equivalent parent product concentrations by converting the daughter mass/volume concentrations to moles/volume, attributing that number of moles to the parent, and then converting the parent concentration to mass/volume.

Example Calculation

Consider a groundwater sample containing 500 ppb PCE, 400 ppb TCE, 1300 ppb cis-1,2 DCE and 44 ppb VC at a site where it is known that only PCE was released to the subsurface. It is assumed that biodegradation has not progressed beyond VC. The PCE concentration of 500 ppb is less than 1% of the PCE solubility (1% PCE solubility is 2000 ppb). Given TCE, cis-1,2 DCE and VC molecular weights of 131.5, 97.0 and 62.5 g/mol, respectively, the groundwater concentrations of these compounds are equal to 3.042E-06 mol/l, 1.340E-05 mol/l and 7.040E-07 mol/l, respectively. Assuming that each mole of daughter product derives from one mole of parent product, the equivalent total concentration of parent product is 2.016E-05 mol/l. This corresponds to an equivalent parent (PCE) concentration of 3343 ppb (PCE molecular weight 165.8 g/mol), which exceeds the 1% solubility value of 2000 ppb.

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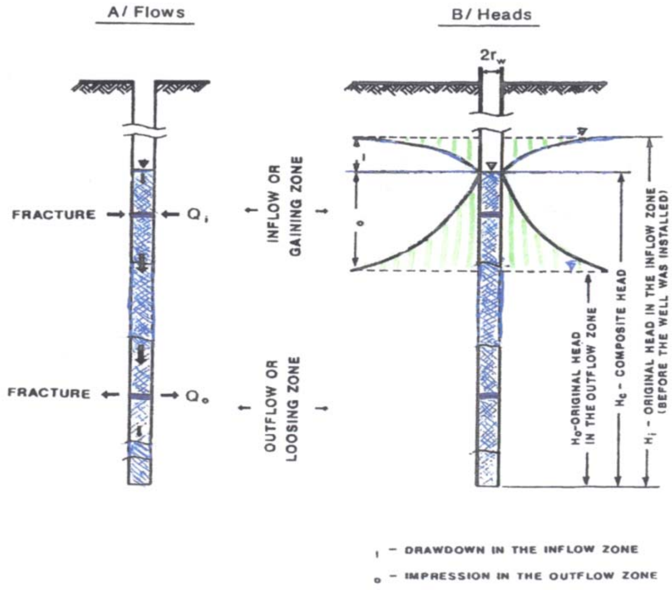
Appendix B: Open-Hole Vertical Cross-Flows Tracing Using Salt

OPEN-HOLE VERTICAL CROSS-FLOWS TRACING USING SALT

Principle and Applicability

Open-hole vertical flow tracing is conducted to measure the amount and direction of spontaneous vertical cross-flows and to determine the locations and type (inflow or outflow) of transmissive fractures involved in the cross-flows. Vertical cross-flows develop in an open hole wherever two or more transmissive fractures, with different hydraulic heads, are penetrated by an open hole. For the simplest case of two horizontal transmissive fractures within otherwise impermeable rock, with the upper fracture at a higher initial head than the lower fracture, the cross-flow hydraulics involves simultaneous inflow (or pumping) from the upper fracture and outflow (or injection) of the inflow water into the lower fracture, as illustrated below (Michalski & Klepp, 1990).

Figure B-1 Open Hole Cross-flow



For $Q_i = O_o$ (Steady-state cross-flow):

$$H_c = \frac{H_i T_i + H_o T_o}{T_i + T_o}$$

Source: Michalski & Klepp, 1990

The above formula indicates that the water level measured in an open hole represents a weighted average, or a composite of hydraulic heads (H_c) in transmissive fractures penetrated by the hole, with fracture transmissivity serving the weighing factor. This formula can be extended to cases of multiple fractures penetrated by an open hole.

Other consequences of vertical cross-flows in open holes include: 1) Occurrence of seeps above the “standing” (composite) water level, indicative that water-saturated condition extends above the “standing” water level; 2) Induced migration of NAPLs and/or dissolved contaminants, a common occurrence in bedrock domestic wells near leaking heating oil USTs; 3) Self-purging nature of open-hole segments with active upward or downward flows, with a dead flow zone also possible (generally near the hole bottom); and 4) Apparent ground water flow directions obtained from water-level measurements in longer open holes.

Because of their potential for inducing contaminant migration, long bedrock test holes should not be drilled at first through a known or suspected source zone. An outside-in approach is recommended. Targeted drilling into the source zone should be attempted only after the location, strength, and vertical extent of the source zone have adequately been assessed from results of an initial set of outside test holes and monitoring wells installed in these holes. Any bedrock supply wells located near the source zone should be tested first, because of their potential of altering the ground water flow and contaminant migration pattern, even if such wells are no longer used/pumped.

With due recognition of potential adverse impacts of open holes, salt tracing conducted in such holes can reveal and investigate transmissive fractures involved in the cross-flow. Out of numerous fractures apparent on visual logs or in cores, very few are transmissive (Michalski 1990; Morin et al. 1997). Typically, the tracing test starts with the emplacement of a small slug of saline tracer into the water column to tag the vertical flow. The movement of the slug is then tracked through collecting a series of tracer images over time using a downhole electrical conductivity probe.

The salt tracing is capable of detecting vertical flow velocities as low as 0.01 cm/min, which is approximately 200 times more precise than the reported resolution of the high-resolution heat-pulse flowmeter (Hess 1985) of 2 cm/min. The latter resolution still corresponds to an undetected vertical flow of about 130 gal/day in a six-inch-diameter well. Some commercial flowmeters may not be reliable for measuring flows below 1,000 gal/day (0.7 gpm), and are about 80 percent accurate for determining upward flow rated between 0.7 to 25 gpm (Herman 2006). The majority of bedrock wells in the Newark Basin, when tested under non-pumping conditions, show vertical flow below the resolution of such commercial flowmeters (Michalski 2010). In addition to its better resolution, flow tracing with salt provides a continuous record of vertical flow in the hole and average vertical flow velocities for selected hole segments. The vertical flow data also provides qualitative information on vertical distribution of hydraulic head within the tested open hole relative to a composite water level in the hole based on this simple rule: the head in any inflow-producing zone or fracture is higher than the composite head of the open-hole and lower in any outflow fracture/zone.

Testing Equipment and Supplies

- Downhole electrical conductivity probe capable of measuring temperature-corrected specific electrical conductance (EC). Hand-operated, small-diameter probes (e.g., Solinst TLC Meter) work well for this application. Due to the much larger size, routine geophysical logging EC probes would have a disruptive impact on tracer movement.
- Tracer release tubing consisting of a pre-cut segment of small-diameter (3/8” ID), kink-resistant PVC tubing, with a small dead-weight plumb attached to the lower end of the tubing and a funnel attached to the upper end to rest on the casing rim. Wire attached to the plumb and to a pin or nail piercing the end of tubing can provide a hinge-like connection, creating a small gap that facilitates sideway discharge of the poured tracer.
- Common salt (NaCl), approximately less than 3 to 4 oz per injection into a 6-inch diameter hole. This amount suffices to increase initial EC values upon saline slug release

5 to 10 times above the background EC, without producing any significant density-driven flow in the water column.

- Approximately 1 to 2 gallons of spring or tap water, depending on the planned depth of tracer release, a measuring cup and a mixing container.

Testing Procedure and Practical Hints

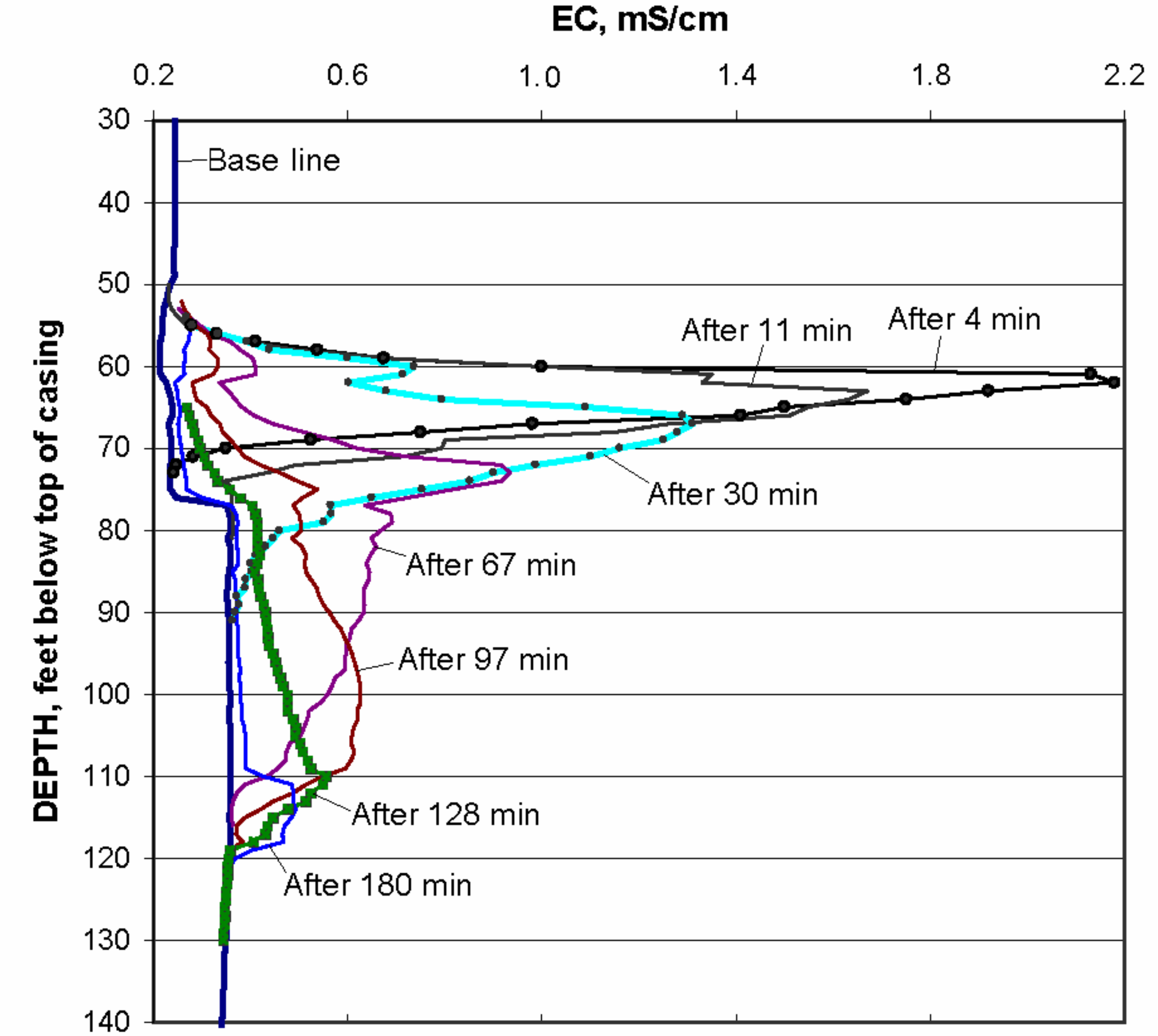
1) Obtain baseline logs of EC and temperature. Interpret these logs, and other geophysical logs (acoustic or optical televiewer, caliper, fluid temperature and conductivity) if available, to select a saline slug emplacement depth. For downward flow cases, a depth several feet below an interpreted uppermost transmissive fracture makes a good choice for the tracer release.
2) Lower an appropriate length of the tracer release tubing into the hole.
3) Prepare saline slug solution by dissolving the common salt in a small volume (0.2 to 0.5 gal) of the spring or the tap water. The small volume of saline slug will minimize transient hydraulic slug effects due to tracer release on measured vertical flow.
4) Pour the salt solution slowly into the tubing and then a pre-measured volume of spring/tap water chaser slug needed to completely displace the salt solution from the tubing. For 3/8" diameter tubing, approximately 0.22 liters of the chaser is needed per 10 feet of the tubing used. Note that the time the tracer is released into the water column is when the saline slug is completely displaced.
5) Gently pull the tubing out of the hole.
6) Lower the downhole EC probe and obtain the first image of the injected tracer through EC profiling of the water column within 5 to 10 minutes after the tracer release. Start profiling some 10 ft above the release depth and end when background EC level is reached. In addition to the depth-EC logging data, note the clock time log was started, peak EC value was reached, and the end time of the EC logging. The time data is needed to calculate vertical velocity of the tracer and the amount of vertical flow.
7) Obtain two more EC logs (tracer profiles) at 15 to 20 minutes intervals. Then select the frequency of subsequent tracer profiling based on the rate of tracer movement indicated by the earlier logs. Plotting the logged EC profiles, on paper or using a laptop, helps to select appropriate logging frequency. Maintain the same logging direction through the migrating tracer zone: Switching the logging direction during a test may introduce time-lag errors into the calculated tracer and flow velocity by virtue of the non-instantaneous nature of logging with a hand-held probe.

Interpretation of Tracing Data

To interpret the salt tracing results, plot the EC profiles representing sequential tracer images over time on one graph of EC values versus depth, as shown below in an example from Michalski (2010). Look for changes in the position, shape and size of the images as the tracer slug moves through the water column under ambient cross-flows. The direction of vertical flow (downward or upward) is determined from the sequential positions of the tracer image. In an example below, a downward flow in the hole is indicated by lower positions of the peak values and leading limbs in a sequence of tracer images between 4 minutes and 97 minutes after the tracer injection. By evaluating changes in the shape and the size of such a sequence of images, one can interpret the locations of water inflow and exit fractures, and the type of inflow or exit flow (discrete from a more prominent fracture versus diffuse from several minor fractures).

Evidence of an inflow fracture includes an increase in tracer velocity downstream of the inflow location, and sometimes a persisting indentation of the sequential tracer images at the inflow location (e.g., nicks at depths of 62 and 77 ft on the figure below). The size of sequential tracer images, determined by the areas between the baseline and the current plots, is directly proportional to the total tracer mass still remaining within the water column. An exit zone/fracture is recognized by a decreasing size and disappearance of the tracer image at the exit zone (the loss of tracer mass from the hole, e.g. at 111 and 119 ft on figure below), and by a decrease of tracer velocity past the exit location.

Figure B-2 Example of Salt Tracer Images Over Time



Source: Michalski, 2001 & 2010

Example of series of tracer images obtained from a salt tracing test. Saline tracer slug was emplaced to straddle a bedding fracture at a depth of 62 ft. Inflow fractures mapped at 62 and 76 ft, exit fractures at 110 ft and 118 ft. Downward flow was calculated at 0.35 gpm below 62 ft depth and 1.3 gpm below 76 ft depth.

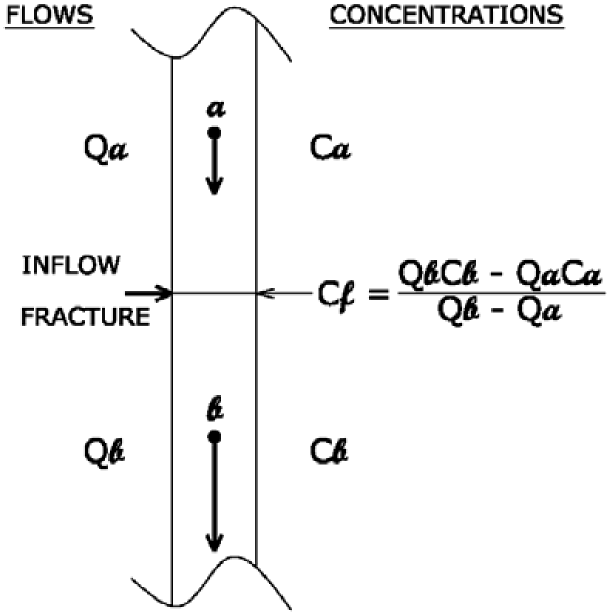
For determination of vertical flow velocity, the tested hole is first partitioned into segments defined by locations of inflow and exit fractures. Local tracer velocity values are then calculated for each segment by dividing the vertical distance travelled by the same reference point on any two adjacent tracer images and the elapsed time difference, as shown in an example calculation below. A reference point selected on an upstream limb of the image near the EC peak value makes a good choice. Vertical flow is then calculated as a product of the local velocity and a unit volume of the open hole (1.47 gal/ft for a 6-inch diameter hole). If caliper log is available, one should check for hole diameter changes within the tested segment, as they may impact the measured vertical flow velocities, so that apparent inflow or exit zones due to hole diameter change are recognized.

Appendix C: Depth-Discrete Sampling and Profiling of Vertical Cross-Flows

DEPTH-DISCRETE SAMPLING AND PROFILING OF VERTICAL CROSS-FLOWS

Principle and Applicability

A logical next step after the salt tracing, depth-discrete sampling of vertical cross-flows permits a rapid screening-level assessment of contaminant concentrations in individual inflow fractures identified through the salt tracing. For a given open-hole segment straddling an inflow-producing fracture, contaminant concentration in this inflow fracture can be resolved based on the measured vertical flows above (Q_a) and below (Q_b) the inflow fracture and contaminant concentrations in flow samples collected above (C_a) and below (C_b) the inflow-producing fracture, using the following formula:



$$C_f = \frac{Q_b C_b - Q_a C_a}{Q_b - Q_a}$$

Source: Michalski 2001& 2010

This formula is derived from the principle of contaminant mass preservation assuming a perfect mixing and a steady-state character of vertical cross-flow in the hole. The setup is comparable to determining the contaminant concentration in an inaccessible tributary stream based on the measured flows and contaminant concentrations in the main river at sections located upstream and downstream of the confluence. The depth-discrete analytical samples should be collected outside a mixing zone of water from the inflow fracture with water moving through the hole. The sampling should also be conducted in a way that limits disturbance to the cross-flows. A low-flow sampling technique is suitable for this sampling, as purging is limited to the pump and connecting tubing.

Equipment and Supplies

- A low-flow submersible pump;Sampling tubing, ¼” or3/8” ID;
- Calibrated bucket and EC probe (optional);
- Sampling vials/bottles.

Sampling Procedure

1) Select the depths of collection of depth-discrete samples (usually above and below identified inflow fractures – but outside the mixing zone - and at the top of the water column, if this top is below the casing bottom).
2) Measure an appropriate length of sampling tubing for the deepest sample, and mark all of the selected sampling depths on the tubing. Attach the submersible pump.
3) Lower the pump with the tubing to the lowest sampling depth.
4) Start pumping at a low rate (<<1 gpm) to purge approximately 2-3 volumes of the tubing. Check for stabilization of EC values in the discharge at the end of the purging period and compare the stabilized EC value against an EC value obtained on the baseline EC log at a given sampling depth (if available).
5) Once purging of the tubing is completed, immediately collect analytical samples from the discharge tubing.
6) Raise the pump to the next higher sampling depth, cut-off excess tubing, and repeat Steps 4, 5 and 6 for the remaining sampling depth.
7) Arrange for a quick turnout (<48 hrs) of analytical results for the collected samples.

Interpretation of Results

The depth-discrete sampling, if conducted after completion of in-well flow tracing, offers an inexpensive tool for a speedy determination of vertical distribution of contaminants within inflow segments of the test hole. The calculated contaminant concentrations for inflow fractures using this tool are generally comparable with concentration obtained using a standard straddle packer sampling of the same inflow fractures. An early determination of vertical contaminant profiles in test holes aids in identifying contaminant migration pathways and locating the source zone(s) of detected contamination. No samples need to be collected from segments with exit/outflow fracture zones; as such samples would not be representative of native water quality in the exit fractures away from the hole.

Together with the measured vertical cross-flows, the results of depth-discrete sampling allows one to quantify contaminant migration (flux) through the hole. Based on the measured contaminant flux, a decision is made whether the test hole may be kept open or the migration needs to be stopped by inflating a temporary packer or a flexible borehole liner.

Appendix D: Packer Testing in Bedrock Aquifers

PACKER TESTING

Principle and Applicability

The utility of straddle/dual packer tests is often limited to sampling of packer-isolated test zones, but the testing can easily be expanded to include hydraulic head-measurement (head profiling), transmissivity determination for individual transmissive fractures, and short-term pumping tests when existed monitoring wells serve as observation wells. The expanded testing provides an opportunity for a complete characterization of the packer-isolated bedding fractures (aquifer units).

Equipment and Supplies

- A complete packer test setup includes three pressure transducers that are set 1) within, 2) directly above and 3) directly below the packer-isolated interval that contains a transmissive fracture; a submersible pump with discharge tubing and in-line flowmeter, if available; data loggers;
- Water-level meter and physical parameter meters;
- Water sampling bottles;
- Pressure transducers deployed in existing monitoring wells, if the testing is also used as short-term pumping tests.

Testing and Sampling Procedure

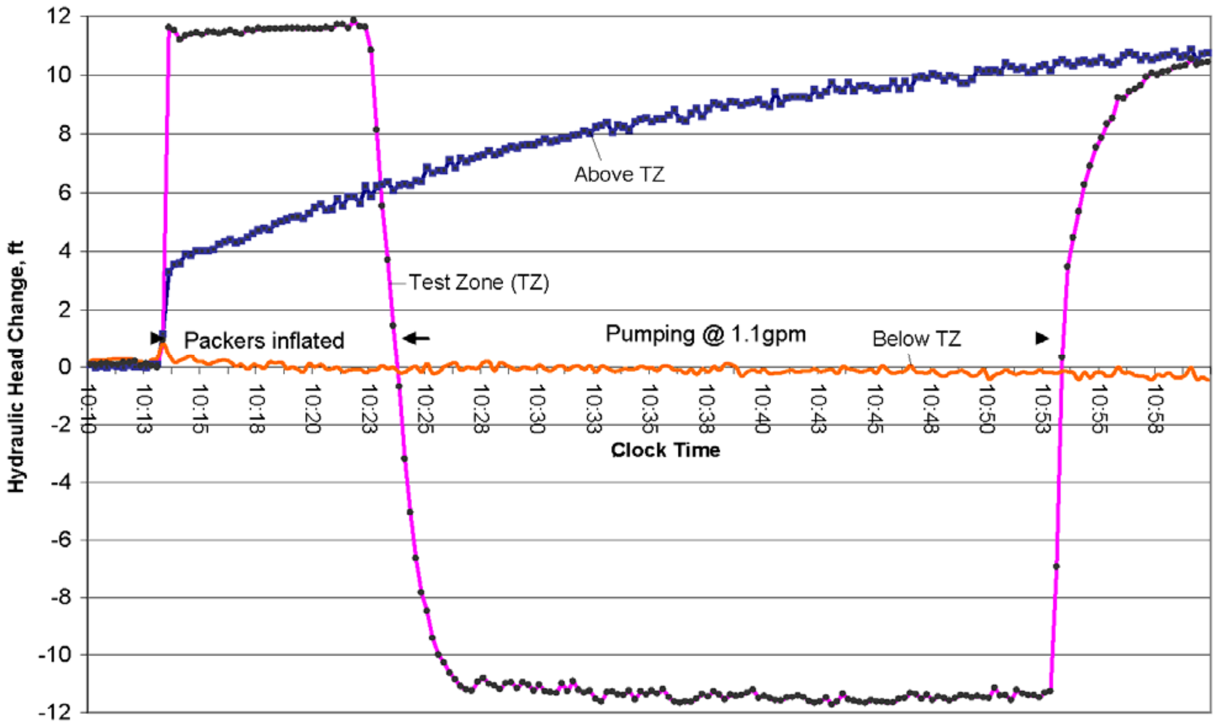
1) Lower the packer assembly against the first intended test interval and allow water level in the water column to stabilize. Synchronize data loggers with the laptop clock.
2) Record the final stabilized water level in the water column and the initial transducer readings. Then re-set data loggers pressure readings to zero for all three transducers.
3) Program a new test to record pressure readings every 10 to 30 seconds. Begin logging transducer readings and check their stability for about 5 minutes.
4) Inflate packers and allow time for transducer readings to stabilize in the test zone. This stabilized reading will be used to determine the hydraulic head in the test zone.
5) Select an initial pumping rate based on the results of the saline tracer testing and/or flow observations during drilling. Determining an appropriate pumping rate may require trial and error.
6) Conduct constant pumping rate/purging test at the sustainable pumping rate. Purge approximately three test interval volumes. Verify the stabilized pumping rate several times during the pumping period, using the flowmeter readings and a calibrated container. Also, measure the physical parameters (at least EC and T) of the discharged water.
7) Proceed with ground water sampling directly from the discharge, once the appropriate volume of water is purged and/or EC and T in the pumped zone have stabilized. See *Important note on packer sampling* at the end of this Section. For the sampling period, pumping rate should be reduced to <1gpm.
8) Turn off the submersible pump to begin the recovery phase. Continue to monitor water levels until the level in the pumped zone returns to 90 percent of the pre-pumping level.

9) Deflate the packers, stop logging/recording and save the logging file.
10) Move the packer assembly to the next test zone. Repeat Steps 2 through 9.

Interpretation of Results

An example pressure transducer records from a packer test isolating a minor bedding fracture is provided below:

Figure D-1 Pressure Transducers Record for an Inflow Bedding Fracture



Source: Michalski, 2010.

All three pressure transducers were zeroed prior to inflating the two packers at 10:14 to better observe head relations between the three zones. Upon inflating, the head in the test zone increased by nearly 12 ft above an ambient water level in the open hole. Constant rate pumping from the test zone started at 10:23 and continued for 30 minutes. Note that this pumping had no impact on the transducer readings above and below the test zone, indicating its hydraulic confinement (practically no leakage). The transducer readings above the test zone continued to rise, responding to continued water buildup above the upper packer after suppression of prior downward flow. Slightly negative readings below the test zone reflected a new composite head in fractures below the lower packer. This test zone straddled an inflow fracture identified at a depth of 62 ft. Main outflow fracture in this test hole was at a depth of 110 ft .

A stabilized reading of the test-zone transducer upon inflation of the packers (but prior to any pumping) corresponds to the hydraulic head value in the fracture tested relative to an ambient water level in the test hole measured just prior to the inflating of the packers. This ambient head represents a composite of the heads in all transmissive fractures involved in the cross-flows. In the example above, the head in the inflow zone tested was nearly 12 ft higher than the (composite) head of the open hole.

A constant-rate pumping of the packed-off fracture interval, performed as part of a purging routine prior to sampling, is used to calculate the transmissivity (T) of the fracture-straddling length of test interval (L) in a given radius (r) of the test hole, and the measured stabilized drawdown value (s) at the applied constant pumping rate (q), based on the following form of the steady-state (Thiem) formula (National Research Council 1996; Cedergren 1989; p. 53):

$$T = (q/2\pi s) \ln (L/r).$$

In addition to the transmissivity and head data, the packer test record can provide information on hydraulic relations of the packer-isolated bedding fracture to other transmissive bedding fractures (aquifer units), including degree of vertical confinement or leakage, and position of a controlling fracture within the hole (below or above a given test zone). The lateral continuity of the packer-isolated fractures is verified through observing hydraulic impacts of the packer tests in existing monitoring wells open to specific aquifer units.

Important note on packer sampling: While packer samples from inflow-producing fractures are representative of native water quality, those from outflow/exit fractures generally are not because of the quantities of water injected into such fractures by cross-flows from the time of borehole drilling till the packer testing. Routine purging of a packer-isolated outflow zones usually cannot remove enough water to eliminate prior cross-flow impacts. If this tenet is not accounted for, contaminant concentrations obtained from packer testing of an exit fracture will be typical of water quality of the cross-flow that had exited through this fracture rather than of native water quality in the receiving fracture. The results of vertical flow measurements/tracing and depth-discrete sampling can be used to identify exit fracture zones and quantify the volume of water and associated flux of contaminants that had entered a given exit fracture prior to the time of packer testing. True native water quality of the receiving zone can be determined after the test hole is converted into a short-screen monitoring well open to this zone and the cross-flow impacts will have dissipated.

Appendix E ACRONYMS

ACRONYMS

AOC	area of concern
CEA	classification exception area
CPT	cone penetrometer
DNAPL	dense nonaqueous phase liquid
DPT	direct push technologies
EC	electrical conductance
FID	flame ionization detector
FSPM	Field Sampling Procedures Manual
GWQC	Ground Water Quality Criteria
GWQS	Ground Water Quality Standards
IGWSRS	Impact to Ground Water Soil Remediation Standard
IRM	Interim Remedial Measure
ITMS	ion trap mass spectrometer
ITRC	Interstate Technology and Regulatory Council
LNAPL	light nonaqueous phase liquid
LSRP	Licensed Site Remediation Professional
MIP	membrane interface probe
MNA	monitored natural attenuation
NAPL	nonaqueous phase liquid
N.J.A.C.	New Jersey Administrative Code
NJDEP	New Jersey Department of Environmental Protection
NJGS	New Jersey Geological Survey
N.J.S.A.	New Jersey Statutes Annotated
OPRA	Open Public Records Act
PID	photoionization detector
SRRA	Site Remediation Reform Act
UIC	underground injection control
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
UST	underground storage tank
UV	ultra violet
VOCs	volatile organic compounds