Technical guidance - Protocol 7

Groundwater investigation and characterization

Technical guidance for use of Protocol 7

Groundwater investigation and characterization

Note: This guidance document does not supersede the *Environment Act*, its regulations or protocols. It does not list all provisions related to contaminated sites. If there are any difference or omissions in this guidance document, the *Act* and regulations apply. This guidance document is intended to provide additional information in support of the Protocol 7 requirements.

List of acronyms

AEC   area of environmental concern

APEC   area of potential environmental concern

AW   aquatic life water use standards

CPT   cone penetrometer test

CSM   conceptual site model

CSR   Yukon *Contaminated Sites Regulation*

DNAPL   dense non-aqueous phase liquid

DQOs   data quality objectives

DW   drinking water use standards

ESA   environmental site assessment

IW   irrigation water use standards

LIF   laser-induced fluorescence

LNAPL   light non-aqueous phase liquid

LW   livestock water use standards

MIP   membrane interface probe

NAPL   non-aqueous phase liquid

PCOC   potential contaminant of concern

QA/QC   quality assurance/quality control

VOCs   volatile organic compounds

Definitions

Aquifer: a geologic unit that is water bearing and capable of storing, transmitting and yielding water.

Aquitard: a geologic unit that is less permeable than an aquifer.

Direct push: technologies that acquire information on subsurface conditions by pushing a probe or tool directly into the ground without the need to remove soil as the tool is advanced.

Duplicate samples: two consecutive samples collected at the same location and time by the same person, and using the same equipment and procedures.

Dissolved-Phase plume: region of an aquifer occupied by dissolved constituents in groundwater emanating from a source of contamination.

Field blank: aliquots of analyte-free reagent water that are sent from the laboratory to the field, where the seal is broken and the sample is handled identically to a genuine sample.

Groundwater: all water **in a zone of saturation** beneath the land surface, regardless of its origin.

Monitoring well: a well that (a) is used or intended to be used for the purpose of ongoing monitoring, observing, testing, measuring or assessing (i) the level, quantity or quality of groundwater, or (ii) subsurface conditions, including geophysical conditions, and (b) is not used or intended to be used for the purpose of (i) exploring for or extracting groundwater for use, or (ii) injecting water or any other substance into groundwater on an ongoing basis.

Non-Aqueous phase liquid: a liquid that is immiscible in water and forms a separate phase when in contact with water.

Plume: a region of ground occupied by groundwater contamination emanating from a source of contamination.

Potential contaminant of concern: A contaminant that might be expected to occur at a site based on known spills or historical site use.

Qualified professional: means an applied scientist or technologist specializing in a particular applied science or technology including, but not limited to agrology, biology, chemistry, engineering, geology, or hydrogeology and:

a)   who is registered with an appropriate professional organization, acting under that association's Code of Ethics and subject to disciplinary action by that association.

b)   who, through suitable education, experience, accreditation and knowledge, may be reasonably relied on to provide advice within his or her area of expertise.

Quality assurance: comprises a range of management and technical practices to ensure that the data are of adequate scientific credibility to permit interpretations that lead to acceptable management decisions.

Quality control: describes specific formal goals, such as data quality objectives (DQOs), methods for collection of data to assess data quality, methods to assess data quality, and remedial measures to be taken when DQOs are not realized.

Source zone: a region of soil or rock containing contaminants that may act as a source of contamination to groundwater.

Trip blanks: aliquots of analyte-free reagent water that are sent from the laboratory to the field, and are later returned to the laboratory together with genuine samples. The seals remain unbroken in the field.

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1.0 Introduction

This document contains guidance for use by qualified professionals for the investigation and characterization of groundwater at sites in Yukon in relation to contaminated sites, land treatment facilities and solid waste management facilities.

The guidelines and procedures outlined here are not applicable at every site; others may also be used. However, regardless of the procedures used, it is the responsibility of the site owner or operator to retain a qualified professional with demonstrable experience to ensure that the groundwater is properly characterized and remediated while adhering to applicable Yukon laws, regulations, and standards.

Key questions that are addressed by this guidance are as follows:

- When is groundwater investigation necessary?

- What water uses are defined for groundwater in Yukon?

- What is a conceptual site model?

- How do you design a field program to investigate groundwater?

- How detailed does a groundwater investigation need to be?

- What methods and approaches are acceptable to use in Yukon?

- What level of groundwater characterization is required during site investigation and site assessment, as defined in the Yukon CSR?

- What level of groundwater assessment is required for confirmation of remediation?

- What is required for post-remediation groundwater monitoring?

- When and how do you de-activate or close monitoring wells?

Suggested methods and approaches for acquiring field and laboratory data are provided in Appendix A, *Field and Laboratory Data Acquisition - Methods and Approaches*. The Appendix

includes a discussion on monitoring well development, well purging and sampling, and acceptable groundwater sampling approaches and methods.

A summary table of **Groundwater Investigation Methods** is provided in Appendix B. Direct and indirect approaches to acquire groundwater quality information are compared including, for example, sampling from conventional monitoring wells versus sampling using direct-push technologies.

A **Groundwater Investigation Checklist** for site assessments is provided in Appendix C. The Appendix highlights many of the important features of a comprehensive groundwater investigation program. External reviewers and others using the checklist to assess the completeness of an investigation should also consider site-specific factors and information developed as part of previous investigations.

2.0 Requirement for groundwater investigation

2.1 When is groundwater investigation necessary?

The Yukon *Contaminated Sites Regulation* (CSR) under the *Environment Act* contains requirements to ensure that groundwater at a site is suitable for direct use, based on groundwater use at the site, and is of adequate quality to protect adjacent groundwater users. Where a groundwater investigation is necessary to ensure these requirements are met, section 8 (1) of the CSR requires that a site investigation, also commonly referred to as a Phase I ESA, be undertaken to include any information as to what contaminants may occur on the site. The site investigation comprises a review of historical and current practices on the site.

If potential issues are identified, a subsequent site assessment, commonly referred to as a Phase II ESA, is conducted as per Section 9 of the CSR and sections 114 (6)(d) and 115(1) (d) of the *Environment Act*. In the site assessment, relevant environmental media are sampled for the potential contaminants of concern (PCOCs). If contamination is identified then subsequent assessments must be undertaken in accordance with Section 9(c) and 9(g)(ii) of the CSR to define the extent of contamination, and to provide information necessary for conducting a risk assessment, if applicable, and developing a remediation plan.

Groundwater investigation is necessary where groundwater may be present for direct use (i.e., groundwater is suitable for drinking, irrigation and/or livestock or groundwater flows to an aquatic receptor) and the potential exists for the quality of groundwater to be unsuitable for direct use. *In practice, where areas of potential environmental concern (APECs) are identified by a site investigation (Phase I ESA), relevant environmental media will include groundwater, which should be sampled as part of the subsequent site assessment (Phase II ESA).* If groundwater is not sampled, detailed supporting rationale for its exclusion must be provided in the site assessment report.

2.2 Yukon's Water Use Standards

The CSR defines the following types of water use, which are applicable to surface water and groundwater:

- DW—drinking water

- AW—aquatic life protection

- IW—irrigation water

- LW—livestock watering

AW standards are further subdivided into standards to protect freshwater aquatic life and standards to protect marine and estuarine aquatic life.

Numerical standards for many potential contaminants in water are listed in Schedule 3 of the CSR. Monitoring results may be compared directly to these schedules to determine whether a groundwater source may be used directly for the intended use. Guidance for selecting water uses is provided in Protocol No. 6: Application of Water Quality Standards.

2.3 Groundwater and Aquifers in Yukon

Groundwater is water that occurs beneath the surface of the ground. Almost always, groundwater that is used (groundwater is some areas may not be suitable for use) in Yukon is extracted from water-saturated, permeable geologic units. These units are referred to as aquifers when they are capable of storing, transmitting and yielding significant quantities of water. Typically, aquifers can yield water in a usable quantity to a well, spring or stream. They may be composed of unconsolidated materials (e.g., typically sands and gravels) or consolidated materials (e.g., fractured bedrock).

The majority of Yukon is covered by mountainous terrain of the Canadian Cordellera and consists of north-south trending mountain ranges, intermontane valleys, plateaus and lowlands. Aquifers can occur in alluvial settings in valley bottoms and in upland bedrock. Aquifers consisting of alluvial or colluvial deposits in valley bottoms and lowland areas may be confined or unconfined.

Unconfined aquifers of fluvial, glacial or glacial-fluvial origin are common and may have a direct connection with surface water when located adjacent to rivers, streams and lakes and be vulnerable to contamination. Confined aquifers may develop high artesian heads in inter-montane valleys or beneath permafrost when connected to adjacent elevated bedrock systems. Flow in bedrock aquifers is primarily through fracture systems such as bedding planes, joints and faults. Water quality varies throughout the Territory due to variation in the type of bedrock, composition of overburden and presence of permafrost. Precipitation is generally low and groundwater recharge is greatest in late spring to early summer during snowmelt. (Wei et al., 2014). Groundwater is an abundant resource in some areas of Yukon and 97% of the population utilizes groundwater for drinking water. For example, the Riverdale (Selkirk) aquifer, a sand and gravel aquifer, supplies the municipal drinking water system in Whitehorse.

Permafrost, ground that remains frozen for more than two consecutive years, is continuous over much of Yukon north of Dawson while in the south it becomes sporadic and underlays less than 25% of the land area (Bonnaventure et al. 2012). The presence of permafrost can create significant challenges to accessing groundwater resources. Sub-permafrost groundwater in the thawed zone beneath permanently frozen ground can be a source of groundwater supply in permafrost impacted areas, but well installation in permafrost can be problematic due to wells freezing or potentially high groundwater pressures beneath the permafrost.

Further information and discussion on groundwater and water wells in Yukon is provided by the Department of Environment, Water Resources Branch (Government of Yukon, 2013 a,b).

3.0 Groundwater investigation during the site investigation

Groundwater underlies virtually every site. Groundwater investigation should begin during site investigation and should include attempts to acquire geological, hydrostratigraphic and groundwater use information about the site and vicinity (refer to Canadian Standards Association standard Z768-01, Phase I Environmental Site Assessment for guidance on completing a site investigation). If APECs are identified, then the assembled information should be evaluated to assess the potential for contamination of the environmental media, including groundwater. To assist in this evaluation, a Conceptual Site Model (CSM) should be developed with a hydrogeologic focus. The same CSM will also serve to plan the subsequent initial phase of the field investigation, the site assessment.

3.1 The hydrogeologic conceptual site model

Where even the simplest site investigation is undertaken, it is critical for effective planning that the various historical, physical, chemical, and biological components that define a problem are drawn together into a conceptual site model (CSM). In a hydrogeological context, the CSM should comprise a three-dimensional understanding of the physical site setting that spans the depth and breadth of the area to be investigated and should include key hydrogeologic features and properties. The CSM should, as a minimum, include:

- The *physical geologic and stratigraphic setting* including the known or inferred thickness, extent and continuity of all relevant geologic and stratigraphic units of interest.

- Hydrostratigraphic units including all *aquifers and aquitards* of relevance that are beneath and in the vicinity of the site and that define the local and regional groundwater flow systems.

- *Groundwater levels* (pressure heads, water-table elevations, potentiometric surfaces) and *hydraulic gradients* (vertical and lateral) within and between each groundwater flow system of interest.

- The *physical and hydrogeologic boundaries* that define the groundwater flow systems of interest, including recharge and discharge areas, pumping wells, hydraulic or physical no-

flow boundaries or divides, boundaries of constant hydraulic head or flux, and other relevant conditions such as locations and depths of permafrost.

- The locations and character of on-site and relevant off-site *potential source zones* of contamination to groundwater, including any non-aqueous phase liquids (NAPL) if present, and their composition, nature, and extent.

- The locations, extent and character of associated *dissolved-phase plumes* of contamination that may presently exist.

- All *pathways for contaminant transport* from source zones to known or potential receptors, including pathways that may convey non-aqueous phase, dissolved-phase or vapour-phase plumes that may be expected to develop in the future.

3.1.1 Unconsolidated deposits

In most cases, geologic materials of relevance that should be described in the CSM will comprise unconsolidated deposits of gravels, sands, silts, clays, tills, etc. In a hydrostratigraphic context, the more permeable materials with similar hydrogeologic properties may behave as aquifers whereas less permeable materials may behave as aquitards (Figure 1). Where aquitards are present, the possibility of secondary porosity features within the materials should be considered, such as vertical fractures and root holes. Such features may significantly affect the transport of contaminants by allowing vertical penetration into and sometimes through the otherwise low permeability material, and diffusion into the material matrix.

Geologic Units   **Hydrostatigraphic Units**
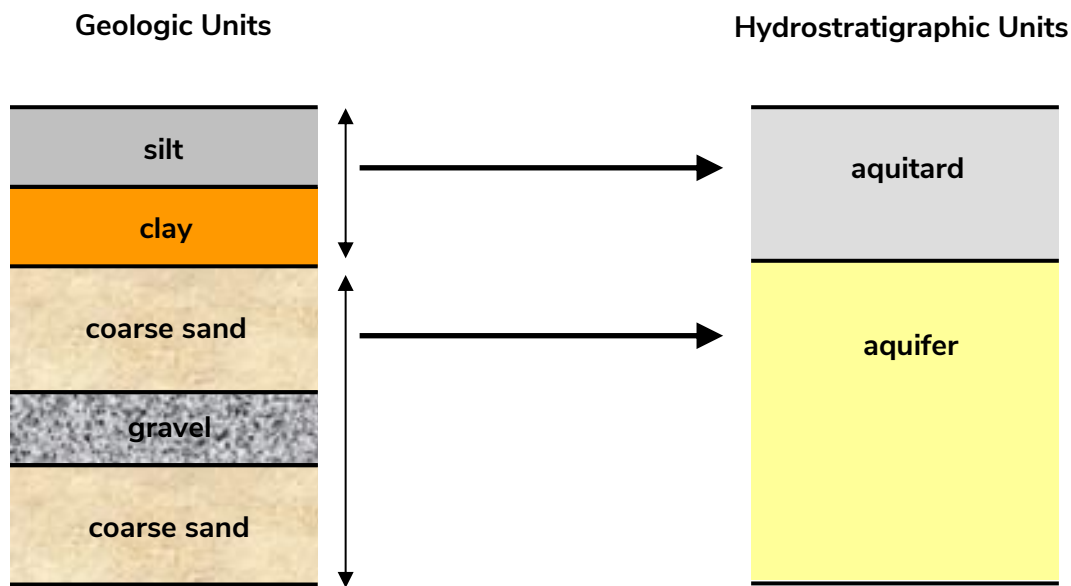
Figure 1: A hydrostratigraphic unit

Notes: Comprises one or more geologic units with similar hydrogeologic properties (i.e., hydraulic conductivity).

3.1.2 Bedrock aquifers

Bedrock serves as an important groundwater resource in many parts of Yukon, although most bedrock wells are installed for use by individual residences and not for community supply wells. Groundwater wells installed in bedrock commonly intercept a network of interconnected water-bearing fractures within the rock. The properties of bedrock aquifers can vary widely, ranging from granitic, sparsely fractured rock to highly fractured limestone deposits where fracture openings and rock partings may be enlarged by dissolution (commonly referred to as karst deposits). In most bedrock aquifers, the groundwater migrates primarily through discontinuities (fractures and joints) in the rock matrix or in formations with significant secondary porosity (e.g., in vuggy limestone).

The groundwater investigation methods described in this guidance are, to varying degrees, applicable to fractured bedrock settings. Unlike unconsolidated aquifers, groundwater velocities can be rapid through fractured bedrock and the influence of pumping may be observed over large

areas. Where bedrock may be of interest to a site investigation, bedrock properties should be described in the CSM, including the rock type, rock quality designation (RQD) and competency, degree of weathering, presence of secondary porosity, presence, extent and connectivity of the fracture network, and presence and degree of infilling of fractures. Specialized drilling and borehole logging technologies and test procedures may be necessary to adequately describe bedrock and hydrogeologic properties of relevance to the site investigation, and specialists in this area should be consulted, where warranted.

3.1.3 Describing the conceptual site model

The CSM must be supported with a data set derived from sound practices, as provided in this and other guidance documents, and any pre-existing information or data that have been identified and relied upon. Supporting rationale should be provided for such data. Depending on the level of site complexity, the CSM may be portrayed together with descriptive text as a set of plan maps and stratigraphic cross sections or fence diagrams. Three-dimensional visualization software is often an effective means to visually describe more complex sites. Examples of drawings used to assist in developing conceptual models for sites contaminated by light and dense NAPL are presented in Figure 2. As part of the CSM, key processes that may significantly affect or influence the presence, distribution and fate of the PCOCs at the site (e.g., advection, dispersion, retardation, ion exchange, precipitation, dissolution, diffusion, volatilization and biodegradation) should be identified and described where available data make such assessments possible.

A
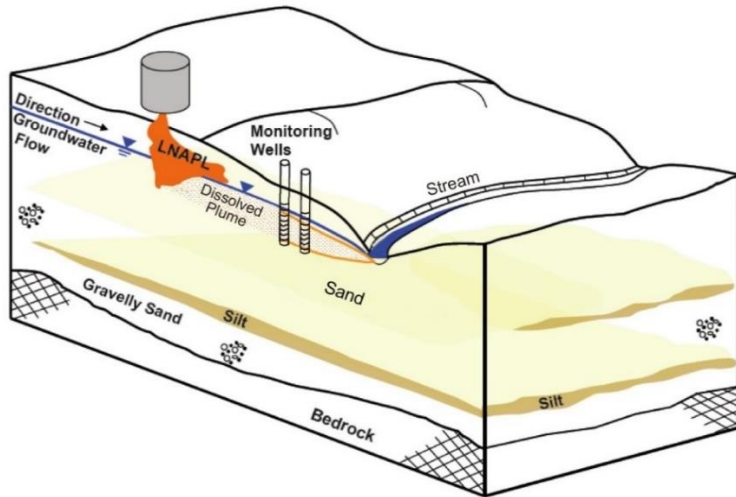
B
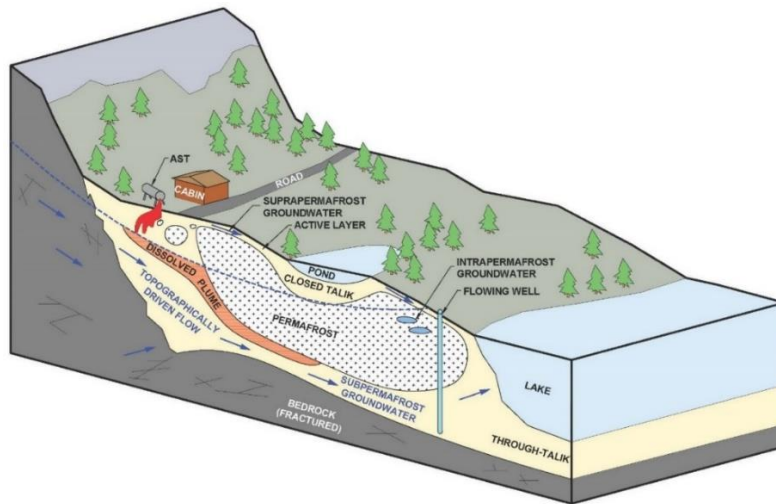
Figure 2: Examples of conceptual site models
Notes: A - conceptualization of LNAPL site. B - conceptualization of a contaminated site in permafrost conditions

4.0 Groundwater assessment during the site investigation

The type and scale of investigation selected and implemented during the site assessment, and the media to be sampled, will be highly dependent on local site conditions and on characteristics of the potential contaminants of concern (PCOC). The success of the program will also depend on the degree of understanding of these conditions as described by the CSM. A well-developed CSM is likely to better achieve site assessment objectives than a program based on limited information that has not been cohesively assembled within a logical framework. Where site-specific information is lacking but groundwater quality is to be addressed, the site assessment could consider the drilling and logging of a "stratigraphic" borehole, located beyond all zones of potential contamination, to establish site-specific stratigraphic conditions and to identify target intervals for further testing (e.g., water table, saturated geologic units, aquifers).

A range of direct and indirect approaches are available to acquire information on groundwater quality (e.g., Health Canada, 2008). ***The preferred approach should include the acquisition and analysis of representative groundwater samples.***

4.1 Designing the field program

Almost all groundwater investigations will include an intrusive field program that will typically involve drilling, hydrologic monitoring and groundwater sampling. Depending on the nature of the contamination and the physical setting, non-intrusive assessments using techniques such as electromagnetic geophysical surveys may also prove invaluable in establishing the extent of contamination. However, follow-up intrusive programs to acquire groundwater samples for verification and longer term monitoring are also usually required.

The types of data and the manner in which the data are acquired for a particular site setting will be constrained by factors such as the depth to the water table, soil density and consistency, competence of bedrock where present, and other factors. Consequently, the optimal approaches for data collection (e.g., use of conventional drilling rig technologies versus direct-push technologies) and the best technology to use (e.g., the type of drilling rig, continuous coring versus discrete sampling, depth profiling of soil or groundwater concentrations, surface geophysics) will likely vary among settings, and among sites falling within similar settings.

In some circumstances in Yukon where only a shallow soil horizon without groundwater is present above a competent bedrock, drilling extensive depths to attempt to find groundwater will not result in meaningful data related to confirming the absence of contamination. Vertical groundwater transport in bedrock, limited by the absence of significant vertical fractures would be through overland flow based on topography to the nearest surface water body. The sampling of groundwater could then be through the use of drive points near the surface water body, and/or the collection of surface water samples.

A good site investigation will yield a data set composed of representative physical and chemical information on soil, soil vapour and groundwater conditions collected from a number of locations and depths at various times. It is important that applicable groundwater standards are compared to accurate and reliable groundwater quality data that have been acquired in a manner that represents, to the greatest extent practical, the quality of the groundwater at a site.

Field investigations focused on groundwater quality will have several components, depending on investigation objectives and data needs. Virtually all will include the acquisition and chemical analysis of representative groundwater samples, which are most frequently obtained from conventional monitoring wells installed in drilled boreholes at a site. However, groundwater quality data may also be acquired using other methods, depending on site objectives.

Groundwater investigation is not intended solely to provide data with which to compare against Schedule 3 of the CSR. Other objectives include, for example, the need to determine applicable groundwater standards (refer to Protocol 6, Application of Water Quality Standards). Additional hydrogeologic information is often necessary to assess groundwater velocities, and to allow an estimate of future concentrations of a contaminant in groundwater at an off-site location such as a water supply well, or near a stream where groundwater may discharge. Depending on the level of certainty required, such an exercise will require, at a minimum, that groundwater flow velocities are understood within each of the flow zones where the contamination currently resides. Further effort may be necessary to infer or predict groundwater flow velocities in regions hydraulically downgradient of the site that are located between the contaminant and the nearest receptor. These regions may serve as pathways for contaminant migration in the future.

4.2 Conventional monitoring wells

For site assessments, monitoring wells are commonly installed to acquire such samples. Conventional monitoring wells are commonly composed of a riser pipe inserted into a drilled borehole, and a screened completion interval at the base which is placed within a targeted geologic unit. The well screen is commonly enveloped in a sand filter pack, and is isolated from the overlying borehole and geologic units by an annular seal (commonly bentonite).

Of note, the use of drill cuttings as backfill is to be avoided unless they are known to not be contaminated. Recommended sealants along the well annulus include non-shrinking bentonite-based grouts or solids. In cases where granular bentonite, bentonite chips, or bentonite pellets are utilized, proper hydration of the bentonite during placement should be ensured, particularly if the seal is located in the unsaturated zone. ***Any deviation from these requirements should be identified, together with supporting rationale.***

4.3 Well construction

Well drilling methods commonly used in Yukon include air rotary, sonic drilling, cable tool, hollow stem auger, and Becker hammer. The method selection is usually dictated by the anticipated ground conditions and the availability of equipment. Whenever feasible, drilling procedures should be utilized that do not require the injection of water or drilling fluids into the borehole, and that optimize cuttings control at ground surface. Where the use of drilling fluids is unavoidable, the selected fluid should have as little impact as possible on the water samples for the constituents of interest (ASTM D5092-90). Preliminary laboratory testing of the fluid may be useful in determining potential for contamination.

Monitoring wells must include a protective casing that preserves the integrity of the borehole and if required, be monitored to meet design specifications. This casing must be fitted to a screen that is packed with a filter to enable the collection of sediment-free groundwater samples. Well screen slot size should be based on hydrologic characteristics and on the grain-size distribution of the aquifer being monitored. The primary filter pack material should be a chemically inert material, well rounded, and uniform in size. The most common filter packs are made of sand or gravel. Filter pack

material with a thickness of at least two inches should be installed in the annular space between the borehole and well casing, and sealed with at least 0.5 metres above the sampling depth to prevent contamination of samples. The seals and grout are generally constructed of bentonite and/or cement, as appropriate. Under permafrost conditions, special low temperature grouts and cements may be required. Monitoring wells can range in diameter from 25mm (1 inch) to 150mm (6 inch), with a 50mm (2 inch) diameter the most common. The diameter of a monitoring well should be the minimum practical size which will allow for proper development of the well screen and operation of the sampling device. Large diameter wells (greater than 50 mm) are not recommended as they hold large volumes of water which require more purging prior to sampling. Refer to the USEPA Document SESDGUID-101-R0, Design and Installation of Monitoring Wells, 2008 for more information on drilling techniques and monitoring well construction.

4.3.1 Double cased wells

Double cased wells should be constructed when there is reason to believe that interconnection of two aquifers by well construction may cause cross-contamination, or when flowing sands make it impracticable to install a monitoring well using conventional methods. A highly contaminated surface soil zone may also be cased off so that drilling can continue below the casing with reduced danger of cross contamination. A large diameter pilot borehole should be bored through the overburden and/or the contaminated zone into a confining layer or bedrock. An outer casing (sometimes called a surface or pilot casing) should then be placed into the borehole and sealed with grout from the bottom to the ground surface. A tremie pipe should be used to place grout along the length of the borehole between the outer casing and the borehole wall and at the bottom of the borehole. The borehole and outer casing should extend into a low-permeability material such as clay to a minimum of two feet, or into competent bedrock to a minimum of 1 foot. The total depths into the clay or bedrock will vary, depending on the plasticity of the clay and the extent of weathering and\or fracturing of the bedrock. The size of the outer casing should be of sufficient inside diameter to contain the inner casing, and the 50 mm (2-inch) minimum annular space. In addition, the borehole should be of sufficient size to contain the outer casing and the 50 mm (2-inch) minimum outer annular space, if applicable. A minimum of 24 hours should be allowed for the grout plug (seal) to cure before attempting to drill through it. Depending on the

nature of the contamination, prior to drilling through the plug the inside of the casing may be flushed with clean water to remove traces of contamination. Additional information can be found in US EPA Guidance document, 'Design and Installation of Monitoring Wells', 2008.

4.3.2 Well screen length

Efforts to limit the length of the well screen to the affected hydrostratigraphic unit are preferred to prevent the introduction of a pathway to other stratigraphic units. Based on site-specific information obtained during the site assessment, **monitored depth intervals in each aquifer may range from a few centimetres to a few metres**, recognizing that dilution of constituents is likely to occur for the longer well screens (refer to Figure 3).

$$C_{SAMPLE} = \frac{\sum_{i=1,n} c_i q_i}{\sum_{i=1,n} q_i}$$
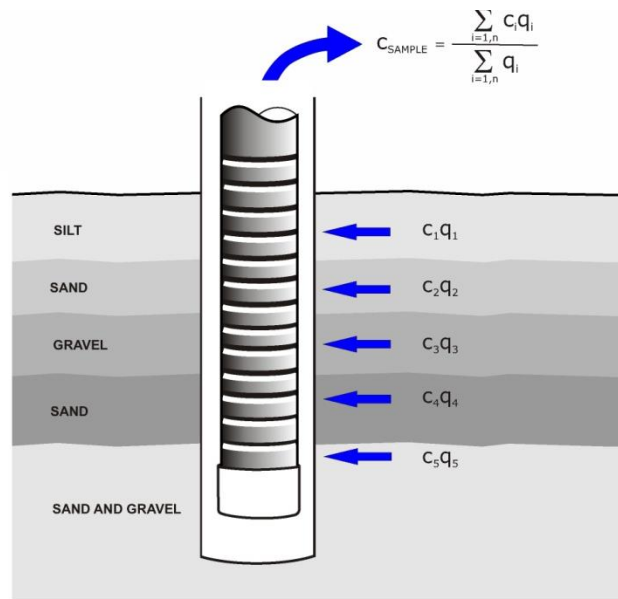
Figure 3: Monitoring well completed within a multi-layered system with varying concentrations
Notes: In heterogeneous aquifers such as the multi-layered system shown, each soil type may carry a different concentration, C_i, of the same dissolved chemical species. When the well is pumped, a sample of the pumped water will represent a quasi-average, C_{SAMPLE}, of the concentrations in each layer, and will be diluted.

Preliminary investigative approaches, such as the use of wells with long screen intervals, can be acceptable during the site assessment provided that cross communication between potentially

separate groundwater zones is avoided. Such approaches are sometimes preferred to establish the presence of potential contamination in groundwater that may be associated with an APEC. However, *where saturated well screen intervals exceed 1.8 m some dilution is to be expected, and chemical data for samples from such wells should not be compared directly with groundwater quality standards unless supporting rationale can be provided.* Wells that are no longer being used, or that are inadvertently screened across more than one aquifer should be decommissioned promptly (i.e., at or prior to the completion of the site investigation), regardless of screen length, to avoid risk of future cross contamination.)

Preference should be given to much smaller intervals, on the order of **0.3 m or less**, so that any expected averaging effect at a receptor (e.g., a water supply well) can be established. In aquifers where the contamination may exceed one to two metres in thickness, multiple wells completed in well nests, or vertical groundwater profiles, should be completed to define conditions over the depth of the aquifer. Multiple, conventionally constructed wells should not be installed within a single borehole due to the potential for contaminant migration between screened sections. Multi-level wells could include standard well construction in separate boreholes, a specially constructed well (i.e. Continuous Multichannel tubing, Waterloo multilevel monitoring system or Westbay multilevel well system), or the use of bundle wells (tubing installed at discrete sampling depths with seals between in a single borehole. *Where a water table aquifer is monitored, the screen length should not extend beyond a depth of one metre below greatest depth to the water table as defined by the seasonal minimum.*

Longer well screen intervals may be used in circumstances where reconnaissance sampling remains appropriate, provided that the risk of cross communication is addressed and minimized. However, in absence of supporting rationale, the chemistry results should **not** be considered directly comparable to applicable standards because of dilution effects.

As each site is unique, variations from these default values are to be expected. However, *any deviation from the default values should be identified, together with supporting rationale and consequent implications on the uncertainty of the acquired data set.*

Well screens longer than 1.8 metres may be justified. Examples are provided below; however, in such cases, rationale should be provided and additional wells with smaller screen lengths must be

installed in nest formations at strategic locations to facilitate contaminant sampling:

- When natural water level fluctuations dictate a longer screen length.

- When the interval monitored is slightly greater (thicker) than the appropriate screen length.

- When a homogeneous, extremely thick aquifer (i.e., greater than 90m) is being monitored, a longer screen (i.e., up to 6 m), representing a comparatively discrete interval, may be necessary.

- Where soils with very low hydraulic conductivity (i.e., less than 10^{-8} m/s) are encountered.

- When monitoring a significant thickness (i.e. > 0.5 m) of a light non-aqueous phase liquid (NAPL) on top of groundwater.

4.3.3 Monitoring wells in frozen ground

Groundwater may occur as supra-permafrost water (water above permafrost within thawed ground during summer months, and which is frozen during the winter), sub-permafrost water (water within the thawed zone beneath permanently frozen ground), or intra-permafrost water (water within thawed zones between masses of frozen ground).

Key principals to consider when installing monitoring wells in permafrost areas or frost susceptible soils include:

1) Minimize effects on the subsurface thermal regime.

2) Maintain wells to ensure sample collection at any time of year.

3) Seal the annular space between the casing and any permafrost to prevent upward seepage.

Considerations for minimizing effects on the subsurface thermal regime

Monitoring wells and well points should be designed and constructed to minimize effects on the subsurface thermal regime (permafrost) and to withstand freeze-thaw forces (seasonal frost). Caution should be used when installing a well through permafrost that may be acting as a

confining unit because flowing artesian conditions may occur. The thawed annulus between the pipe and the permafrost should be firmly sealed to prevent seepage upward from the confined aquifer. Additional information can be found in the Alaska Department of Environmental Conservation, Monitoring Well Guidance document, dated September 2013. Avoid drilling with heated drilling fluid or using warm sealants (grout or hydration water) where the thawing of permafrost around the borehole has the potential to create a conduit for artesian flow (unconsolidated materials). Specialized, higher density grouts used for artesian conditions and grouts that are resistant to cracking due to freeze thaw cycles may be necessary. Where appropriate, packers can also be used to control artesian flow from subpermafrost aquifers.

Frost-susceptible soils

When installing monitoring wells in frost susceptible soils (silts and clays with high moisture content), frost jacking can cause significant upwards movement of the well. This can be minimized by installing a larger diameter pipe (steel or PVC) around the upper portion of the monitoring well to a depth below the frost line (i.e., 2 m) to protect the inner well pipe. More frequent surveying of well elevations may be required to provide accurate groundwater elevations where well casing elevations are changing.

Supra-permafrost wells

Monitoring wells installed above permafrost (i.e. screened in the seasonally active layer) should be designed and constructed so that a representative groundwater sample can be obtained during the period of thaw (i.e., during summer months). In spring months, more ice may be present within the well than in the surrounding soils as the open pipe may provide a conduit for heat transfer, resulting in lower temperatures. To remove the buildup of ice within a well, a saline solution or steam can be introduced to the well. The volume of saline solution or steam added to the well should be monitored or estimated and then purged, in addition to the normal purge volume, prior to sampling.

Sub-permafrost wells

In some locations there may be a seasonal or localized hydraulic connection or window between the supra-permafrost and the sub-permafrost zones, such as in zones of discontinuous

permafrost. In these situations, monitoring wells may need to be installed within the sub-permafrost. Unique permafrost conditions create requirements for specifically designed and constructed monitoring wells. As sub-permafrost groundwater rises in the well to equal the local potentiometric surface, it will pass through the permafrost zone (which is typically at a temperature of -0.5°C to -1°C). If water remains in the well without movement, it will freeze in the well casing thereby preventing sampling (McCullom and Cronin, 1992).

In order to construct a well that can be sampled when needed, it is necessary to employ one of the following methods:

- Install a heat trace cable during well installation that allows the well to be heated and thawed prior to sampling. This requires a source of power to run the heat trace cable and additional time to thaw the well or sampling lines prior to purging and sampling the well.

- Install a system that keeps water from entering the section of the well within the permafrost zone. For example, by maintaining an air pressure in the well that is higher than the formation pressure, sealing the sub-permafrost zone from the permafrost zone using packers, and/or by using a fluid with a depressed freezing point such as glycol. Specialty sampling systems are generally required and often necessitate special training for operation. Such systems include the Westbay well system, gas-push U-tube sampling systems (e.g. Friefeld at al. 2009).

- Use a specially designed well cap that allows the well to be pressurized to a specified level. By adjusting the pressure in the well, groundwater can be depressed in the well casing until it reaches the top of the screened interval below the permafrost (McCullom and Cronin, 1992).

Freezing of fluid within the permafrost zone can also occur during drilling and well installation. A drilling company knowledgeable in drilling in frozen ground should be employed. Drilling can be done under dry conditions using air circulation (e.g. Reverse Circulation drilling), using a heated drilling fluid or using a brine with a depressed freezing point.

Additional information on well installation and groundwater sampling in permafrost regions can be found in the CCME document "Guidance document for Sampling of Contaminated Sites in Permafrost Regions".

4.3.4 Monitoring wells in artesian conditions

If monitoring wells are to be installed in an area that has a history of flowing wells, artesian conditions may be expected. In Yukon, topographically driven flow in mountainous terrain can often create artesian conditions in valley aquifers. For example, artesian conditions are observed in the Haines Junction area where the village distribution system is supplied by artesian wells. Drilling contractors should be engaged that have capability and expertise prevent a well from flowing out of control, and to stop the flow of water if it occurs. Precautions should be taken during well design and installation to accommodate, and be compatible with, the appropriate device (e.g., a control valve) to control the discharge of water. The device should be capable of stopping the flow of water from the well casing and withstanding the freezing of water in the casing. Once a control device is installed, the driller should be able to stop the flow of water by adjusting the control valve to ensure that no water escapes through the well annulus. If water does escape, the annulus should be effectively sealed by grouting. (BC MOE, 2009). Specialized grouts with higher density may be necessary to maintain or gain control. Further information on installation of monitoring wells in Artesian Conditions can be found in the BC Government guidance document "Flowing Artesian Wells" (BC Government, 2009) or the Michigan Department of Environmental Quality "Flowing Well Handbook: (Michigan DEQ, 2005).

4.4 Locating monitoring wells and data interpretation

Properly installed monitoring wells are a common and accepted tool for characterizing groundwater during the site assessment. The resulting well network (at least three monitoring wells, and usually many more) should be used to monitor the progress or confirm the absence of particular contaminants associated with each contamination source, or to establish temporal trends in the behaviour of each plume. The design of the monitoring wells and the monitoring well network requires careful consideration of the local hydrostratigraphic conditions, and the potential receptor(s).

During the site assessment, monitoring well locations should be selected to intercept the highest concentrations of potential contaminants in groundwater associated with each suspected source zone within each on-site APEC, and at the property boundary as close as practical to off-site

APECs. Care should be taken to avoid drilling within and through suspect DNAPL source zones, however, as drilling may result in vertical downward migration of the DNAPL. If underground utility corridors exist, a review of the utility location maps and depth information is required to assess the presence of preferential pathways near APECs and aid in determination of well locations.

Because groundwater flow direction is unlikely to be established with precision during the site assessment, and the presence and extent of each source zone is probably not known, most approaches (e.g., installation of relatively few monitoring wells, wells with long screen intervals, geophysical surveys, soil vapour surveys) should be considered as preliminary investigations. The data from preliminary investigations should almost always be viewed with caution, and usually should not be regarded as conclusive with respect to the absence of contamination. For example, subtle differences between actual and assumed groundwater flow direction (refer to Figure 4), or the dilution effects of relatively long well screens, may lead to erroneous conclusions regarding the detection of low levels of contamination.
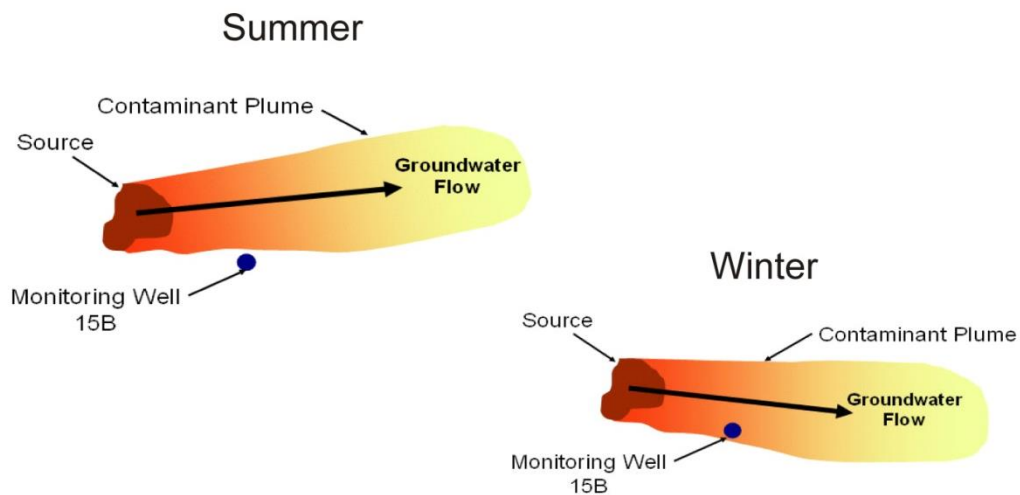

Figure 4: Seasonal changes in groundwater flow direction
Notes: May significantly affect groundwater chemistry at monitoring wells (adapted from Patrick and Thomas, 2007).

To circumvent these shortcomings, groundwater flow direction should be established as part of the site assessment, and the data used to re-assess the optimum sampling locations with respect

to anticipated highest concentrations. ***Groundwater flow direction should be estimated using water-level measurements acquired from a minimum of three locations arranged in a triangular plane within the same hydrogeologic unit (i.e., the same aquifer)***. Caution is advised, however, where groundwater flow patterns are complex (e.g., where groundwater mounding may result in radial flow or in fractured bedrock), as data from more than three wells will be necessary to resolve flow directions.

At the completion of the site assessment, information should be presented and conclusions drawn regarding the presence or absence of groundwater contamination associated with each APEC. In addition, confirmation of the groundwater flow hypotheses that were used to base the inclusion and exclusion of potential APECs during the site investigation should be conducted. Where the groundwater flow information determined during the site assessment differs from that inferred in the site investigation, the site investigation conclusions with respect to APEC selection should be re-examined. The CSM should also be updated. ***Further assessment in the form of an expanded site assessment should be conducted where:***

- Groundwater is observed to be contaminated, as defined in the CSR.

- A well or set of wells is determined to have missed the highest concentrations of a possible plume because it is not located directly downgradient of the potential source zone.

Further assessment should also be considered where groundwater quality data are limited and indicate that PCOCs are present in the groundwater.

Several factors may result in deviations from expected groundwater flow directions. These include the possible influence of local recharge and groundwater mounding; vertical hydraulic gradients resulting from local recharge or discharge conditions resulting in, for example, diving or sinking plumes; and, the influence of unforeseen high permeability features, such as buried gravel channels or utilities, that may act as local groundwater interceptors or drains. It is critical that the possibility for such deviations in groundwater flow are examined within the context of the CSM, so that monitoring locations are established at optimum locations to define the groundwater flow

field and contaminant transport pathways. Further discussion on the assessment of diving or sinking plumes is provided by API (April 2006) and Griesemer (May 2001).

4.5 Acquiring hydrogeological information during the site assessment

In addition to defining the presence, distribution and fate of the contaminants, groundwater investigation during the site assessment should also define site-specific hydrogeologic conditions including:

- The presence, extent and properties of aquifers, aquitards and permafrost zones underlying the site that are or may be of relevance;

- Zones of high hydraulic conductivity that may serve as preferential transport pathways, and zones of low conductivity that may impede or re-direct transport;

- Unconfined and confined aquifers;

- Vertical and lateral hydraulic gradients, groundwater flow direction and velocities within and between the relevant, permeable geologic units;

- Physical and hydrogeologic boundaries that define the groundwater flow systems of interest, including recharge and discharge areas, pumping wells, hydraulic or physical no-flow boundaries or divides, and other relevant conditions; and

- Event and seasonal contributions to the hydrogeologic regime , with a focus on identifying conditions required for sampling to be conducted such that it is characterizing as close to the "worst case" scenario as possible.

These should be resolved at a scale that is compatible with the size of the contamination sources and associated plumes, and the rate of plume migration and evolution. Stratigraphic conditions should be well-defined over the area where the contamination sources and plumes currently exist and over the region that they are predicted to occupy in the future. Stratigraphic conditions should also be understood in detail within the vertical zone or thickness of soil or rock that is occupied by the source zones, with particular emphasis on defining or estimating permeability and permeability contrasts among the various strata and the potential for preferential pathways for contaminant transport.

Hydrogeologic information should be acquired through drilling, well installation, and well monitoring and testing programs, or through alternative approaches that yield comparable site-specific data. Soil and/or rock core samples are usually obtained and used to describe physical aquifer conditions, and hydraulic tests or measurements are made to acquire hydraulic information about the aquifer. Field tests may range from simple static water-level measurements that can be used to assess the water table or piezometric surface of the aquifer, to more involved aquifer pumping tests that hydraulically stress a region of the aquifer, and thereby allow estimation of local and/or regional-scale hydraulic parameters (e.g., hydraulic conductivity, transmissivity, specific yield, storativity). Further information on this topic can be found in various reference texts (e.g., Fetter, 2001; Domenico and Schwartz, 1998; Freeze and Cherry, 1979).

4.5.1 Groundwater flow direction

4.5.1.1 Elevation survey

The groundwater investigation should clearly identify groundwater flow direction and velocity in each of the flow zones of interest. *At a minimum, a) all wells should be surveyed with reference to an elevation datum (a geodetic datum is preferred, although a site-specific reference datum is acceptable and may be necessary at remote locations), b) static water levels should be measured on the same day from monitoring wells at several locations within the same aquifer, and c) the elevation data should be calculated and tabulated.*

4.5.1.2 Elevation data presentation

Groundwater elevation data should be posted on drawings and, where sufficient data are available, contoured in plan for each aquifer of interest, and potentiometric surfaces should be shown for each aquifer on each stratigraphic cross section. The flow direction in each aquifer should then be estimated respecting the data and data contours, and shown on the drawings. Where the data allows, contouring should be conducted within the context of the CSM, with

particular attention paid to apparent "outliers" that may or may not be used in contouring. Potential or probable causes for the outliers should be described. Some common causes for outliers include, for example, data acquired from wells completed with long well screens and/or at different depths within the aquifer, where vertical hydraulic gradients are present within the aquifer, or where wells are installed across more than one aquifer or groundwater flow zone. These types of well completions are not encouraged as they may yield non-representative water-level data and also may allow flow between zones and serve as conduits allowing contaminant migration between the zones.

Using the groundwater elevation contours, a hydraulic gradient (i) can be calculated. Hydraulic gradient is the change in hydraulic head (**dh**) over a certain distance (**dl**).

$$i = dh/dl$$

The hydraulic gradient can be determined by calculating the distance between two groundwater elevation contours.

4.5.1.3 Accounting for LNAPL

The presence of LNAPL in a well may also yield erroneous measurements of water elevation. ***Where significant LNAPL is present (i.e., greater than a few centimetres), the elevation of the NAPL must be corrected*** to determine the actual groundwater elevation, to account for the density difference between the LNAPL and groundwater. As discussed by SABCS (2006), the water elevation can be calculated using the relative density of the oil to water (ρ_{ro}), the elevation of the water-oil interface (Z_{ow}, m), and the LNAPL thickness measured in the well (H_o, m). The theoretical water elevation (Z_{aw}, m) in a well containing LNAPL can be estimated as follows:

$$Z_{aw} = Z_{ow} + (\rho_{ro}H_o) \qquad [1]$$

It also should be recognized that the thickness of NAPL measured in a monitoring well is commonly greater than the actual NAPL-saturated thickness of the formation. Further discussion of this topic is provided by API (2007, 2003).

4.5.1.4 Influence of short-term changes in water levels

Estimates of groundwater flow direction may also be influenced by short-term changes in water elevation or hydrostatic pressures in confined or unconfined aquifers caused, for example, by changes in river stage during spring freshet. In such cases, water levels in an aquifer should not be considered static, and one or two simple "snapshot" measurements of water levels in wells from a site are unlikely to yield reliable data for estimating average groundwater flow direction or velocity.

4.5.1.5 Groundwater velocity

Groundwater velocity estimates may be derived using the following simple analytical model which is a modification of the Darcy equation:

$$v = K\,i\,/\,n \qquad\qquad [2]$$

where:

v is the estimated advective groundwater velocity

K is the formation hydraulic conductivity

i is the hydraulic gradient, and

n is the effective porosity of the aquifer.

This model assumes a homogeneous, isotropic porous medium and laminar flow. Of these variables, n is typically assigned an appropriate value based on the grain size or lithology of a

geologic unit and is rarely measured. The hydraulic gradient, *i*, should be based on contoured elevation data as described above. The hydraulic conductivity, **K**, may be estimated by a variety of means, depending on the level of certainty required. Common methods include:

- Use of simple **"textbook" values**, based on descriptions of soil type, with no actual field tests conducted (simplest approach with highest uncertainty).

- Use of **empirical relationships** drawn between soil grain size and hydraulic conductivity (e.g., Hazen method, as described by Freeze and Cherry, 1979, and Fetter, 2001) (unreliable for soil with more than a few percent of fine materials).

- **Laboratory permeameter tests**, conducted on small samples (typically a few centimetres in length) of formation material (many tests may be required to estimate large-scale hydraulic conductivity).

- **Single-well response tests**, also referred to as slug tests, which are field tests performed at individual monitoring wells, and provide an indication of local horizontal hydraulic conductivity at the well screen.

- **Pumping tests**, conducted on individual wells, with water-level drawdowns monitored at other wells (this approach stresses a much larger volume of aquifer than single-well response tests, and commonly provides more useful and reliable information). However, pumping tests conducted at contaminated sites should be designed to avoid smearing the contaminant plume or producing large volumes of contaminated water that must be properly disposed of.

- **Tracer tests**, whereby the travel time of an introduced groundwater tracer or a contaminant serving as a tracer (usually a conservative inorganic anion such as chloride), is monitored over time and used to directly estimate velocity (usually the most accurate method to estimate velocity).

Each of the variables used to estimate groundwater velocity should be defined so that the uncertainty in the velocity estimate can be provided and is sufficiently narrow for decision-making purposes. An acceptable approximation for expressing the uncertainty is to assign a range to each variable (i.e., **K, i, n**) based on reasonable assumptions and/or test data, and then carry the uncertainty for each variable through the calculations to express a range for the calculated velocity

estimate. For many site assessments, tightly bounded estimates of groundwater velocity are not necessary, and the investigator may use relatively low-cost approaches to derive the estimates.

Of the variables, **K** almost always carries the highest uncertainty. The more sophisticated (and usually more costly) methods for estimating **K** usually provide a higher level of certainty than the simple approaches. Where aquifer response tests (i.e., slug tests) are used to characterize an aquifer, estimates may be improved by acquiring data from several monitoring well locations. However, such methods commonly underestimate actual formation hydraulic conductivity due to borehole smearing during drilling. Therefore, the following guidance is provided:

- Where five or fewer test results are available for five separate locations, the highest value should be used for the velocity estimate.

- Where greater that five results are available, then the 90th percentile should be calculated and used for the velocity estimate.

Where multiple aquifer pumping tests of the same type are used to characterize an aquifer (e.g., all constant head or all step-drawdown tests), then either the highest value or the geometric mean of all values should be calculated and used for the velocity estimate. Where different types of tests are conducted, then the most representative and reliable data estimate should be used.

Where the receptor is located off site at some distance hydraulically downgradient of the site, then groundwater velocity estimates should be based on estimates of the hydraulic properties of the aquifer (i.e., **K** and **n**) between the site and receptor, and on the estimated the hydraulic gradient. *It is considered acceptable practice to extrapolate on-site data to off-site evaluations where it can be reasonably demonstrated that it is the same aquifer. However, where the hydraulic properties cannot be estimated with sufficient certainty for decision-making purposes, then off-site investigation should be conducted.*

4.6 Groundwater sampling

4.6.1 Defining the analytical program

The site assessment should include the chemical analysis of representative samples for an appropriate range of chemicals and parameters in the analytical program, and at the appropriate detection limits. The chemical suite should, at a minimum, include the potential and known contaminants of concern and their potential transformation products. In addition, there is often value in assessing a range of inorganic constituents such as the major cations (e.g., calcium, magnesium, sodium, potassium), major anions (e.g., chloride, sulphate, bicarbonate) and dissolved gases (e.g., nitrogen, methane, oxygen) that may assist in addressing the hydrogeologic characterization (e.g., groundwater age, mixing zones, recharge and discharge areas, etc.). Finally, geochemical information including, for example, redox potential, pH, and nutrient concentrations may provide important information to address the transport and fate of the contaminants as they migrate and attenuate through the subsurface.

A detailed description of field and laboratory sample acquisition methods including a description of monitoring well development, purging, sampling, preservation and QA/QC procedures are provided in Appendix A.

Chemical Characterization

- **contaminants** of known or potential concern, modifying factors (pH or hardness) that are used to establish the applicable standard for certain contaminants, and the potential **transformation products** in the subsurface that may pose risk to potential receptors.

- *field parameters (pH, electrical conductivity, water temperature, dissolved oxygen and oxidation-reduction potential). Field parameters are used to determine the representative quality of the sample following purging, compare to laboratory results as a quality assurance measure, and provide in-situ measurements of parameters, such as redox potential, that may be altered ex-situ.*

In addition, the chemical suite **may also include** the following:

- **inorganic constituents** (more commonly major ions, and less commonly dissolved gases and/or isotopes) that may assist in addressing the hydrogeologic characterization.

- **geochemical and chemical information** that will assist in assessing contaminant transport and fate in the subsurface (e.g., redox conditions, soil and dissolved organic carbon content, dissolved oxygen and pH, nutrients, hardness, etc.) during migration through the aquifer to the receptor.

4.7 Complementary tools

A variety of preliminary approaches and multi-depth tools are available that can be used during the PSI to complement groundwater data and identify the presence or absence of groundwater contamination. These range from simple descriptive observations of continuous cores of soil or rock, to direct-push profiling tools such as laser-induced fluorescence (LIF) tools, the membrane interface probe (MIP) or drive point systems (e.g. the Waterloo Profiler™) A summary of several tools and techniques as described in Health Canada (2008) is provided in Appendix B. These approaches may be used to complement data obtained using monitoring wells and, in some cases, may supplant the need to use monitoring wells during the PSI.

5.0 Groundwater assessment – delineation

5.1 Groundwater objectives and sampling approach

If contamination is confirmed or suspected, the site assessment may be followed with subsequent investigations to define the extent of contamination associated with each area of concern (AEC). During each stage of investigation, the data acquired should be interpreted and used to refine and update the CSM until the objectives of the investigation are met.

As prescribed in section 9 g) ii) of the CSR, a Site Assessment must, among other items, interpret and evaluate the data "in a manner which clearly shows the classification of contamination in soil, groundwater, sediments or surface water relative to Schedules 1, 2 and 3 of this regulation, and shows specific areas, depths, and degree of contamination including migration which may have occurred to adjoining properties." **For purposes of this guidance, and except where the site assessment report has concluded that groundwater quality at a site is acceptable for direct use, groundwater assessment must be undertaken during subsequent investigations.** The sampling program must be sufficiently detailed to satisfy data requirements for a risk assessment, if applicable, and for developing a remediation plan.

5.2 Defining plume size

For groundwater investigation during the site assessment, minimum sizes have been specified in this guidance for plumes that should be identified and characterized with reasonable certainty. These minimums should be used to design the field program, and may serve as a guide to assess the adequacy of the groundwater investigation. As each site is unique, variations from these default values are to be expected. However, **any deviation from the requirements presented below should be identified, together with supporting rationale and consequent implications on the uncertainty of the acquired data set.** Such uncertainty may be described in terms of the level of confidence achieved by the investigation, or by the approximate size of source zones or associated plumes that could have been missed by the investigation.

Vertically, where a groundwater plume is confirmed or suspected, it should be resolved to a scale that is compatible with the scale of the stratigraphic layering that is likely present. ***In absence of site-specific rationale***, data to define and bound the vertical extent and thickness of a plume should be derived from locations that are **separated vertically by no more than one metre** from the bottom of one well or sampling point and the top of the next, within each aquifer of interest (Figure 5). Where monitoring wells are used, care must be taken to select a small monitored interval in order to avoid cross-communication between aquifers, or even between significant stratigraphic layers within the same aquifer.

In aquifers where groundwater contamination may exceed one to two metres in thickness, groundwater quality profiles should be defined using clusters or nests of wells completed at different depths, or by using other groundwater profiling technologies. Alternative approaches to conventional monitoring wells include a variety of tools and technologies that can be demonstrated to yield reliable quantitative or semi-quantitative information on groundwater quality data either directly (e.g., direct-push groundwater sampling technologies), or indirectly through acquisition of soil quality profiles *in situ* (e.g., using technologies such as LIF or MIP) or *ex situ* (e.g., analysis of discrete soil samples or extracted fluid samples from soil cores).

Spatial characterization

The site assessment should characterize the three-dimensional spatial scale of chemical concentration variations with reasonable certainty so that:

- all groundwater plumes of significant size (typically **10 m or longer** longitudinally**, 5 m or wider** laterally, and **0.1 m or thicker** vertically) at a site are identified with reasonable certainty such that the horizontal and vertical boundaries are resolved at a scale identified in Figure 5.

- The effects of **well screen length** and **dilution** at a potential receptor are understood and taken into account in the investigation.

An example of acceptable well spacing intervals and well completion depths for a groundwater plume is illustrated in Figure 5.
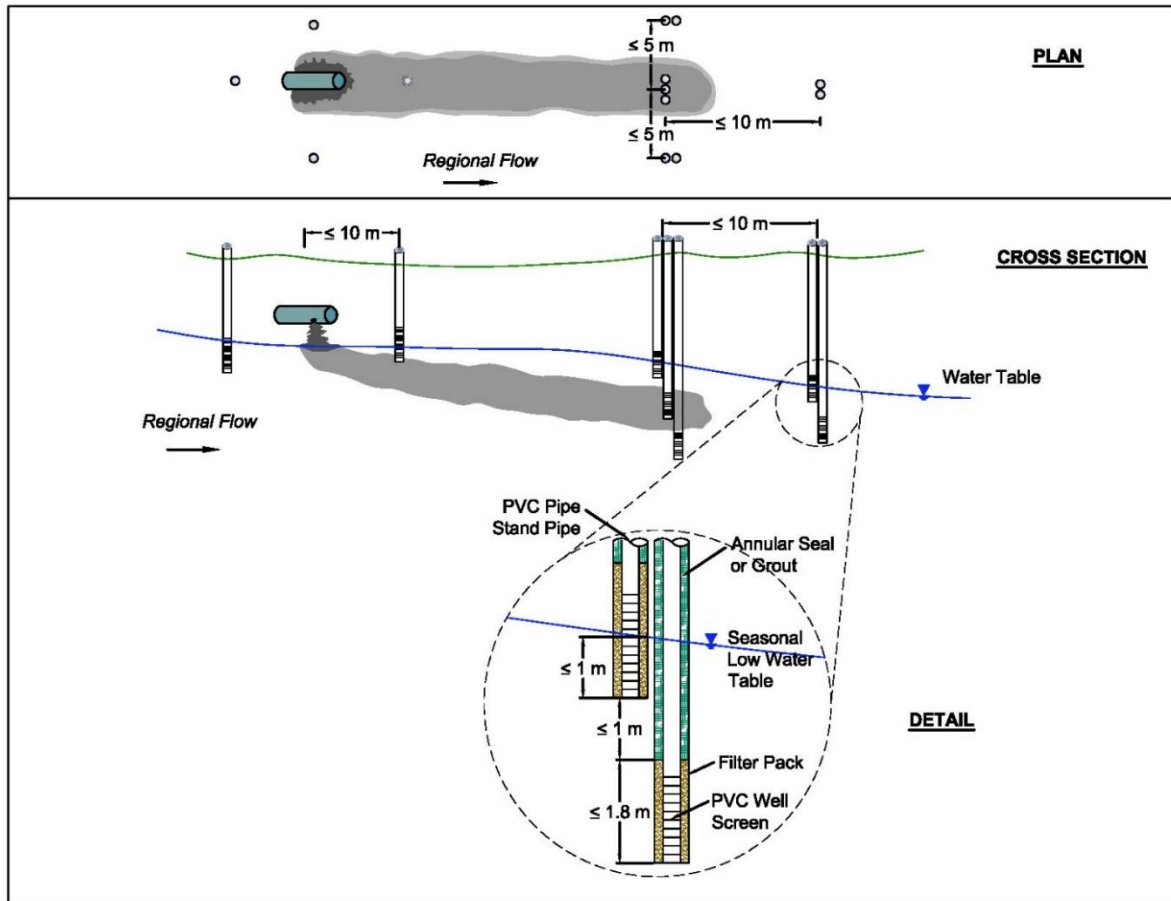
Figure 5: Example Illustrating Acceptable Well Spacing Intervals and Well Completion Depths
Defines the vertical and lateral extent of a plume of groundwater contamination. Note that the number of wells used to define the internal plume size and the spacing between these wells will likely vary from site to site.

Resolution of plume sizes at small scales will reduce uncertainty in the investigation and provide increased confidence in predictions that may be made regarding future plume size, extent and concentration. In addition, or as an alternative, to installing and sampling more monitoring wells, methods to reduce uncertainty may be used such as:

- direct-push technologies (e.g., MIP or LIF)

- geophysics to map certain types of plumes (e.g., highly conductive shallow groundwater plumes such as dissolved salts)

- soil vapour surveys to more accurately map source zones associated with volatile chemicals such as NAPL

5.3 Frequency of sampling

Groundwater chemical concentrations rarely remain constant over time. Factors influencing observed changes in concentration over time include changing water levels caused, for example, by changes in seasonal infiltration rates, and changes resulting from biotransformation or source depletion. Variance is also introduced by the analytical laboratory and by the field sampling procedures. Concentration variations will be significant when the concentration approaches or slightly exceeds applicable numerical standards, and repeat sampling will be necessary in these cases to establish current or to predict future concentrations. The following guidance is provided.

Temporal characterization

The site investigation should obtain a sufficient number of samples over time to:

- establish the magnitude of temporal concentration variations (e.g., seasonal) or allow predictions to be made with reasonable certainty

Where seasonal effects may be significant, or where concentrations are likely to vary significantly for other reasons, then quarterly sampling should be performed over at least one year followed by biannual sampling.

Any deviation from these requirements should be identified, together with supporting rationale and consequent implications on the uncertainty of the acquired data set. Quarterly sampling may be problematic in cold and/or remote locations in Yukon. Wells may need to be designed to function in frozen soils (see Section 4.3) to allow for sampling in winter. Where quarterly sampling is not possible due to winter conditions, at a minimum three seasons should be sampled in the first year to determine the most appropriate time for biannual sampling to be conducted the following year.

5.4 Preferential pathways

The delineation of NAPL and dissolved phase contamination should include an assessment of likely primary release locations (e.g., areas of leaks, spills or releases from storage tanks, sumps, liquid transfer lines) where contamination may have intercepted and migrated along a preferential flow pathway. ***All site investigations should address the possibility of preferential contaminant transport along utility corridors or drainage improvements.*** This will involve a review of utility location maps and depth information, which can then be compared with known information concerning the site stratigraphy, water table elevations, and presence and extent of contamination. Examples of preferential flow pathways which may be a concern include:

- migration along the backfill of a buried utility or buried stream channel, with subsequent vertical migration and lateral spreading;

- migration along from the utility backfill and into a utility (e.g., into a storm or sanitary sewer), with subsequent migration along the utility to a receptor;

- migration and discharge to a storm or sanitary sewer, and subsequent leakage from the buried utility (e.g., leakage out of pipe joints) into the backfill and/or into the surrounding soil; and

- migration into building drainage systems and subsequent discharge to storm sewers and ultimately to a receptor.

Emergency situations such as the presence of flammable or corrosive liquids or vapours in utility corridors and drainage improvements must be addressed upon discovery, usually before an investigation begins, and actions to halt migration of contamination may be undertaken at that time.

Further guidance on the assessment of utility corridors may be found in Wisconsin Department of Natural Resources (October 2013): "Guidance for Documenting the Investigation of Utility Corridors".

5.5 Technologies to acquire groundwater quality information during delineation programs

High-quality samples yielding reliable, precise and accurate chemistry data are required during subsequent site assessments to ensure that appropriate groundwater chemistry data are available for comparison against CSR standards. However, to satisfy investigation requirements, preliminary approaches also may be used to acquire additional data over a broad area, time period and/or vertical thickness of soil or rock. Once the scale of the contamination is understood, then high-quality data may be acquired from a few strategic locations for verification and quantification.

Several technologies are available for groundwater investigation. Many are compared and contrasted in Table B1 of Appendix B, in terms of the quality of the data provided, and the relative resolution of scale that may be achieved by the data. Spatial scales are often best satisfied by technologies that acquire many data points from many locations on a one-time basis (i.e., they provide a "snap-shot" of current conditions). Of these technologies, some are more suited to resolving lateral spatial scales (e.g., mini-piezometers) while others can better resolve vertical scales (e.g., direct push profilers). Satisfying temporal scales is usually best accomplished by multiple samplings over time from permanent or semi-permanent installations (e.g., conventional monitoring wells).

In addition to direct methods for groundwater sampling, there are a number of indirect methods available to infer subsurface conditions and groundwater quality, and thereby supplement a limited water quality data set. These range, for example, from qualitative, detailed descriptions and logging of field observations during drilling and sampling (e.g., odours, NAPL sheens, colour and staining, etc.), to the more complex methods of vertical profiling using special down-hole tools and "direct-push" technologies. Use of such technologies is encouraged as part of detailed investigations.

5.6 Non-Aqueous Phase Liquids (NAPLs)

Where light or dense non-aqueous phase liquids (NAPL) may have been released at a site, they will migrate down towards the water table and, if volumes are sufficient, will tend to either accumulate and spread laterally along the approximate water-air interface in the subsurface, if LNAPL, or sink below the water table, if DNAPL, and accumulate and spread laterally along finer

grained capillary barriers such as silts or clays. Conventional monitoring wells are commonly used to identify the presence and recoverability of LNAPL (e.g., SABCS, 2006; API, 2007), whereas alternative approaches are often necessary to establish the presence of DNAPL (e.g., Pankow and Cherry, 1996).

Where LNAPL is suspected, monitoring wells should be designed so that the well screen interval straddles the water table, thereby allowing LNAPL, if present, to enter the well. If the well may be used for long-term monitoring purposes, then the well screen length should be selected to straddle the water table over the anticipated seasonal high and low water-table conditions. The presence of LNAPL will usually be identifiable at the time of well installation or during well development.

Where NAPL is likely present, the presence and thickness of any immiscible layers should be established prior to purging and sampling through the use of a reliable technique such as an interface probe. In those wells with NAPL, groundwater sampling is not advisable as measured concentrations may often reflect lower-than-actual dissolved concentrations due to sample dilution, or higher-than-actual concentrations due to entrainment of NAPL in the samples.

Where NAPL is present and in contact with groundwater, chemical equilibrium between the NAPL and groundwater is likely, and concentrations of NAPL constituents dissolved in groundwater will be high. To assist with data interpretation, the effective solubility of each PCOC at wells with NAPL should be calculated and representative PCOCs posted on figures and contoured with other groundwater data. *Where NAPL thickness in a well is significant (i.e., greater than 2 mm), then dissolved concentrations equivalent to the effective solubility of the constituents in the NAPL should be estimated and used to compare against applicable groundwater standards.*

The nature of NAPL flow in the subsurface is rarely simple and not easily predicted (e.g., SABCS, 2006; Cohen and Mercer, 1993). Once NAPL is suspected, the boundary which defines the possible presence and likely absence of NAPL sources should be defined and, depending on whether LNAPL or DNAPL is present, delineated with reasonable certainty. Many of the technologies and approaches described above and in Appendix B can be effective for NAPL identification and delineation, and their use is encouraged.

5.6.1 LNAPL investigations

Where LNAPL is present, at least one LNAPL monitoring well should be placed within each zone where LNAPL is inferred to be thickest. The lateral boundary of the LNAPL zone should be resolved at a scale of 5 m to 7 m or less, depending on proximity of NAPL to property boundaries, structures and other sensitive site features. For most LNAPL investigations, it is expected that the spacing of data points to define the LNAPL perimeter will be about 5 m to 7 m. Data to define the boundary may be acquired using various technologies such as monitoring wells, LIF, soil cores or test pits. However, some LNAPL monitoring wells should be installed in downgradient locations along the perimeter of the LNAPL zone to monitor LNAPL thickness and the possibility of LNAPL migration.

LNAPL monitoring wells should be constructed as water table monitoring wells so that the well screen straddles the water table. The screen should be sufficiently long to straddle water table during seasonal changes in water table elevation. Where groundwater samples may be acquired for chemical analysis, the *well screen should not extend more than one metre below low seasonal water level*.

Following well development, LNAPL may not enter the well immediately. Therefore, the well should be allowed to rest at least 24 hours and preferably at least one week before confirming the presence or absence of LNAPL. Where NAPL is present, a regular monitoring program should be established for at least 12 months (or as long as necessary to assess NAPL migration), or until remediation has been undertaken, or until the LNAPL is demonstrated to be immobile. In absence of site-specific rationale, monitoring frequency at each LNAPL well should be **at least once every two months and preferably monthly**, for the following:

- total organic vapour measurements at the well head using a photoionization detector or similar field instrument

- water and product levels using a reliable method such as an interface probe

Groundwater concentrations from wells located downgradient of the LNAPL plume should be analyzed to evaluate potential LNAPL movement. A temporal increase in dissolved concentrations

downgradient of the plume may be indicative of LNAPL advancement; conversely, a temporal decrease in dissolved concentrations may be indicative of LNAPL plume retreat.

During the monitoring period and until a well is decommissioned, its integrity should be checked and maintained, including maintaining effective well plugs and seals to prevent cross-contamination.

5.6.2 DNAPL investigations

Where DNAPL is present, care must be taken when drilling into or through suspect NAPL source zones, as the risk of causing cross contamination between locations can be high. DNAPL distributions in the subsurface are often highly complex, and as a result, direct evidence of DNAPL is rarely obtained using conventional drilling and sampling techniques. Thus, a precautionary approach is advised wherein a region is defined within the site that is likely to fully contain the DNAPL source. Drilling into and through the inferred region of DNAPL should be avoided and, instead, the vertical and lateral extent of the DNAPL zone should be inferred using soil, groundwater and soil vapour data obtained mainly from the surface and perimeter of the zone. In such cases, ***the outer horizontal boundary of the inferred DNAPL zone should be resolved at a scale of 5 m to 7 m, and the vertical extent should be resolved to a depth of 1 m to 2 m,*** depending on proximity of DNAPL to property boundaries, structures and other sensitive site features. For further information on DNAPL assessment, please refer to Pankow and Cherry (1996) and USEPA (2004).

5.7 Data assessment and interpretation

5.7.1 Data presentation and reporting

Groundwater investigation reports should include a description of the methodology used to evaluate site hydrogeology and the rationale for the methods used. Summaries of key information should be provided in tables and on figures as a means to convey relevant information. Much of

the regional and local information in a groundwater investigation study, such as the surface topography, water-table surface, stratigraphic conditions, spatial distribution and inferred extent of contamination, and locations of human and/or aquatic receptors, describe physical conditions and spatial relationships that are most effectively portrayed both with text and pictorially using plans, cross sections and three-dimensional representations (e.g., fence-diagrams). More innovative approaches to convey site information may provide more effective alternatives and include digital slide presentation formats and three-dimensional visualizations. The data should be presented in a manner that communicates an accurate portrayal of the CSM, and clarifies the rationale used to conduct and complete the investigation. The recommended types of raw data that should be presented, and the recommended figures and tables summarizing the groundwater studies are provided below.

Guidance for data presentation

<u>Borehole and well construction logs</u>

Logs must be provided for all geotechnical wells, boreholes, and all wells and piezometers, presenting complete technical records of conditions encountered, scaled to depths of at least 0.1 m. Logs should contain, at a minimum:

- Site name and location

- Name of driller and on-site professional

- Borehole number and location coordinates (lat/long or UTM)

- Start date, completion date, date abandoned or completed as monitoring well

- Borehole depth

- Ground surface elevation, top of casing elevation (for wells)

- Sample type, depth and depth interval for all *in situ* samples

- Sample condition, present recovery, and other field data (e.g., blow counts, moisture content)

- Materials classification (based on field and laboratory descriptions using the Unified Soil Classification System (USCS; ASTM D2487) or equivalent and RQD if applicable.

- Observations including colour, stains, odours

- Drilling observations such as loss of circulation, heaving sands

- Volume and quality of water added during drilling

- Depth to water following drilling

- Water level and date following well completion

Raw data from *in situ* hydraulic tests and copies of laboratory analytical certificates should be provided.

Tables should be provided that include, at a minimum, summaries of all field and laboratory data acquired from current and previous investigations, including:

- Water level depths and elevations along with screened interval elevations
- Analytical chemistry results for each environmental medium compared to relevant environmental quality standards

Figures and/or drawings should include, at a minimum:

- A scaled regional location plan and site plan, showing relevant hydrological, topographical and physiographic features

- A plan of posted data at measurement locations, and contours, where sufficient data are available, of piezometric heads in each aquifer of interest

- Stratigraphic cross sections that are longitudinal and transverse with respect to the known or estimated groundwater flow direction, and that include physical conditions (e.g., stratigraphy, water table, piezometric surface elevations), location and depth of all boreholes, monitoring wells and well screen intervals falling on or near the section, and vertical and horizontal scales

- Posted data at measurement locations, and contours where sufficient data are available, in plan and cross section, of chemical concentrations that show the specific lateral and vertical distribution of either each COC or a representative set of COCs in on-site and off-site soil and groundwater

- Sample locations with corresponding analytical results used to develop each figure, that are shown on the figure and in tabular form with reference to applicable standards

Other

- Field sampling sheets should be provided in an appendix documenting information such as volume of water purged, observations during purging, and relevant field-measured parameters (e.g., electrical conductivity, temperature, pH, dissolved oxygen, redox potential, etc.).

Where data are contoured, the contours represent an interpolation between data points, and are therefore subject to some uncertainty. Areas of obvious uncertainty should be demarcated on posted data presentations and contour plots, so that the uncertainty is effectively communicated.

5.7.2 Modelling

In developing the conceptual site model, analytical or numerical models may be used to better understand the limitations and areas of uncertainty of the current data set, and to predict future conditions. Of the models available, they may be stochastic or deterministic, transient or steady state, and one-, two- or three-dimensional. Groundwater models are generally grouped into either groundwater flow models or fate and transport models. Flow models simulate the direction and rate of groundwater flow through the subsurface, whereas fate and transport models simulate the movement and chemical alteration of contaminants as they migrate through the subsurface. The latter model types, which are usually coupled with a calibrated groundwater flow model, often require, at a minimum, that the groundwater velocity is accurately determined.

A discussion of models and modelling approaches is provided elsewhere (e.g., Bear et al., 1992), and a good discussion on their use in assessing contaminated sites is provided by the Ohio Environmental Protection Agency (November, 2007). As a rule, most problems in groundwater investigation can be readily framed and often resolved through the use of simple analytical models using, for example, formulae based on Darcy's Law. Once the data needs have been identified and the degree of allowable uncertainty has been established, then more complex models may be necessary to better understand the hydrogeologic system or to predict future hydrogeologic

conditions. In this way, field programs may be used to refine what models should be used and the model results may better define the field program by identifying gaps in the data needed to define the hydrogeological system.

5.8 Refinement of the conceptual site model

5.8.1 Conceptual site model development

Data assessment and interpretation should be on-going during the groundwater investigation, with each new piece of information feeding into the CSM. New data should be used to update the CSM, test the model assumptions, and revise the CSM where necessary to account for the new information. As an ultimate goal, a robust CSM should be developed that allows predictions to be made with the confidence necessary for successful and reliable decision making. Understanding current conditions is fundamental to the development of the CSM, and this provides the platform for data extrapolations and predictions of future conditions.

5.8.2 Groundwater flow regime

Updating the CSM with respect to groundwater flow can be facilitated as field data are acquired by preparing simple two-dimensional stratigraphic cross sections and contour plots, or two- or three-dimensional visualizations of the field stratigraphic and hydraulic information. Particular attention should be paid to data outliers, as discussed above, so that the causes are understood. By undertaking such forms of data assessment and interpretation in the field, and routinely updating the CSM, issues can be identified promptly, and timely and effective field decisions can be made. Once the investigation is complete, the CSM should be sufficiently detailed such that hydrostratigraphic conditions and groundwater flow can be clearly presented in a three-dimensional context in the report.

5.8.3 Chemical characterization

As field and laboratory chemistry data are acquired, they should be verified for accuracy and completeness, and then reviewed within the context of the CSM. Uncertainty should be assessed and quantified in accordance with the data quality objectives established as part of a quality assurance/quality control program for the site assessment. As part of the data review and interpretation, care should be taken to note any unusual or unanticipated chemical concentrations

or constituents. Data should be posted on plans and sections and reviewed to identify patterns and concentration gradients that may or may not be consistent with contaminant plumes emanating from source zones. Unusual or unexpected patterns in the data, either spatially or temporally, or expected or unexpected relationships among the constituents, should be identified and assessed.

5.8.4 Contaminant migration

The hydrogeologic and chemical information should be drawn together and integrated in the CSM to describe contaminant distribution and relevant processes that may be acting to attenuate or transform the contamination as it migrates through the subsurface. Relevant processes may include, for example, advection, dispersion, chemical retardation, biodegradation, and volatilization. Where these processes are relied upon to support conclusions reached in the site assessment, then they must be supported by site-specific data. For example, where the site assessment must rely upon transport processes such as retardation to draw conclusions regarding off-site migration and travel times to a receptor, then the process should be supported by site-specific measurements. Retardation typically requires the measurement of organic carbon fractions in the aquifer materials so that site-specific estimates of partitioning coefficients can be made between the soil and contaminants of concern.

As new information or data are acquired, each data piece should be assessed within the context of the CSM and its assumptions. Inconsistencies should be identified and addressed either by revising the assumptions of the model, or by deriving a reasonable and supportable explanation for the inconsistency. Where data do not fully support the assumptions used to draw conclusions, then the conclusions should not be regarded or stated as firm, uncertainty should be explicitly noted, and further investigation may be necessary.

6.0 Monitoring to confirm remediation

6.1 Planning the remediation monitoring program

Where groundwater is to be remediated, remediation planning must consider where and how monitoring will be conducted and how the data will be used to confirm that the remediation objectives have been achieved. Changes in groundwater quality that may result from the remediation should be identified within the context of the CSM, and the CSM should provide the rationale and serve as a guide to identify remediation and confirmatory sampling requirements.

When establishing confirmatory sampling locations, a number of factors should be considered including the following:

- The known extent of contamination.

- The groundwater flow direction prior to remediation.

- The possibility and duration of short-term changes in groundwater levels resulting from the remediation (e.g., temporary low water levels following remediation).

- Where active groundwater controls are required, changes in groundwater flow are long-term and result in new steady state conditions which need to be confirmed.

- Transitioning periods in water levels or groundwater chemistry until long-term or steady-state conditions are re-established.

- Expected transport velocities and travel times between remediated areas and monitoring locations.

- Changes in geochemical conditions such as redox potential that may affect the solubility or mobility of some constituents.

Even when groundwater contamination is not present, groundwater monitoring may be a necessary component of the remediation to verify that the remediation does not adversely affect groundwater quality. For example, where *in situ* treatments are applied, hydraulic or geochemical conditions in the aquifer may change, resulting in changes to groundwater quality. Such changes should be monitored to verify acceptable conditions following treatment.

6.2 Confirmation of remediation

Spatial, chemical and temporal scale should be considered in the design of the monitoring program to confirm remediation. Post-remediation monitoring locations should be selected to include locations that will intercept each of the zones most likely to contain highest concentrations of the contaminants of concern. Note that remediation wells (e.g. extraction or injection wells) should not be used for evaluation of post-remediation groundwater quality. Groundwater should be monitored for COCs as well as transformation products or constituents that may have been mobilized by the remediation (e.g., metals dissolution in response to low redox conditions caused by enhanced *in situ* biodegradation of petroleum hydrocarbons). Where trends are to be established, the frequency of monitoring should be based upon known hydrogeologic conditions including estimated groundwater and contaminant transport travel times.

The scope of the remediation monitoring program will be different for every site. However, the following items should be regarded as minimum requirements to confirm groundwater remediation:

- A monitoring network should be established that includes **a minimum of three monitoring locations** within each affected aquifer associated with each area of groundwater contamination.

- Each groundwater monitoring station (usually a monitoring well, but alternative technologies may be equally effective) from the designated network should be strategically **located within the remediation zone or along its immediate perimeter**, within the relevant permeable geologic units. Installation by placement of post-remediation monitoring wells during excavation backfilling is to be avoided.

- Groundwater levels and groundwater quality indicator parameters (i.e., temperature, electrical conductivity, pH, redox potential, dissolved oxygen, turbidity) should be monitored prior to each sampling event to **verify that static conditions have been attained**. As a minimum subset, pH, electrical conductivity plus one additional parameter should be monitored until they have stabilized (see Appendix A).

- Once static conditions have been attained, **at least two sets of groundwater samples should be collected on different days, at least 24 hours apart, but preferably greater than two weeks apart, where practical.**

- ***Representative samples should be analysed from all designated locations or wells*** for the contaminants of concern and for possible contaminants that may have resulted as a direct or indirect consequence of the remediation.

Post remediation groundwater monitoring may be considered complete when substance concentrations are less than applicable standards, concentrations can be shown to be stable or decreasing, and where rebound can be discounted.

Following completion of remediation, a confirmation of remediation report should be prepared that documents the quality and performance of remediation measures on completion of the remediation, including compliance with remediation standards, criteria or conditions prescribed in the CSR.

6.3 Long-term groundwater monitoring

With many sites where remediation is undertaken, groundwater quality may not improve immediately, or improvements may not immediately meet applicable standards. In such cases, long-term monitoring will be necessary to either establish trends toward meeting the remediation objectives, or to provide sufficient data to demonstrate that the standards are met over time regardless of changes in water levels or groundwater flow direction. Statistical approaches to establish trends in concentration, periodicity, and long term average or mean concentrations include the application of the Students t-test (Zhou, 1996), regression analyses, Spearman's rho test and the Mann-Kendall test (e.g., Yue et al., 2002; Hirsch et a*l.* 1991).

Long term monitoring is also required at solid waste disposal facilities located across Yukon to monitor potential leachate transport from former buried waste. The permit for each facility establishes required sampling locations, frequencies, and parameters.

7.0 Well deactivation and closure

Monitoring wells that no longer serve their intended purpose require appropriate decommissioning. Reasons for decommissioning wells include:

- Wells that may remain at the completion of a site investigation or remedial monitoring program;

- Wells that have been damaged and are unusable;

- Wells that are not serving the purpose they were installed for (e.g. downgradient well that is actually cross gradient); or

- Wells that are interfering with operations at a site and need to be moved, provided this does not interfere with site characterization.

Neglected wells often become damaged and/or buried, and may provide conduits for contamination (e.g., a surface spill at an industrial site) to enter the subsurface. The objectives of successful well deactivation and closure are to prevent surface infiltration of contaminants to an underlying aquifer, and to prevent cross communication between flow zones intercepted by a well screen and monitored interval.

1) For wells where the screen and filter pack intervals do not cross communicate between separate groundwater flow zones then, if possible, the well casing should be pulled, and the resulting borehole backfilled from its base using a tremie pipe to deliver a low permeability grout such as bentonite or a cement-bentonite mixture. If the borehole collapses after casing removal or where long well screens cross communicate between flow zones, then the well should be re-drilled and grouted from its base to surface.

2) As an alternative to well removal, the well may be sealed by injecting grout into the well under pressure, with the intent of injecting grout through the well screen and into the surrounding filter pack. Simple placement of grout into the well casing will not necessarily address the filter pack of the well. In some cases, such as where there is a risk of contaminant transport through the annulus to the screened interval, it may be necessary to perforate the casing to allow grout to penetrate the well annulus. In situations where the well completion interval is one metre or

less, the issue of hydraulic cross communication by the filter pack will be of less concern, and simple sealing of the casing with bentonite to surface may be appropriate.

Where the well is damaged below grade and cannot be accessed, attempts should be made to drill out the well and then grout the borehole to surface. Caution is advised, however, as attempts to over drill piping such as polyvinyl chloride (PVC) can sometimes result in lateral displacement of the pipe into the sidewall. As per the Groundwater Protection Regulation and Schedule 4 therein, information on the abandonment of a permanent monitoring well should be promptly recorded in a well closure report that should be accessible for review for at least five years, if requested.

8.0 Acknowledgments

This document was prepared by Golder Associates Ltd.

9.0 Additional information

For more information on contaminated sites, please contact:

Environmental Programs Branch (V-8)	T: 867-667-5683 or
Department of Environment	1-800-661-0408 ext. 5683
Government of Yukon	F: 867-393-6205
Box 2703, Whitehorse, YT Y1A 2C6	E: envprot@gov.yk.ca

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Appendix A

Field and laboratory data acquisition methods and approaches

Several types of groundwater information are best acquired in the field rather than by a fixed analytical laboratory, provided that the data are acquired by trained personnel using acceptable procedures and protocols. Important field measurements such as groundwater pH, temperature, dissolved oxygen, redox potential, electric conductivity, and alkalinity should be taken in the field as they are subject to significant and often rapid changes once the groundwater has been removed from the subsurface. Such data and procedures should be provided in the site assessment report, as they are often critical to the interpretation of site conditions.

These and other measurements that are easily obtained in the field can provide the investigator with useful information to direct the field program. For example, electrical conductance is simply measured with a probe and provides a rapid estimate of total dissolved solids content, which sometimes serves as an excellent indicator of plume strength such as landfill leachate. Total organic vapour concentrations, measured in the head space of an enclosed jar sample of groundwater using an organic vapour meter, can often provide a rapid estimate of total VOC concentrations in the water sample. In addition to simple probes, several types of direct measurements of soil or groundwater chemistry can also be obtained using direct push technologies.

Well development

Where data are to be acquired using samples from monitoring wells, it is important that the well is developed soon after installation to remove fluids potentially introduced to the well during drilling, and to remove particulates that may have become entrained in the well and filter pack. However, well development should not be performed prior to 24 hours after installation to allow the hydration of sealant (bentonite) and proper setting of bentonite grout. Monitoring well development is intended to correct any clogging or compaction that may interfere with water quality analysis, to improve hydraulic characteristics and to restore groundwater properties disturbed during the drilling process. Monitoring well development should follow the installation process and continue until the water is representative and free of the drilling fluid cuttings, or other materials introduced during the drilling process. Representative water is assumed to have

been obtained when pH, temperature, and specific conductivity readings have stabilized and the water is virtually clear of suspended solids (ASTM D5092-90).

Well development can be achieved in several ways. Some of the more common methods involve a) use of a surge block to flush and move water in and out of the well screen, and then to surface, and b) briefly over-pumping and then resting the well using a submersible pump, and c) air-lifting fluids from the well by injecting air from a compressor through a downhole pipe that discharges the air near the well bottom. Development should be conducted by experienced personnel to avoid compromising the integrity of the well and formation.

Well purging and sampling

Following well development, it is unlikely that the monitoring well will be in equilibrium with conditions in the surrounding geologic medium. For example, the sand filter pack between the well and geologic formation will not be in geochemical equilibrium, gases may have been introduced where drilling methods such as air rotary have been used, and NAPL, if present in the formation, may not have achieved a new static equilibrium with respect to well and pore geometry and hydraulic pressures following drilling. In tight clays, well development may lead to a week or longer delay as the well recharges. To reduce uncertainty in the subsequent monitoring data set, it is common practice to acquire samples at least one week following well development and preferably after two weeks. However, it is recognized that, in some circumstances, near-immediate results are required and that many locations in YT are remote and require long travel times. In such cases, where it is logistically necessary to sample immediately following well development, the results should be considered preliminary pending a subsequent sampling event.

Water which has resided in a well casing for an extended period of time has the opportunity to exchange gases with the atmosphere and to interact with the well casing. Water standing in the

> ### Resting time between well development and sampling
>
> ***Groundwater sampling from newly installed monitoring wells should be conducted at least one week following well installation and development***. Where shorter intervals are desired or required, the data acquired should be considered "preliminary" until a subsequent second sample can be obtained and analysed after one week to confirm or revise the data set. Where further certainty may be desired, particularly where decisions are to be made

columns inside the well casing must, therefore, be purged prior to sampling so that a representative sample can be obtained.

For purposes of this guidance, the following practice is recommended:

At the time of sampling, groundwater is usually first removed from the well and field measurements are monitored over time prior to sample collection in a process referred to as **purging**. Field measurements are monitored until stable (within an acceptable tolerance, listed below) to infer that groundwater conditions representative of the aquifer are present, and that a representative groundwater sample for chemical analysis can be obtained.

In order to verify that the well has stabilized, groundwater levels and groundwater quality indicator parameters (i.e., temperature, electrical conductivity, pH, redox potential, dissolved oxygen, turbidity) should be monitored prior to each sampling event. As a minimum subset, pH, electrical conductivity, plus one additional parameter should be monitored until three successive readings fall within the following limits (ASTM, 2013);

- pH +- 0.1 units

- Electrical conductivity +- 3%

- Dissolved oxygen +- 10%

- Turbidity +- 10%

- Redox +- 10 mV

- Temperature +- 0.2 °C

Note that laboratory measurements of these parameters are not considered to be accurate representations of groundwater conditions due to the unstable nature of these parameters.

The most common practice used to obtain reliable field measurements (e.g., pH, conductivity, temperature, and others) involves placement of field probes into a flow-through cell. As groundwater is pumped from a monitoring well through the cell, direct measurements of each variable are then obtained from calibrated instruments attached to the probes. Stabilization of field parameters is likely to be indicative of a quasi-equilibrium condition and subsequent samples may

be considered representative of the aquifer. Where a flow-through cell is not used, care must be taken to minimize exposure of the water to the atmosphere prior to measurement. Even a few seconds exposure to the atmosphere may significantly alter readings of variables such as dissolved oxygen.

Conventional purging practice is to remove at least three to five "well volumes" prior to sampling, where a well volume comprises the volume of standing water in the well. Some practitioners include the additional water volume entrained in the sand filter pack in the annulus between the wells screen and borehole wall. Either approach is usually acceptable, provided that the practice is consistent among wells and different sampling events. Methods used and volumes purged should be reported as part of the site assessment report.

Purging should be accomplished by removing groundwater from the well at low flow rates using a pump. Because they can operate at variable speeds, pumps such as the submersible and bladder variety are considered particularly useful for purging stagnant water from a well. The use of bailers should generally be avoided as the 'plunger' effect of their use can result in the continual development or overdevelopment of the well. A low purge rate will reduce the possibility of stripping VOCs from the water and reduce the likelihood of mobilizing colloids in the subsurface that are immobile under natural flow conditions. For further reference, refer to the designation guide ASTM D4448-85a. If contaminants are suspected in the groundwater prior to purging, then appropriate disposal measures should be performed. The purged groundwater should be collected and tested and disposed of in accordance with established sanitary/stormwater sewer use criteria and other applicable regulatory requirements.

In many site investigations, project objectives often necessitate installation of monitoring wells in relatively low-permeability formations (e.g., clays and silts, or fractured rock). Purging such wells is sometimes difficult, and frequently results in purging the well dry. In such situations, it is recommended that such wells be carefully and slowly purged, with the objective of **avoiding drawdown of the water level in the well to the well screen** (Puls and Barcelona, 1996). The purge water should be monitored for field parameters. Water levels in the well should be recorded at the beginning and end of the purging process, and then be allowed to recover prior to sampling. Where water-level recovery may take several hours to days, it must be recognized that the sampled water is likely to have established partial or full equilibrium with atmospheric conditions,

and that a truly representative groundwater sample may not be possible. In particular, VOCs may be substantially lower in the sample than the groundwater, and constituents such as metals may be biased low due to precipitation.

Within the context of well averaging that occurs while pumping, as discussed previously, it should be recognized that the stable field measurements (conductivity, temperature, turbidity, and pH) are likely to be indicative of a quasi-equilibrium condition. The groundwater sample obtained following purging will represent a mixture of formation waters that enter the well screen from the various permeable zones encountered at the well screen and/or well filter pack. Uniform purging and sampling techniques serve to stabilize the mixing process, yielding stabilized field measurements.

Once conditions in the well are considered stable, then a variety of acceptable sampling methods are available to acquire the groundwater sample. Some of the more common and innovative methods are discussed briefly below:

- **Conventional sampling approaches** — Some of the more common sampling methods used to recover groundwater samples include the use of bailers, inertial lift pumps (e.g., Waterra™), bladder pumps and downhole submersible electrical pumps. When applied conventionally, the pumps are used to purge the well of at least three to five volumes of water from the well prior to sample collection. Field parameters, as discussed above, are monitored to infer that representative groundwater conditions have been achieved. In low-permeability formations, it may not be possible to remove at least three well volumes of water from the well, and fewer well volumes, or alternative sampling methods, should be considered, recognizing that the sample may not truly represent groundwater conditions. Once purging is complete, samples are then obtained in sample containers and preserved, if required, prior to transport (usually in a chilled container) to the analytical laboratory for analysis.

- **Low-flow purging and sampling**—Low-flow purging and sampling refers to procedures that minimize the flow of water through a well screen during pumping, resulting in less disturbance at the well screen and production of a smaller volume of purge water prior to obtaining a stable, representative groundwater sample. Common techniques involve setting the tubing or intake of a pump (e.g., peristaltic, bladder, centrifugal, variable speed low-flow electrical submersible) at the well screen and withdrawing formation water at rates of about 100 to 500 mL/minute. Withdrawal rates in excess of one litre per minute should be avoided. Water levels are typically monitored during purging, to ensure that minimal formation drawdowns (i.e., about ten centimetres or less is preferred but not mandatory) are achieved. With low-flow sampling, the intake of the sampling device is set at a low velocity to minimize

drawdown in the well, thereby minimizing hydraulic stress and disturbance on the well and adjacent geologic formation. Greater stable drawdowns (i.e., greater than ten centimetres) may yield acceptable samples, although the increased hydraulic stress imposed on the formation at the well screen may yield disturbed (e.g., turbid) samples. In situations where the well is completed in a low-permeability formation, it may be necessary to purge at very low flow rates (i.e., less than 100 mL/minute), taking care to avoid dewatering the well screen (Puls and Barcelona, 1996). If dewatering remains a problem, then alternative approaches, such as no-flow or passive sampling described below, should be considered. Where applicable, low-flow sampling of monitoring wells is usually favoured over conventional procedures (e.g., bailers or inertial lift pumps) because minimized disturbance at the well screen during sampling will also minimize volatilization losses and re-suspension of colloidal materials. The procedure also usually reduces the volume and handling of large volumes of purge water. Examples of acceptable low-flow sampling procedures are provided by ASTM (2003), Puls and Barcelona (1996), and Health Canada (2008).

- **No-flow purging and sampling**—No-flow purging and sampling refers to sampling procedures that negate the need for any purging prior to sample collection. Examples include micro-purging, wherein only the sample tubing of, for example, a peristaltic pump is purged prior to sample collection, and discrete downhole samples (e.g., Hydrasleeve™,; and Snap Sampler™), wherein a sampling device is submersed downhole, opened and filled at a discrete depth, and returned to surface for chemical analysis. Sampling using such approaches is predicated on the assumption that the natural horizontal groundwater flux across a monitoring well screen is sufficiently high to develop groundwater chemical conditions in the well that are representative of conditions in the adjacent geologic formation. Such an assumption is likely to be valid in permeable formations (e.g., sands and gravels), but may be invalid in less permeable materials where stagnant water may be present in the well. Where the approach is used, it should be validated for site-specific conditions by comparison with alternative conventional or low-flow procedures. Alternatively, the techniques should be considered to provide screening level information to determine the presence or absence of potential contamination.

- **Passive diffusion sampling**—Passive diffusion sampling refers to a group of sampling devices that are typically composed of elongate semi-permeable membrane bags (often polyethylene plastic), which can be submersed in monitoring wells, allowed to equilibrate, and then withdrawn for chemical analysis. The bag is filled with a liquid (usually distilled water) and inserted to a discrete depth within the well screen of a monitoring well. After allowing a period to achieve chemical equilibrium across the membrane (usually several days), the bag is retrieved and the liquid analysed for the constituents of concern. Single and multi-interval passive diffusion bags are available. Similar to no-purge sampling, passive diffusion bags rely on the assumption that the groundwater in the monitoring well is not stagnant, but rather, represents conditions in the aquifer adjacent to the well screen. Consequently, similar caveats on their use should be applied as those for no-flow sampling.

It is recommended that dates for drilling, well development, and sampling be noted in field notes.

Decontamination between wells

Decontamination is an important step to ensure that contaminants are not carried over from one location to another at a site, and that the potential for cross contamination of samples is minimized. While the focus of this section is on decontamination of equipment, personnel should also exercise appropriate and protective health and safety measures during sampling and while exiting the work area.

The level of effort for decontamination should be commensurate with the level of QA/QC required for a particular site investigation. Typically, a decontamination area or stations should be identified where equipment is taken for cleaning. Each area should be located away from the drilling and sampling locations and in an area where potential dust generation is minimal. The decontamination area is typically covered with a polyethylene tarp or similar protective layer to prevent potential contamination with surface soils.

Equipment requiring decontamination will usually include sampling devices, sample inspection tools, downhole equipment, drill rods or augers, and drill bits. Materials that cannot be completely cleaned should be disposed of properly. At a minimum, the drill rig should be decontaminated before arrival and leaving the site, and between sampling locations, and all sampling equipment should be cleaned between sampling locations. Further precautions may be necessary when, for example, penetrating through contaminated zones into deeper uncontaminated strata.

The methods used for effective decontamination and the fluids used in the process will depend on the level of QA/QC required by the investigation. Typically, at least one wash and several rinses are conducted that will include the use of phosphate-free detergents and de-ionized/distilled water. Where organic compounds are of concern and may be present in high concentrations, then the decontamination procedures may include an organic solvent rinse (e.g., methanol) whereas metal compounds of concern may include an acid rinse. The rinsing agents should not include the contaminants of potential concern. Further information on decontamination is described by ASTM (2002).

Where a series of samples is to be collected using common equipment, sampling should begin with the (assumed) lesser contaminated sites and progress to sites with higher anticipated levels of contamination. Decontamination procedures used should be documented and described in the

investigation report. As part of the QA/QC program, field blanks and trip blanks may be used to assist in assessing the effectiveness of the decontamination and sampling programs.

Field laboratories

With respect to quantitative groundwater chemistry data, data acquisition in the field by an accredited laboratory's field lab can sometimes be beneficial to the program as it can allow timely decisions to be made as the investigation program proceeds. Changes in chemistry resulting from factors such as mass losses are usually minimized because the groundwater samples are preserved, sealed and refrigerated soon after retrieval. The advantages of a field laboratory are often of more significance for analysis of soil rather than groundwater, because soil samples are much more prone to chemical losses resulting from volatilization and degradation. Analysis of regulated parameters through a field laboratory must be completed by an accredited lab otherwise the results can be used for on-site decisions regarding sampling programs, however the samples would need to be followed up by analysis at an accredited lab (Contaminated Sites Regulation – Protocol 2).

Special considerations

Metals

Where groundwater samples are obtained for quantifying metals concentrations, it is important that the samples be filtered in the field under pressure during or immediately after retrieval, and prior to preserving the sample (e.g., with nitric acid). Typically a clean 0.45 micron membrane filter is employed. The sample should be collected in an acid-cleaned plastic container. Because aquifers normally act as filters and prevent significant migration of particulates, analysis of samples containing particulates will not represent actual groundwater conditions. Unfiltered samples, when analysed by a laboratory, will commonly contain elevated metals concentrations because the particulates contain metals and are digested at the laboratory prior to analysis. On the other hand, filtered samples may contain non-representative low metals concentrations if the sample was allowed to sit for some time prior to filtering, allowing dissolved metals to precipitate from the water as a consequence of gas exchange and a rise in redox potential.

NAPLs

Caution should be exercised when drilling, installing and sampling wells suspected to contain NAPL. Many NAPLs are clear and colourless, or are easily missed because they co-dissolve natural organic materials, taking on the same colour as the surrounding medium. If suspected, meticulous care should be taken to avoid cross contamination and drawdown from one water bearing unit to another. Once the well is installed, monitoring should be conducted to determine NAPL presence. Special probes, such as an interface meter, may be inserted into the well to verify the presence and thickness of any LNAPL or DNAPL. Alternatively, special bailers and/or oil-finding pastes may be used.

NAPL characterization is usually best achieved by direct sampling (prior to well purging) and analysis, although assessment of dissolved-phase constituents can often be used successfully to infer NAPL composition. NAPL sampling involves the careful use of special bailers or pumps. Acceptable monitoring approaches are described by API, 1996. Where a NAPL sample is able to be obtained, the sample should be analyzed for both physical properties (e.g. viscosity, density, interfacial tension) and chemical properties (at a minimum a chromatogram). It may also be beneficial to analyze the product sample for a full range of hydrocarbon constituents such that molar fractions of individual components can be derived. This data can in turn be used to estimate effective constituent solubilities and potential worst case constituent vapour concentrations following Raoult's law.

In wells with detected NAPL, groundwater sampling is not advisable because the NAPL may easily become entrained in the water sample, yielding false high concentrations of constituents. Sometimes false high concentrations are obtained in groundwater samples because the NAPL was not obvious. For example, the NAPL may be missed because it is clear and colourless, or because small entrained blebs of NAPL are masked by a silty, cloudy sample.

Where NAPL is present, it is reasonable to assume that groundwater in contact with the NAPL is at a quasi-equilibrium state, where constituent concentrations in groundwater approach their theoretical effective solubility limits, and no laboratory analysis is required. Such limits may be estimated using reference solubility limits for pure-phase chemicals, and known NAPL composition (e.g., USEPA, 1992).

<u>Volatile organic compounds</u>

VOCs comprise a range of organic chemicals that, as their name implies, are volatile and therefore require special consideration during sampling to avoid mass losses to air. Methods that may entrain air in the sample, such as the rigorous (and improper) use of bailers or inertial lift pumps downhole in a well, may entrain air within the sample and strip out VOCs, and should therefore be avoided. Other methods, such as peristaltic pumps, draw a vacuum on the sample water in the downhole tubing, potentially causing degassing and stripping of VOCs. Sometimes bubbles may be observed in the tubing where significant degassing is occurring. VOC samples retrieved using a peristaltic pump from depth greater than about 3 m should be viewed with caution, and treated as screening-level data in absence of quantitative, comparative tests with other acceptable methods. Further precautions should be taken at ground surface to ensure minimal or zero contact between the sample and air. Special VOC bailers are available, for example, to assist in minimizing air exposure during transfer to sample containers such as a standard 40 ML glass VOC sampling vial. With such vials, it is important that no air bubbles are entrained in the sample, as mass transfer to the bubble can also compromise the sample concentrations.

Groundwater samples collected for analyzing organic constituents should not be field-filtered prior to laboratory analysis. The recommended container for collection is a solvent rinsed, amber coloured glass with an aluminum foil or Teflon liner cap. VOCs should be the first sample that is collected following the purging process. Samples should be placed directly in glass bottles with no air space left and capped with a Teflon septum cap. Samples for extractable organics should be collected after the VOCS samples. Glass or Teflon bottles with Teflon lined caps should be used as sample containers.

Selection of analytical tests

The analytical program should focus on resolving the objectives of the investigation program, including the information needs of the risk assessor. Analytical tests should be selected to address not only the known or suspected contaminants of concern at a site (e.g., the chemical constituents initially released to the subsurface), but also the potential contaminants that may form in the subsurface as a consequence of chemical or biological transformation (e.g., vinyl chloride from trichloroethene), or changes in geochemical conditions (e.g., decreasing redox potential, leading to

dissolution of metals). For example, increased concentrations of manganese and other metals in groundwater can often result from the geochemical reduction of metals to their more soluble form, as a consequence of biodegradation of organic substrates such as petroleum hydrocarbons.

In addition to analytical tests associated with the contaminants and their transformation, consideration should be given to measurement of other variables, such as the concentrations of major ions (e.g., sodium, calcium, magnesium, chloride, sulphate, bicarbonate and carbonate) and isotopes (e.g., tritium, carbon 13), to the extent that they can assist in defining the subsurface groundwater flow regime or contaminant transport and fate.

Consideration should be made to hold times for each parameter between sample collection and analysis especially at remote locations with longer travel distances. Most parameters have a maximum hold time of between 5 and 10 days. Some parameters such as Total and Fecal Coliforms, Biological Oxygen Demand and Dissolved Organic Carbon have hold times of approximately 24 hours. Where this analysis is required, plans should be made for immediate transport and use of the closest available laboratory.

Sample preservation

To assist in maintaining the natural chemistry of a sample, it is necessary to preserve the sample. Methods of sample preservation are relatively limited and are intended to reduce the effects of chemical reactions, the effects of sorption and to arrest biological actions. Preservation methods are generally limited to pH control, refrigeration, and protection from light. Selected parameters or groups of parameters (e.g., metals) may be preserved by addition of a reagent (e.g., acid) that stabilizes their concentration but may preclude the analysis of that sample for other parameters. Glass, stainless steel, Teflon, or plastic (polyethylene and polypropylene) are the types of containers acceptable for most kinds of sample collection. There are some exceptions to this general rule; for example, plastic is not acceptable for organics and stainless steel is not acceptable for metals. Containers should be kept full until samples are analyzed to maintain anaerobic conditions. The sample container material should be non-reactive with the sample and especially with the particular analytical parameter to be tested. Sample containers used to transport samples to the lab must undergo pre-treatment procedures. Pre-treated containers may be purchased commercially; however, pre-treatment must be repeated if they are re-used. Samples should be

placed in bottles immediately upon collection and, where preservation of the sample is required, it should be carried out immediately. Handling of the sample and contact with the atmosphere should be kept to a minimum. The samples should be properly packaged so as to prevent breakage and should generally be kept at 4°C plus/minus 2°C until analyzed by the laboratory.

For some analyses with short holding times (3 days or less) it may be difficult to collect and transport the samples to the laboratory within the recommended hold times. One sample preservation technique that may be possible for some parameters collected from remote, northern locations is the storage of groundwater samples at sub-zero temperatures (USEPA, 1983). Samples should not be frozen in glass containers and a sufficient air gap should be present in the container to allow for expansion during freezing. They should be thawed slowly to an ambient temperature and mixed before analysis. The sample should be analyzed quickly following thawing to prevent subsequent instability (Philbert, 1973). It is recommended that the sampler consult with the laboratory to discuss sampling protocols and sample treatment options prior to sample collection.

Data validation and quality assurance/quality control

Data validation and quality assurance/quality control (QA/QC) are important considerations for groundwater investigation programs. Care should be taken to use appropriate and consistent field procedures, and to quantify analytical data using approved methods by an accredited laboratory. Data quality objectives should be established at the beginning of the field program, and the data should be compared against these objectives for completeness of the data set, and to define the approximate level of precision and accuracy for decision-making purposes. Commonly, for groundwater investigation studies at least 10% of the samples or one sample per batch, if less than ten, are obtained in duplicate for assessment of reproducibility. Field equipment blanks and/or travel blanks should also be collected and submitted for analysis to confirm the presence or absence of cross contamination during field activities, travel or laboratory analysis. Investigation reports should always include a discussion of QA/QC, including an assessment of sample variance, and the consequent level of uncertainty that should be attached to the more critical variables that may be considered in a subsequent action such as remediation or risk assessment. The contaminant concentrations in blanks should be recorded, and if concentrations are more than an

order of magnitude greater than the detection limit for the parameter and the sample result is less than 5 times detection limit, the groundwater should be resampled to ensure QA and QC standards have been satisfied.

In addition to field duplicates, it is good practice to **obtain at least two groundwater samples on different days from any monitoring well** prior to making decisions based on the chemistry data. Groundwater chemistry may change over time at a particular location as a result, for example, of seasonal changes in flow direction and/or changes to the saturated thickness of the aquifer. Where a monitoring well is sampled and found not to be contaminated, a second sample should be considered for analysis to provide redundancy in the data prior to well decommissioning. Sampling over more than one season may be appropriate in some cases, but not necessary in many cases. This should be addressed on a case-by-case basis, taking into consideration local hydrogeological conditions.

Appendix B

Groundwater investigation methods

Table B-1: Investigation methods and groundwater data quality

Sampling Method		Relative Data Quality	Relative Resolution of Scale			Comment
			Spatial	Temporal	Chemical	
Direct Methods	Monitoring Wells	Quantitative	Poor	Good	Good	▪ Sample represents an average over the well completion interval ▪ Suitable for long-term monitoring to establish trends ▪ In addition to samples, provides hydraulic information (e.g., water levels) ▪ Commonly available technology suitable for most geologic conditions
	Mini-piezometers	Quantitative	Poor to Good	Good	Poor to Good	▪ As above; however, usually limited to shallow water table aquifers. Many piezometers can be deployed to resolve lateral spatial scales ▪ Sample volumes typically small, which can limit range of chemicals analysed

Sampling Method	Relative Data Quality	Relative Resolution of Scale			Comment
		Spatial	Temporal	Chemical	
Well Points	Quantitative	Poor	Good	Poor to Good	▪ Same as per mini-piezometers
Direct-Push Groundwater Samplers (e.g., Waterloo Profiler)	Quantitative	Good	Poor	Poor to Good	▪ Discrete groundwater samples acquired along vertical profile ▪ Sample volumes typically small, which can limit range of chemicals analysed ▪ Not suitable in dense tills, cobbly soils or bedrock

Sampling Method		Relative Data Quality	Relative Resolution of Scale			Comment
			Spatial	Temporal	Chemical	
Indirect Methods	Discrete Soil Samples	Semi-Quantitative	Good	Poor	Poor to Good	▪ Detection limits usually much higher in soil than groundwater ▪ Porewater concentration must be estimated ▪ Soil sampling technologies are common and available
	Passive Diffusion Bags	Quantitative	Poor	Poor to Good	Poor to Good	▪ Effective for several groundwater constituents, but not for all ▪ Provides an average of concentrations over the period of deployment
	Direct Push Profilers (general)	Qualitative to Quantitative	Good	Poor	Poor	▪ Non- or semi-quantitative data need to be correlated with analytical chemistry data for meaningful results ▪ Many are not suitable in dense tills, cobbly soils or bedrock
	Membrane Interface Probe (MIP)	Semi-Quantitative	Good	Poor	Poor to Good	▪ Targets in situ concentrations of volatile organic chemicals (VOCs) in soil along a vertical profile ▪ Not suitable in dense tills, cobbly soils or bedrock
	Laser or Ultraviolet-Induced Fluorescence (LIF /UVIF)	Qualitative to Semi-Quantitative	Good	Poor	Poor	▪ Targets in situ concentrations of susceptible compounds (e.g., fluorescent aromatic and poly-aromatic hydrocarbons) in soil along a vertical profile
	Field Observations	Qualitative	Poor to Good	Poor to Good	Poor	▪ Data should be correlated with analytical chemistry data ▪ Detailed descriptions over continuous sampled intervals (e.g., continuous soil or rock cores) preferable

Sampling Method	Relative Data Quality	Relative Resolution of Scale			Comment
		Spatial	Temporal	Chemical	
Geophysics: surface (e.g., electromagnetic), down hole	Qualitative	Poor to Good	Good	Poor	▪ Data should be correlated with hydrostratigraphic and analytical chemistry data ▪ Applicable to most sites, although often subject to interferences (e.g., structures, buried utilities)

Direct-push technologies for groundwater characterization

Direct-push technologies include a variety of methods to obtain information on subsurface conditions such as soil stratigraphy, engineering properties, and soil and groundwater chemistry. Environmental samples may be acquired using direct-push technologies, or information may be obtained *in situ* using specialized downhole tools or equipment. A brief summary of some of the more common direct-push technologies currently in use in North America is provided below. A good discussion of range of available direct-push technologies, and the advantages and limitations of the technologies, is provided by Nielsen (2006, Chapter 6). Further information may be found in the referenced materials and links below.

Stratigraphic profiling

Stratigraphic profiling using direct-push technologies was pioneered by the Dutch in the 1930s, with the development of the Dutch Cone to determine bearing capacity of soils *in situ*. Since that time, cone penetrometer testing (CPT) has evolved into a common technology used in many geotechnical investigations to obtain information on subsurface stratigraphy and engineering soil properties. CPT procedures typically comprise attaching an electronic cone to the tip of a drill string, which is pushed into the subsurface by hydraulic rams mounted on a relatively heavy cone truck. Because the cone displaces soil rather than excavating the soil, no drill cuttings are produced, and therefore there are no soil handling or disposal costs. Electronic data generated by the cone may include soil resistivity (to infer soil moisture content), skin friction (to measure soil cohesive strength), and piezometric head (i.e., hydraulic head). The data are typically acquired at a

resolution of a few centimetres or less, yielding a very detailed vertical profile of soil properties and inferred stratigraphy. Depths of 30 m or more may be profiled under favourable soil conditions.

Over the past decade, specialized sampling tools and procedures have been developed to obtain multiple groundwater samples along a vertical profile, and *in situ* measurements of soil chemical conditions. Common direct-push technologies include drive point profilers (e.g. the Waterloo Profiler™), laser-induced fluorescence (LIF), membrane interface probes (MIP), and others (e.g.,US EPA, 2005). A few of these are further discussed below.

Groundwater profiling

Groundwater profiling gained prominence in the late 1980s and early 1990s with the development of the Waterloo Profiler™ (Pitkin et *al*., 1999). Drive-point profilers (e.g. Waterloo Profiler™ or Geoprobe™) comprises a steel tip with small-diameter screened ports connected to small-diameter tubing (typically quarter-inch). The tip is fitted to a hollow drill string (e.g., "A" rods), with the tubing running up the hollow centre of the rods to ground surface, where a groundwater sample may be acquired into a vial using a peristaltic pump. During tip advancement, water may be pumped at very low flow rates downhole and into the probe to assist in keeping the screened ports open and silt-free. During a typical application, groundwater samples are obtained at depth intervals of 0.3 m to 0.5 m, providing relatively good resolution of the groundwater profile. The technology can be very useful where the water table is relatively shallow (the use of a peristaltic pump limits the effective depth of the water table to a few metres or less below ground surface), and where small sample sizes are adequate for chemical analysis (e.g., 40 mL samples, although larger sample sizes can be obtained). Caution is advised at highly contaminated sites, where there is some possibility of contaminant dragdown, leading to an overestimate of the thickness of the contaminated zone.

Other technologies are available that can be used to develop groundwater profiles include discrete downhole samplers (e.g. Hydropunch sampler) which can be deployed using a hollow-stem auger drill rig.

Laser-induced fluorescence

Laser-induced fluorescence (LIF), sometimes referred to as ultraviolet-induced fluorescence (UVIF), is a technology based on variable or fixed wavelength lasers (typically an ultraviolet wavelength). The laser transmits optic pulses into an optic fibre, which runs down a CPT drill string to a 6.4 mm diameter sapphire window that is mounted flush with the probe rod, approximately 0.6 m above a standard CPT cone. The ultraviolet light excites molecules of aromatic hydrocarbons that may be present in soil at the window, and causes them to fluoresce. Emitted light is carried back to a detector at ground surface via a second optic fibre. The spectral intensity of the fluorescence can be directly related to the concentration of the aromatic hydrocarbons present, allowing concentrations to be quantified. In field applications, LIF results are often calibrated in the field by comparison against soil concentrations in samples obtained from an adjacent borehole. Contaminants that can be measured using LIF technology include petroleum hydrocarbons (e.g., gasoline, diesel, kerosene), coal tars, creosote, and any other liquid containing significant concentrations of aromatic hydrocarbons.

Membrane interface probe

The membrane interface probe (MIP) comprises a semi-permeable membrane mounted flush with the side of a cone. After pushing the cone to the desired depth, the membrane is heated to between about 100°C and 125°C, promoting diffusion of VOCs in the soil across the membrane into the probe, where a carrier gas sweeps the inside of the membrane and carries the gas to surface. Detectors at surface record VOC concentrations in the gas, as well as soil electrical conductivity and temperature. VOC concentrations may be measured semi-quantitatively using various detectors such as photo-ionization detectors (PID), flame ionization detectors (FID) and electron capture detectors (ECD). Quantitative measurements may be made by coupling the system with a GC mass spectrometer.

The MIP has become a relatively common direct-push technology for the *in situ* quantification of volatile organic compound (VOC) concentrations in soil, and to infer the presence of LNAPL and DNAPL. Measurements are commonly made over short depth intervals (about 0.3 m intervals), providing a vertical profile or log of concentrations with depth. Examples are the Geoprobe MIP or

Cascade Environmental Membrane Interface Hydrdaulic Profiling,

Other technologies

A range of tools have been or are currently under development to provide quantitative *in situ* measurements of specific compounds or groups of compounds. As discussed by Nielsen, 2006 (Chapter 6), these include, for example:

- Fuel fluorescent detectors for sensing petroleum hydrocarbons
- CPT-based Raman spectroscopy to detect a variety or compounds including, metals and metals complexes, DNAPLs (e.g., trichloroethene or TCE, and tetrachloroethene or PCE)
- Metals sensors using x-ray diffraction (XRF) or laser-induced breakdown spectroscopy
- Explosives sensors to characterize soil containing various nitro-aromatic explosives materials

Appendix C

Checklist for detailed groundwater assessments

Issue	Yes	No	N/A	Page or Section
Does the report address the following:				
3) Are the objectives of the groundwater investigation program clearly stated				
4) Has available information from earlier investigative phases (e.g., groundwater well data, regional hydrogeology, surficial geology, utilities) been integrated in the work plan				
5) Given the objectives and each APEC identified, is the groundwater investigation appropriate with respect to:				
a) **Vertical spatial scale**: Maximum saturated well screen length 1.8 m or less				
b) **Horizontal spatial scale**: Maximum well separations for suspected or known plumes of 10 m in longitudinal and 5 m in transverse direction				
c) **Temporal scale**: Sufficient monitoring to characterize temporal variability				
d) **Chemicals**: All COPCs and transformation products; inorganic constituents and geochemical parameters, where warranted				
e) **NAPL zones:** Do sampling locations consider variability in NAPL source zones and possible NAPL migration pathways				
6) Was an assessment of background groundwater quality conducted				
If not, is a background study warranted				
7) Has complementary data been obtained on soil stratigraphy or hydrostratigraphic units where warranted (e.g., through deep boreholes or collection of soil cores)				
8) Were appropriate methods utilized to obtain groundwater data				
Were methods adequately documented				
9) Were appropriate QA/QC methods used and documented				
a. Was an accredited laboratory used				

Issue	Yes	No	N/A	Page or Section
b. Were field duplicates, equipment blanks and trip blanks acquired and submitted as appropriate				
c. Does reporting include a discussion of QA/QC?				
10) Does the groundwater data analysis and interpretation include:				
a) Summary of sampling strategy and design and whether representative data was obtained				
b) Integration of historical information and investigation results to identify potential contamination sources and different contaminant plumes that may exist				
c) Depths to water table				
d) Seasonal variation in water table				
e) Physical extent of and likely boundaries to aquifer(s) of interest (thickness of each unit and lateral extent)				
f) Hydraulic properties of each aquifer and aquitard				
g) Regional and local groundwater flow directions; seasonal variation in flow direction				
h) Groundwater flow rates				
i) Groundwater recharge and discharge zones				
j) Dissolved plume extent and mobility				
k) Free-phase NAPL and residual NAPL extent and potential mobility				
l) Possible influence of background levels in the surrounding area for contaminants that occur naturally or that may have been deposited by non-point sources				
11) Figures providing the following:				
a) Piezometric heads in each aquifer of interest posted on plan and head contours and groundwater flow direction, where appropriate				
b) Stratigraphic cross-sections longitudinal and transverse to groundwater flow direction that include interpolated extent of				

Issue	Yes	No	N/A	Page or Section
identified strata, physical hydrogeologic data, water levels, soil sample locations, and well completion intervals				
c) Chemical concentrations in groundwater posted beside measurement locations on plans and cross sections (or shown in table on plans and sections) with reference to applicable standards, and concentration contours, where appropriate.				