Final
Technical Impracticability Evaluation Report
Savage Municipal Water Supply Well
Superfund Site
Operable Unit 1

621 Elm Street
Milford, New Hampshire

Prepared By:

Weston Solutions, Inc.
45 Constitution Avenue
Suite 100
Concord, New Hampshire 03301

Prepared For:

Federal Sites Section
Hazardous Waste Remediation Bureau
Waste Management Division
New Hampshire Department of Environmental Services
29 Hazen Drive
Concord, New Hampshire 03302-0095

July 2015

SDMS Doc ID 578390

FINAL
TECHNICAL IMPRACTICABILITY EVALUATON REPORT
SAVAGE MUNICIPAL WATER SUPPLY WELL
SUPERFUND SITE
OPERABLE UNIT 1

621 ELM STREET
MILFORD, NEW HAMPSHIRE
NHDES SITE NO. 198505002

Prepared for:

THE HAZARDOUS WASTE REMEDIATION BUREAU
WASTE MANAGEMENT DIVISION
NEW HAMPSHIRE DEPARTMENT OF ENVIRONMENTAL SERVICES
29 Hazen Drive
Concord, New Hampshire

WESTON - Project Manager
Vincent DelloRusso, P.G.

7/27/15
Date

WESTON – Senior Technical Director
Frederick Symmes, P.E.

7/27/15
Date

STATE OF NEW HAMPSHIRE
FREDERICK R. SYMMES
No. 9180
LICENSED
PROFESSIONAL ENGINEER

TABLE OF CONTENTS

Section **Title**

1. INTRODUCTION 1-1
1.1 PURPOSE AND ORGANIZATION OF THE REPORT 1-3
1.2 SITE BACKGROUND INFORMATION 1-3
1.2.1 Site Description and Setting 1-3
1.2.2 Site History 1-4

2. SITE-SPECIFIC APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS 2-1

3. CONCEPTUAL SITE MODEL 3-1
3.1 SITE GEOLOGY AND HYDROGEOLOGY 3-1
3.2 NATURE AND EXTENT OF CONTAMINATION 3-3
3.2.1 Soil 3-4
3.2.2 Overburden Groundwater 3-5
3.2.3 Shallow Bedrock Groundwater 3-5
3.2.4 Deep Bedrock Groundwater 3-6
3.3 CONTAMINANT FATE AND TRANSPORT 3-10
3.4 CONTAMINANT MIGRATION PATHWAYS/ASSESSMENT OF SOURCE AREAS 3-12
3.5 CONCEPTUAL SITE MODEL SUMMARY 3-13

4. EVALUATION OF RESTORATION POTENTIAL 4-1
4.1 ASSESSMENT OF MASS RELEASED 4-1
4.2 MASS REMOVAL BY PREVIOUS REMEDIAL ACTIONS 4-2
4.3 FACTORS LIMITING REMEDIATION 4-4
4.4 SPATIAL EXTENT OF THE TECHNICAL IMPRACTICABILITY ZONE 4-6
4.5 EVALUATION OF POTENTIAL REMEDIAL TECHNOLOGIES 4-8
4.6 PROTECTIVENESS UNDER A TI ARAR WAIVER 4-9

5. CONCLUSIONS 5-1

6. REFERENCES 6-1

APPENDIX A GROUNDWATER MODELING USING BIOCHLOR

APPENDIX B DESCRIPTION OF REMEDIAL TECHNOLOGIES

LIST OF FIGURES

Title

Figure 1-1 Site Location Map

Figure 1-2 Site Plan

Figure 1-3 OU1 Site Detail

Figure 3-1 Historic 1990 PCE Overburden Plume

Figure 3-2 PCE Trends in OU1 Outside the Slurry Wall

Figure 3-3 PCE Concentrations in Shallow Overburden Groundwater Fall 2014

Figure 3-4 PCE Concentrations in Deep Overburden Groundwater Fall 2014

Figure 3-5 Bedrock PCE Concentration Trends

Figure 3-6 Lateral Extent of PCE in OU1 Bedrock

Figure 3-7 Lateral Extent of PCE in Bedrock Site-Wide

Figure 3-8 Vertical Distribution of PCE in Bedrock

Figure 3-9 Conceptual Site Model

Figure 4-1 Lateral Extent of TI Zone

Figure 4-2 Vertical Extent of TI Zone, Cross-Section X-X’

Figure 4-3 Vertical Extent of TI Zone, Cross-Section Y-Y’

LIST OF TABLES

Title

Table 1-1 Groundwater Chemicals of Concern and Associated Interim Cleanup Levels and Ambient Groundwater Quality Standards, Savage Municipal Water Supply Well Superfund Site – OU1 & OU2

Table 4-1 Summary of OU1 Groundwater Treatment Plant Volatile Organic Compound Mass Removal

Table 4-2 General Response Actions and Technologies

Table 4-3 Remedial Technology Screening Results

LIST OF ACRONYMS

°	degree
1,1-DCA	1,1-dichloroethane
1,1-DCE	1,1-dichloroethene
1,1,1-TCA	1,1,1-trichloroethane
AGQS	Ambient Groundwater Quality Standards
ARAR	Applicable or Relevant and Appropriate Requirements
AS	air sparging
bgs	below ground surface
CD	Consent Decree
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
C.F.R.	Code of Federal Regulation
cis-1,2-DCE	cis-1,2-dichloroethene
COC	contaminants of concern
CSM	Conceptual Site Model
cy	cubic yards
DNAPL	dense non-aqueous phase liquid
EPA	United States Environmental Protection Agency
ESD	Explanation of Significant Differences
FS	Feasibility Study
ft	feet/foot
gpm	gallons per minute
Gradient	Gradient Corporation
Hendrix	Hendrix Wire and Cable Company
HMM	HMM Associates
IC	institutional controls
ICL	Interim Cleanup Level
ISCO	*in situ* chemical oxidation
ISTT	*in situ* thermal treatment
lbs	pounds
MCL	Maximum Contaminant Level
µg/L	micrograms per liter
mg/L	milligrams per liter
MNA	monitored natural attenuation
NCP	National Oil and Hazardous Substance Pollution Contingency Plan
NH	New Hampshire

LIST OF ACRONYMS
(Concluded)

NHDES	New Hampshire Department of Environmental Services
NPDES	National Pollution Discharge Elimination System
NRC	National Research Council
O&M	operation and maintenance
OK Tool	OK Tool Company
OU1	Operable Unit 1 (Area of the Former OK Tool Property)
OU2	Operable Unit 2 (Extended Plume)
PCE	tetrachloroethene
%	percent
PRP	Potentially Responsible Parties
RAO	remedial action objectives
RG	remediation goals
RI Report	Remedial Investigation Report
ROD	Record of Decision
Site	Savage Municipal Water Supply Superfund Site
SVE	soil vapor extraction
TCE	trichloroethene
TI	Technical Impracticability
trans-1,2-DCE	trans-1,2-dichloroethene
VOC	volatile organic compound
WESTON®	Weston Solutions, Inc.
WSPCC	Water Supply and Pollution Control Commission

SECTION 1

INTRODUCTION

1. INTRODUCTION

On behalf of the New Hampshire Department of Environmental Services (NHDES), Weston Solutions, Inc. (WESTON®) has prepared this Technical Impracticability (TI) Evaluation Report for groundwater contamination within Operable Unit 1 (OU1) at the Savage Municipal Water Supply Well Superfund Site (Site) in Milford, New Hampshire (NH) (see Figures 1-1, 1-2, and 1-3). This evaluation is based on site information contained in the 2014 Remedial Investigation Report (RI Report) for OU1 prepared by WESTON (WESTON, 2014). The 2014 RI Report summarizes and interprets data from investigations performed by WESTON, Gradient Corporation (Gradient) and several other consultants/agencies, including historical data from HMM Associates (HMM), the United States Geological Survey, and the United States Environmental Protection Agency (EPA). A Conceptual Site Model (CSM) is presented in this report that incorporates site data and interpretation from the 2014 RI Report. The CSM forms the basis of the evaluation of whether groundwater restoration is technically impractical and will be used by EPA in a TI waiver determination under the National Contingency Plan (NCP) regulations, 40 Code of Federal Regulation (C.F.R.) Part 300, promulgated under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, also referred to as "Superfund"), 42 U.S.C. §§ 9601 *et seq*. This evaluation is consistent with EPA's *Guidance for Evaluating the Technical Impracticability of Groundwater Restoration, Interim Final* (EPA, 1993).

This TI Evaluation Report is intended as a supplemental attachment to the Feasibility Study Report and evaluates the practicability of returning groundwater within all or parts of OU1 to its beneficial use, as established by applicable or relevant and appropriate requirements (ARARs), or by risk-based standards where no ARARs exist. As identified in the EPA TI Guidance, the NCP provides the regulatory framework for the Superfund program and states at 40 C.F.R. § 300.430 that:

> "EPA expects to return usable ground waters to their beneficial uses wherever practicable, within a timeframe that is reasonable given the particular circumstance of the site."

Under Superfund, restoration cleanup levels are established by ARARs, such as federal or state drinking water standards or by federal risk-based standards where no ARARs have been

promulgated for a particular groundwater contaminant. However, as explained in the EPA TI Guidance, EPA acknowledges that restoration to drinking water quality may not always be achievable due to the limitations of available remedial technologies. The EPA TI Guidance identifies three general categories of factors that may inhibit groundwater restoration:

- Hydrogeologic limitations such as complex sedimentary deposits, aquifers of very low permeability, fractured bedrock aquifers and other factors that make extraction or *in-situ* treatment of contaminated groundwater extremely difficult.
- Contaminant-related factors including properties that allow the contaminants to become sorbed onto or lodged within the soil or rock aquifer that may limit the success of an extraction or *in-situ* treatment process. The presence of dense non-aqueous phase liquids (DNAPL) is one example.
- Inadequate remediation system design and implementation can inhibit groundwater restoration, but it is generally not considered by EPA to be sufficient justification for a determination of TI.

Limitations related to the first two items exist within OU1 at the Site. Use of chlorinated solvents at the OK Tool Company (OK Tool) has resulted in releases of DNAPL to a hydrogeologically complex environment that includes a thick sequence of highly permeable overburden with embedded low permeability lenses, low permeability till, and fractured bedrock. Solvents including tetrachloroethene (PCE) and trichloroethene (TCE) and their breakdown-products have been detected in site groundwater at persistently-high concentrations that suggest the presence of DNAPL in both overburden and bedrock. Given the documented extreme difficulty of fully restoring DNAPL zones and associated dissolved-phase contamination in hydrogeologically complex environments to beneficial use (i.e., drinking water standards) (Interstate Technology & Regulatory Council, 2011), a waiver of ARARs and risk-based standards for groundwater in a section of OU1 where contaminant levels are highest (referred to as the “proposed TI Zone”) is warranted.

WESTON has conducted a comprehensive review of the existing data and CSM in regard to the hydrogeologic and contaminant-related properties outlined in the EPA TI Guidance to assess the likelihood that site groundwater can be restored to beneficial use within a reasonable timeframe within the proposed TI Zone. The purpose of this report is to present the results of the TI

evaluation. Alternative protective remedial strategies are evaluated as part of the Feasibility Study Report to which this TI Evaluation Report is attached.

1.1 PURPOSE AND ORGANIZATION OF THE REPORT

The evaluation summarized in this TI Evaluation Report was conducted in accordance with the EPA TI Guidance (EPA, 1993). In general, the data and evaluations presented are based on site characterization data presented in the 2014 RI Report, which compiles and interprets data from site investigations performed by WESTON and other consultants under contract to NHDES. Further evaluation and interpretation has been performed here to evaluate the technical impracticability of restoring the site groundwater within a portion of OU1 in the proposed TI Zone. The TI Evaluation Report is structured as follows:

- Section 1 provides a summary of background information and history of the Site, a more detailed description of which can be found in the 2014 RI Report.
- Section 2 describes the groundwater chemical-specific ARARs and risk-based standards for site groundwater contaminants that would be waived if it is determined that it is technically impracticable to restore groundwater in portions of the Site.
- Section 3 presents the CSM in terms of geology and hydrogeology, nature and extent of contamination, and contaminant fate and transport properties. Two-dimensional graphical representations of the CSM are presented.
- Section 4 evaluates the groundwater restoration potential, including assessment of the remaining contaminant mass, evaluation of past remedial actions, factors precluding remediation of source areas, evaluation of potential remedial technologies including costs and protectiveness, and predicted analysis of cleanup time. Section 4 also defines the proposed area over which the TI Waiver would apply (proposed TI Zone).
- Section 5 presents the conclusions of the TI Evaluation Report.
- Section 6 provides references.

1.2 SITE BACKGROUND INFORMATION

1.2.1 Site Description and Setting

The Site is located in Milford, NH approximately 2 miles west of the center of town as shown on Figure 1-1. The Site extends beyond the Souhegan River on the north and east and is roughly bounded on the south by Old Wilton Road and Tucker Brook (HMM, 1991a; EPA, 2011a). Site geology consists of an approximately 50- to 110-foot (ft)-thick, highly-transmissive sand and

gravel glacial outwash above a relatively thin discontinuous till layer that overlies fractured crystalline bedrock. The static water table at the Site ranges from approximately 5 to 15 ft below ground surface (bgs). Much of the Site lies within the floodplain of the Souhegan River and the dominant overburden groundwater flow direction is to the east, in the general direction of the Savage Municipal Water Supply Well (Savage Municipal Well).

First developed in 1960, the Savage Municipal Well provided potable drinking water to approximately 10,000 residents in the Town of Milford, NH. In February 1983, as part of the first routine sampling of water supplies for organic compounds mandated by the Safe Drinking Water Act, the New Hampshire Water Supply and Pollution Control Commission (WSPCC) discovered volatile organic compounds (VOC) above drinking water standards in samples collected from the Savage Municipal Well. Chemicals identified in the water supply included PCE, TCE, 1,1,1-trichloroethane (1,1,1-TCA), trans-1,2-dichloroethene (trans-1,2-DCE), and 1,1-dichloroethane (1,1-DCA). Several of these compounds were also detected in water samples collected from a separate water supply well for a nearby mobile home park. Use of both water supply wells was discontinued in 1983 and a replacement well for the Town of Milford was drilled outside the impacted area. EPA conducted an emergency removal action to connect the mobile home park to the municipal water supply system in 1984. The Site was added to the EPA National Priorities List on 1 September 1984 (40 C.F.R. § 300. [SWH-FRL-2646-2]).

1.2.2 Site History

Following discovery of the VOC contamination in the Savage Municipal Well, the New Hampshire WSPCC, in conjunction with the Division of Public Health Services, undertook regulatory inspection of industrial facilities located upgradient of the Savage Municipal Well in an effort to identify the source of the contamination. In 1985, WSPCC issued a report of their hydrogeologic study of the area, and EPA identified a group of local industries as Potentially Responsible Parties (PRP) consisting of OK Tool, Hitchiner Manufacturing Company, Hendrix Wire and Cable (Hendrix), and New England Steel Fabricators, Inc. The PRP Group agreed to conduct a Remedial Investigation/Feasibility Study (RI/FS) for the Site. The 1991 RI/FS study was performed by HMM and completed in June 1991 (HMM, 1991a; HMM, 1991b). The RI/FS focused largely on identifying and defining source-area soil contamination and groundwater contamination within the unconsolidated overburden aquifer. Based on the results of the RI/FS, a

Proposed Plan for cleanup of the Site was released in July 1991 (EPA, 1991a). Following a public comment period, EPA then issued the Record of Decision (ROD) in September 1991 (EPA, 1991b).

The 1991 FS conducted for the Site divided the plume into two zones: the Concentrated Plume and the Extended Plume. On 27 June 1994, a Consent Decree (CD) governing the cleanup of the Site was entered in federal court in NH as *United States of America and State of New Hampshire v. Conductron Corporation d/b/a Hendrix Wire and Cable et al.*, Civil Action No. 94-174L, 27 June 1994 (D.N.H., 1994). The Concentrated Plume near the former OK Tool facility is defined in detail in the CD. The Extended Plume is also defined in the CD and includes the remaining area of the Site where lower VOC concentrations are typically found.

Under the terms of the CD, the Settling Defendants agreed to perform the remedy selected in the ROD in the area of the Extended Plume, and EPA and the State of NH agreed to perform the selected remedy in the area of the Concentrated Plume. Table 1-1 presents the Groundwater Contaminants of Concerns (COCs) identified in the ROD and the associated Interim Cleanup Levels (ICLs) developed in the ROD.[1] The ROD ICLs were based on EPA's Maximum Contaminant Levels (MCL) for drinking water at the time the ROD was signed. Those standards also met the NHDES Ambient Groundwater Quality Standards (AGQS), because the state standards at the time were equal to the MCLs.

The selected remedy included extraction and treatment of contaminated groundwater from the aquifer and the natural attenuation of contaminated groundwater in the furthest downgradient portion of the plume. More specifically, the ROD identified five components of the selected remedy: extraction and treatment of the Concentrated Plume; extraction and treatment of a portion of the Extended Plume; natural attenuation; environmental monitoring; and institutional controls (ICs). Unsaturated soil was not identified in the 1991 RI as a media of concern and was not addressed in the ROD.

[1] Going forward as part of the Amended ROD or any other decision document, consistent with current Region 1 practice, the term ICLs will be changed to Cleanup Levels (CLs). None of the numeric groundwater cleanup values identified in the ROD as ICLs will be changed.

In 1996, EPA issued an Explanation of Significant Differences (ESD) detailing changes to the selected remedy (EPA, 1996). The ESD divided the remedy into two operable units based upon the Concentrated Plume and the Extended Plume. The portion of the plume where the levels of groundwater contamination were the highest (i.e., the Concentrated Plume) was designated OU1 and included the OK Tool Source Area (as defined in detail in the CD) and a portion of the Extended Plume to the north (see Figure 1-2). The remaining portion of the Extended Plume was designated OU2. The ESD only included changes to the selected remedy for OU1. Figure 1-2 is a Site Plan showing the entire Site and includes the OU1/OU2 boundary (note that OU2 includes all areas outside OU1 where contamination has migrated).

The remedy at OU1, as modified by the ESD, included a subsurface slurry wall (constructed of soil-bentonite slurry) to isolate the areas that exhibited the highest concentrations of COCs, groundwater extraction wells (two inside and two outside the slurry wall) to provide hydraulic containment and accelerate remediation of groundwater outside the wall, treatment of the extracted groundwater via air stripping and carbon adsorption, soil vapor extraction (SVE) with air sparging (AS) to remove near-surface sources within the slurry wall, reinjection of treated groundwater via two injection wells and a recharge gallery, groundwater monitoring, and ICs. The remedial system was designed to maintain hydraulic containment of the impacted groundwater inside the slurry wall and within the underlying shallow bedrock via groundwater extraction from two interior wells (IW-1 and IW-2) screened across the deep overburden aquifer. Treated groundwater was injected inside the slurry wall to accelerate flushing of the contamination and to prevent the formation of stagnation zones within the barrier. Figure 1-3 shows site details within OU1.

Construction of the slurry wall was completed in 1998, and operation of the groundwater extraction, treatment, and reinjection system began in April 1999. The AS/SVE system was operated only intermittently from 1999 through 2008 due to high groundwater elevation conditions that limited the thickness of the unsaturated zone and thereby the effectiveness of the SVE system. Use of the AS/SVE was permanently discontinued in 2008 in lieu of *in situ* chemical oxidation (ISCO) as described below. Use of the exterior groundwater extraction wells was discontinued in 2007 when contaminant concentrations in overburden monitoring wells

outside the wall approached the ROD ICLs. The interior extraction wells and exterior recharge gallery remain in use at OU1 (although in a different configuration, as described further below).

Two additional treatment technologies were implemented within the slurry wall to reduce contaminant mass and thereby enhance groundwater treatment inside the slurry wall. One treatment occurred in 2008/2009 and included excavation and on-site treatment of newly discovered soil contamination associated with a former leach field and drain pipe related to the former OK Tool facility that exceeded the NHDES Soil Remediation Standard of 2 milligrams per kilogram. Concentrations of site contaminants in vadose zone soils above the Soil Remediation Standards associated with these two areas were thought to represent an ongoing source of shallow overburden groundwater contamination.

The other supplemental treatment was an ISCO program implemented between 2003 and 2010. The ISCO program has included the injection of approximately 32,000 pounds (lbs) of potassium permanganate and approximately 307,000 lbs of 40 percent (%) sodium permanganate inside the slurry wall. The ISCO injections targeted deep overburden areas with elevated concentrations of COCs and geologic lenses that exhibit lower hydraulic conductivities (which would therefore receive less treatment via the existing groundwater extraction system). The deep overburden was targeted to reduce contaminant mass and provide a passive treatment barrier between the overburden and bedrock aquifers within the area encompassed by the slurry wall. The overburden areas with lower hydraulic conductivity were targeted in order to reduce the potential for recurrent reverse matrix diffusion of contaminants.

During the ISCO program, pumping of groundwater from inside the slurry wall was continued in order to minimize the risk that contaminated groundwater from inside the slurry wall would migrate to areas outside the slurry wall. In order to minimize the risk that permanganate-containing groundwater inside the wall would be drawn into the deep groundwater extraction wells, two new shallow overburden extraction wells (IW-1A and IW-2A) were installed in 2008 inside the slurry wall. The two new wells are screened above the zones where permanganate was injected and replaced deep overburden extraction wells IW-1 and IW-2, which were screened in the ISCO treatment zone. A third shallow extraction well (IW-3A) was installed in 2012 because of the repeated presence of permanganate in the treatment system influent during periods of low

groundwater elevation. This additional well provided better hydraulic control within the slurry wall while reducing the intake of permanganate, which was causing fouling of the treatment process equipment. Extraction wells IW-1A, IW-2A, and IW3A remain in service, extracting a total of approximately 25 gallons per minute (gpm) in order to maintain inward gradients across the slurry wall and upward gradients between the overburden and bedrock inside the wall. Following treatment in the on-site facility, all extracted groundwater is recharged to the overburden via the recharge gallery located outside of the slurry wall.

The remedy implemented at OU2 consists of groundwater extraction and treatment with re-injection of treated water combined with discharge to the Souhegan River, monitored natural attenuation (MNA), and ICs. The treatment system was constructed between 2004 and 2005. It includes extraction of approximately 400 gpm from three extraction wells, recharge via three injection wells, and discharge to the Souhegan River. The treatment system has operated continuously since January 2006, except for brief shutdowns for regular operation and maintenance (O&M) activities (Gradient, 2013).

Volatile organic compound concentrations in shallow bedrock wells have historically been significantly lower than the concentrations detected in overburden wells. But as concentrations in the overburden decreased in response to the remedial efforts in both OU1 and OU2, a corresponding reduction in contaminant concentrations in bedrock, particularly in bedrock wells located outside the slurry wall, was not observed. In fact, VOC concentrations in some shallow bedrock monitoring wells have increased over time, even after implementation of the groundwater remedy within each OU at the Site. The observed increase in VOC concentrations in shallow bedrock at the Site and the lack of an adequate deep bedrock monitoring well network raised concerns that contamination from the Site could migrate through bedrock fractures to nearby residential wells. This concern prompted the bedrock investigations documented in the 2014 RI Report.

The 2014 RI consisted of four significant phases of investigation conducted between 2010 and 2013 to investigate and characterize the nature and extent of contamination in the bedrock aquifer. The additional bedrock investigations supplemented the existing shallow bedrock monitoring well network installed during the 1991 RI, evaluated data gaps identified during

previous investigations conducted in both OU1 and OU2, and provided the necessary data to support an updated FS. The activities described in the 2014 RI included:

- Installation or deepening of 18 bedrock monitoring wells in both OU1 and OU2
- Borehole geophysical logging to map the bedrock fabric
- Discrete interval (packer) sampling to assess vertical contaminant distribution
- 76-hour and 228-hour pumping tests to estimate aquifer properties
- Tracer dye study to assess contaminant migration pathways and rates

Results of the bedrock investigations indicate that contamination extends deep into the bedrock aquifer in portions of the Site within OU1, and to a lesser extent OU2. A CSM for the bedrock contamination developed based on data collected during the 2014 RI is provided in Section 3.

SECTION 2

SITE-SPECIFIC APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS

2. SITE-SPECIFIC APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS

As required under Section 121 of the CERCLA, 42 U.S.C. § 9621, remedial actions carried out under CERCLA must be protective of human health and the environment and attain the levels or standards of control for hazardous substances, pollutants, or contaminants specified by the ARARs unless waivers are obtained. Site-specific Remedial Action Objectives (RAOs) were included in the original ROD.

The only media of concern for OU1 is groundwater. The 1991 RI/FS evaluated overburden soil, overburden groundwater, shallow bedrock groundwater (typically less than 30 ft below the bedrock surface), surface water, and sediment, and concluded that site groundwater (overburden and shallow bedrock) was the only medium at the Site that presented unacceptable risks to human health. No risk to ecological receptors from any media was identified in the 1991 RI (HMM, 1991a). The ROD did not distinguish between the overburden and bedrock groundwater and identified the following RAOs:

- Prevent Ingestion of contaminated groundwater that would pose an unacceptable risk to human health.
- Restore groundwater quality to meet federal and state ARARs.

In order to attain the RAOs and comply with the ARARs, media-specific Remediation Goals (RG) were developed in the ROD. Criteria used to develop the candidate RGs included EPA MCLs, NHDES AGQS, and EPA risk-based standards for carcinogenic and non-carcinogenic health effects. The risk-based standards were based on an individual contaminant risk of lifetime cancer risk >EPA's acceptable range of 10^{-6} to 10^{-4} or a hazard index >1 to meet the remedial objective stated above. Both the cancer and noncancer risk-based standards were developed based on a residential use scenario. The ROD's chemical-specific ARARs that would be waived under a Technical Impracticability Waiver are:

Requirement	Status	Requirement Synopsis
Federal Requirements		
EPA Risk Reference Dose (RfDs)	To Be Considered	Dose levels developed by EPA to protect sensitive individuals over the course of a life-time The RfDs reflect a daily exposure level likely to be without appreciable risk of adverse health effects.
EPA Carcinogenicity Slope Factor	To Be Considered	Slope factors are developed by EPA from Health Effects Assessments and present the most up-to-date information on cancer risk potency. Slope factors are developed by EPA from Health Effects Assessments by the Carcinogenic Assessment Group.
Guidelines for Carcinogen Risk Assessment EPA/630/P-03/001F (March 2005)	To Be Considered	Guidance for assessing cancer risk.
Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens EPA/630/ R-03/003F (March 2005)	To Be Considered	Guidance of assessing cancer risks to children.
Safe Drinking Water Act (42 U.S.C. §300f *et seq.*); National primary drinking water regulations (40 C.F.R. 141, Subpart B and G)	Relevant and Appropriate	Establishes maximum contaminant levels (MCLs) for common organic and inorganic contaminants applicable to public drinking water supplies. Used as relevant and appropriate cleanup standards for aquifers and surface water bodies that are potential drinking water sources.
Safe Drinking Water Act (42 U.S.C. §300f *et seq.*); National primary drinking water regulations (40 C.F.R. 141, Subpart F)	Relevant and Appropriate for non-zero MCLGs only; MCLGs set as zero are To Be Considered	Establishes maximum contaminant level goals (MCLGs) for public water supplies. The MCLGs are health goals for drinking water sources. These unenforceable health goals are available for a number of organic and inorganic compounds.
Health Advisories (EPA Office of Drinking Water)	To Be Considered	Health Advisories are estimates of risk due to consumption of contaminated drinking water; they consider non-carcinogenic effects only. To be considered for contaminants in groundwater that may be used for drinking water where the standard is more conservative than either federal or state statutory or regulatory standards. The Health Advisory standard for manganese is 0.3 milligrams per liter.
State Requirements		
Drinking Water Quality Standards: NH Admin. Code Env-Dw 700	Relevant and Appropriate for MCLs and non-zero MCLGs only; MCLGs set as zero are To Be Considered	State MCLs and MCLGs establish maximum contaminant levels permitted in public water supplies and are the basis of State Ambient Groundwater Quality Standards (AGQS) that are applicable to site groundwater. The regulations are generally equivalent to the Federal Safe Drinking Water Act.
New Hampshire Ambient Groundwater Quality Standards (NH AGQS) (Env-Or 603.03, Table 600-1)	Relevant and Appropriate	Establishes maximum concentration levels for regulated contaminants in groundwater which result from human operations or activities. NH AGQS are equivalent to MCLs for contaminants that have MCLs. NH AGQS have been established for site groundwater contaminants for which no MCLs are established, and are derived to be protective for drinking water uses. The NH AGQS will be used for site contaminants where MCLs are not currently established.

Four VOCs have been detected in groundwater within OU1 at concentrations that exceeded their respective RG concentrations. The RGs established for the four chemicals are as follows:

Compound	Remediation Goal	Basis
1,1-dichloroethene (1,1-DCE)	7 micrograms per liter (µg/L)	MCL
cis-1,2-dichloroethene (cis-1,2-DCE)	70 µg/L	MCL
PCE	5 µg/L	MCL
TCE	5 µg/L	MCL

Under a TI Waiver, EPA would be acknowledging that due to the hydrogeologic and contaminant-related constraints to remediation within the proposed TI Zone within OU1, that remediation of groundwater contamination to achieve ARAR standards, represented by the RGs, is not technically practicable using any known remedial technologies. The TI Waiver would apply only to the four compounds listed above that are known to exceed the MCLs within the proposed TI Zone.

SECTION 3

CONCEPTUAL SITE MODEL

3. CONCEPTUAL SITE MODEL

3.1 SITE GEOLOGY AND HYDROGEOLOGY

The regional geological and hydrogeological setting has been described in detail in previous documents (HMM, 1991a; WESTON, 2014) and is briefly summarized as part of the CSM in this TI Evaluation Report. A more detailed discussion of the regional geology and hydrogeology is provided in the 1991 and 2014 RI Reports.

The Site is underlain by a sequence of unsaturated and saturated alluvium, glacial drift, and other unconsolidated deposits overlying the bedrock surface in the Souhegan River Valley. The overburden stratified glacial deposits that comprise the principal aquifer in the area, referred to as the Milford-Souhegan glacial drift (Harte, 1992), consist of a complex sequence of highly-permeable fine to coarse sands and gravels interbedded with finer sand layers. The aquifer ranges in thickness from approximately 50 ft to a maximum thickness of greater than 130 ft. The deepest portion of the aquifer is located in OU1 between the eastern half of the slurry wall and the boundary between OU1 and OU2 where a bedrock trough is present. The Souhegan River partially penetrates the sand and gravel alluvium aquifer and passes through OU1 directly north of the former OK Tool Source Area that is encompassed by the slurry wall. The river is a major source of recharge to the OU1 portion of the Site; however, farther downstream within OU2, the relationship between the river and overburden aquifer changes and the river becomes a discharge point for the unconsolidated aquifer.

The general direction of groundwater flow within the unconsolidated aquifer is from west to east. Prior to the implementation of the groundwater extraction remedies in both OU1 and OU2, the hydraulic gradient was relatively uniform (WESTON, 2011, and Gradient, 2011). The construction of the slurry wall and operation of the groundwater extraction system has had a particularly significant effect on both horizontal and vertical groundwater flow within the unconsolidated aquifer in OU1. Analysis of historical groundwater elevation data has indicated that the slurry wall and groundwater extraction system is effective in isolating groundwater within the slurry wall from the rest of the unconsolidated aquifer, but that overburden groundwater within the slurry wall area remains in hydraulic communication with the underlying

bedrock. The OU2 groundwater remediation system has had more localized effects on groundwater flow in downgradient portions of the Site.

Groundwater within the bedrock aquifer at the Site exists primarily within joints and fractures (secondary porosity). The primary porosity of crystalline bedrock is generally less than 0.01% but the secondary porosity can be much higher depending on how fractured the rock is. The majority of secondary porosity is composed of dead-end and micro-fractures. Larger, interconnected hydraulically-active fractures typically make up only a small percentage of the total bedrock porosity. Based on pumping tests conducted at the Site, the overall bedrock porosity is estimated to be 0.5%.

Borehole geophysical logging identified a moderate to steeply dipping fracture network with the majority of hydraulically active fractures dipping between 39 degrees (º) and 79º and striking to the northeast and north-northeast. A north to northeast trending fracture strike direction is typical of southern NH. Fractures were observed to be primarily dipping in a westerly direction with a lesser number of fractures dipping in an easterly direction. A review of borehole geophysical logs, including optical and acoustic televiewer results, indicates generally granitic and gneissic composition to the bedrock formation in the area. Foliation characteristic of the gneiss is easily visible in the logs for wells across the Site. Overall however, the bedrock is highly competent and fracture frequency generally decreases with depth. Most boreholes drilled deeper than 300 ft into the bedrock contained between 2 and 4 hydraulically-active fractures, most of which had very low yields (i.e., <1 gpm), and an average of 75 to 100 non-active (dead-end and micro-fractures) fractures per 100 ft. The depth of the boreholes (300 to 550 ft into bedrock) was based on the depth of nearby residential water supply wells.

Based on measured groundwater elevations in bedrock monitoring wells across the Site, the potentiometric surface suggests that groundwater flow is generally to the east, more or less parallel with flow in the unconsolidated aquifer, although there is an east-northeast component in areas north of the slurry wall. However, the primary fracture strike orientation trends to the north-northeast, which is consistent with the bedrock anisotropy depicted by the cone of depression during two pumping tests performed in support of the 2014 RI. Although the hydraulic head distribution suggests overall bedrock groundwater flow is generally to the east,

the direction of fracture strike and bedrock anisotropy facilitates movement of groundwater (and therefore contaminant migration), in a north-northeast direction as it migrates downgradient through the fracture network.

Results from the installation of monitoring wells and packer interval sampling indicate a highly competent bedrock fabric at the Site. Those observations are consistent with low calculated transmissivity values for nearly all bedrock fractures evaluated and by the very low total well yield (<0.25 gpm) exhibited by many of the open-hole bedrock wells. Findings from the RI pumping tests indicate that when a stress is applied to the bedrock aquifer, the primary source of recharge is from the overburden aquifer through a network of shallow, steeply dipping fractures. Within the bedrock aquifer, the stress accentuates a lateral connection through deeper fractures with approximately four times greater flow rates along strike versus perpendicular to strike. These results suggest a propensity for flow in the north-northeast direction within deeper zones of the bedrock aquifer, which substantiates the north-northeast trending anisotropy.

3.2 NATURE AND EXTENT OF CONTAMINATION

The 1991 RI identified four major industrial facilities where process waters and wastes were released to the ground surface or to surface waters flowing through the Site beginning in the 1940s and extending into the 1980s. The primary source of the contamination within OU1 was identified as the former OK Tool where a 1983 inspection of the facility identified potential releases of contaminants to floor drains, the ground surface, and the Souhegan River. For OU2, a discharge of manufacturing process water by the other settling parties to a discharge stream that flowed across Elm Street and into the Souhegan River north of the Savage Municipal Well was determined to be the primary source of contamination. The VOC compounds PCE, TCE, 1,1,1-TCA, 1,1-DCA, and trans-1,2-DCE are the primary contaminants for this Site. This section of the report will summarize the extent of contamination predominantly within OU1.

Processes used by the OK Tool to produce the metal cutting tools and tool hardware included machining, grinding, oxidizing, and heat treating. Those processes required the use of cutting fluids, lubricants, and cutting solvents. The primary cleaning solvent used by OK Tool was PCE. Waste from the manufacturing process included metal shavings, spent solvent, and sludge.

In 1983, a NHDES inspection of the OK Tool plant discovered the following:

- A vapor-degreasing tank was directly connected to a drain in the plant floor.
- Oily wastes and other materials were disposed of on the ground north of the plant.
- Cooling water used within the facility flowed into an open tank located near the vapor degreaser before it was ultimately discharged to the Souhegan River under a National Pollution Discharge Elimination System (NPDES) permit.

In 1983, NHDES ordered OK Tool to cease discharge of any waste and to begin an investigation to determine the extent of the soil and groundwater contamination. An unknown amount of contamination was released to the environment via the routes described above (HMM, 1991a).

3.2.1 Soil

Soil contamination consisting of VOCs was identified in the vicinity of the floor drain and in shallow soils north of the building. Concentrations of PCE as high as 1,150,000 micrograms per kilogram were measured in shallow soils. In 1985, soil remediation activities were completed inside the OK Tool building, including the removal and off-site disposal of approximately 27 cubic yards (cy) of soil from the vicinity of the floor drain. The floor drain excavation was approximately 9 ft by 9 ft in area and extended down to groundwater (approximately 10.5 ft below grade). The remaining shallow soil contamination associated with the northern surface spill area was treated via SVE between 1999 and 2008.

Residual soil contamination was discovered in the vicinity of a former leach field and a drainage pipe for the former OK Tool facility. Elevated concentrations of site contaminants in vadose zone soils associated with these two areas were thought to represent an ongoing source of shallow overburden groundwater contamination. Approximately 2,000 to 3,000 cy of shallow unsaturated soils exceeding the NHDES Soil Remediation Standards were excavated from the vicinity of the former leach field and stockpiled in late summer of 2008. In addition, approximately 59 cy of soil were removed from the vicinity of the drainage pipe. These soils were treated on-site during July 2009 using a combination of chemical oxidation (ozone and peroxide) and SVE. Confirmation samples were collected from the treated soils before they were returned to the excavation as backfill material. (EPA, 2011a)

No significant vadose zone soil contamination is believed to remain within OU1.

3.2.2 Overburden Groundwater

Four principal VOC contaminants have been detected in groundwater within the unconsolidated overburden aquifer: PCE, 1,1-DCE, TCE, and cis-1,2-DCE. Of these, PCE is the most prevalent and is found at the highest concentrations. Figure 3-1 shows the extent of the PCE plume in overburden groundwater as mapped in 1990. Installation of the slurry wall and operation of the OU1 groundwater extraction system since 1999 has significantly reduced VOC concentrations in the overburden groundwater outside the slurry wall in OU1. Figure 3-2 presents PCE concentration trends for selected monitoring wells located outside the slurry wall within OU1 that illustrate the reduction, and supported shut down of the extraction wells outside the slurry wall in April 2007 (WESTON, 2011). Figure 3-3 shows the extent of PCE in shallow overburden groundwater within OU1 as mapped in 2013. It is evident from Figure 3-3 that the overburden plume has been effectively cut off from the source by the slurry wall and the OU1 groundwater extraction system.

In general, the remedy for OU1 has resulted in significant reductions of contaminants in the overburden outside the slurry wall within OU1 with an observed 99% decrease in concentrations since implementation of the remedy (EPA, 2011a). However, significant PCE contamination remains in overburden groundwater within the slurry wall, particularly in the mid-level and deeper portions of the aquifer where concentrations as high as 19,000 µg/L have been detected as recently as October 2014. Figure 3-4 shows the extent of PCE contamination in the deep portion of the overburden within OU1.

3.2.3 Shallow Bedrock Groundwater

As in the overburden aquifer, the primary contaminant in the shallow bedrock is PCE, although the other VOCs discussed above are also present, but at lower concentrations. In contrast to the overburden however, VOC concentrations in the shallow bedrock, both within and beyond the confines of the slurry wall, have not been reduced since implementation of the remedies at both OU1 and OU2. The PCE concentrations in several shallow bedrock monitoring wells located in OU1 (MW-16R, PW-2R, and MW-2R) have generally increased since the construction of the slurry wall, while others (MW-14R, PW-5R and PW-6R) have fluctuated significantly and exhibited no distinct trend. Figure 3-5 presents PCE concentration trends for selected shallow bedrock monitoring wells within OU1 and OU2 that illustrate the trend variability. Measured

concentrations of PCE in the shallow bedrock are generally less than 1,000 µg/L, which is substantially lower than the concentrations observed in overburden during the 1991 RI. This explains why the bedrock investigation was not expanded to include the deep bedrock during the 1991 RI.

3.2.4 Deep Bedrock Groundwater

The recent investigations into the deep bedrock aquifer beneath OU1 identified higher concentrations of PCE than those currently present in the overburden or shallow bedrock aquifers. Deep bedrock well BR-6 was drilled immediately downgradient from known release areas within OU1. Fractures in that well that were determined to be hydraulically-active exhibited the highest concentrations of PCE and TCE observed in bedrock at depths up to 400 ft into bedrock (500 ft below grade). The concentration of PCE in BR-6 ranges from 59,000 to 100,000 µg/L, and the concentration of TCE ranges from 1,900 to 6,300 µg/L. The PCE and TCE concentrations were observed to increase with depth. Figure 3-6 shows the distribution of PCE in deep bedrock within OU1. Figure 3-7 shows the distribution of PCE in deep bedrock in all areas of the Site. Figure 3-8 is a geologic cross-section illustrating the vertical distribution of PCE in bedrock.

The increase in VOC concentration with depth observed at BR-6 suggests that there was a significant vertical component to PCE release at OK Tool and that the full vertical extent of impact to the bedrock aquifer has not been determined. The large vertical distribution of PCE is believed to be the result of downward DNAPL migration from the source area into the deep bedrock via steeply-dipping interconnected (hydraulically-active) fractures. Laterally, the concentrations of site COCs decreased by approximately one order of magnitude at wells BR-2 and BR-3 relative to BR-6. Wells BR-2 and BR-3 are located approximately 300 ft to the east and north-northeast of BR-6, respectively. They are inferred to be directly downgradient of BR-6 based on the eastern hydraulic gradients (BR-2) and along strike based on geophysical data and the observed aquifer anisotropy (BR-3). When compared to the vertical distribution at BR-6 (very little variation in concentration over 400 ft in bedrock), the large lateral decline in contaminant concentrations (nearly three orders of magnitude reduction over 500 ft between BR-6 and BR-1) illustrates the dominant vertical component of contaminant migration at the site (see Figure 3-8).

To the north-northeast of well BR-3, the concentrations of site contaminants decrease by two orders of magnitude within approximately 300 ft. The north-northeast direction from BR-3 is cross-gradient based on the distribution of hydraulic head across the bedrock aquifer. However, monitoring wells BR-11, MW-2R, and MW-30, which are north-northeast from BR-3, appear to be hydraulically connected to the bedrock aquifer beneath the former OK Tool property via the primary fracture strike orientation, as identified during the 2010 and 2013 bedrock pumping tests. Because BR-11, MW-2R, and MW-30 are located in an area where the overburden aquifer is not impacted by site contamination, it is inferred that the contamination in these wells migrated laterally through bedrock fractures rather than downward from the overburden. The impact to these wells is inferred to be either the result of dissolved phase contamination being diverted along strike while attempting to follow the eastern hydraulic gradients within the aquifer, or the result of former hydraulic stresses that altered the historical direction of bedrock groundwater flow.

No COCs were identified in monitoring wells located to the north of the former OK Tool area near several residential properties that obtain drinking water from individual bedrock supply wells. In addition, no COCs have been identified at any of the residential properties included in the drinking water monitoring program. The absence of impacts to the bedrock monitoring wells and the residential drinking water wells, the easterly hydraulic gradients, and the north-northeast fracture strike orientation and aquifer anisotropy, all suggest that the residential areas to the north and northwest are hydraulically upgradient. It is unlikely that those wells will become impacted with site-related contamination under the current hydrogeologic conditions. However, it is important to note that a change in the current hydrogeologic conditions, such as additional pumping from residential development on the property directly to the north of the former OK Tool area and Souhegan River, could alter the direction of hydraulic gradients to this area and thus result in the migration of impacted groundwater to areas located to the north and northwest.

East of BR-2, concentrations of site COCs in the bedrock aquifer decrease by another order of magnitude in monitoring wells near the OU1/OU2 boundary. In addition, concentrations of PCE generally decrease with depth in the areas east of the former OK Tool area. That decrease is different than contaminant trends in the bedrock wells located closer to areas where releases are inferred to have occurred. Historically, elevated concentrations of PCE were present in the deep

overburden throughout areas to the east of the former OK Tool area. The shallow bedrock contamination in those areas is believed to have been at least partially the result of downward migration of dissolved phase contamination from the overburden aquifer. Although contaminant concentrations decrease with depth in each of the bedrock wells present in those areas, PCE was still observed at concentrations ranging between 190 and 300 µg/L in the deepest hydraulically active fractures encountered in the wells located near the OU1/OU2 boundary. Because of the fact that potentiometric head distribution indicates that those wells are positioned hydraulically downgradient from the former OK Tool property, it is possible that concentrations at depth in the wells located near the OU1/OU2 boundary are the result of both the vertical migration of dissolved phase contamination from the overburden and the lateral migration of contamination through a matrix of deep hydraulically active fractures. The reduction of concentrations by another order of magnitude with respect to those observed at BR-6 further corroborates the conclusion that the competent bedrock fabric has limited the lateral migration of impacted groundwater at the Site.

Well BR-16 is located along the OU1/OU2 boundary immediately north of Elm Street. Groundwater quality data from that well provides an indication of the southern extent of impact to the deep bedrock aquifer with PCE concentrations ranging between 13 and 8.5 µg/L. Those values are more than an order of magnitude less than those observed in well MW-16R, which is located approximately 300 ft to the north. That data support the conclusion that the easterly hydraulic gradients and north-northeast anisotropy inhibited contaminant migration to the south. In conjunction with the data from well BR-9 located immediately to the south of the slurry wall, those results indicate that there is likely to be minimal, if any, impact to the bedrock aquifer south of Elm Street.

The VOC concentrations in deep bedrock in OU2 are substantially less than those observed in OU1. Groundwater quality data collected in 2012 for the deep bedrock in OU2 showed concentrations of PCE ranging between 56 and 7.7 µg/L, or approximately one order of magnitude less than wells along the OU1/OU2 boundary. In contrast to OU1, the concentrations of VOCs in bedrock wells in OU2 decrease with depth. This incongruence suggests that the bedrock contamination in OU2 may be at least partially the result of downward migration of

dissolved contamination from the overburden aquifer, rather than solely from horizontal migration through fractures in the deep bedrock. This is particularly true for the shallow bedrock.

The distribution of bedrock contamination across the entire Site can be directly attributed to the steeply dipping fracture network, the physical properties of the contaminants, and the two primary lateral migration pathways. The steeply dipping fracture network and the physical properties of the contaminants have facilitated the significant vertical migration of contamination from the areas where releases occurred to depths greater than 500 ft bgs, and also have facilitated the migration of impacted overburden groundwater into the bedrock aquifer in downgradient portions of the plume to the east. The north-northeast anisotropy of the bedrock aquifer and the overriding easterly hydraulic gradients are each clearly evident based on contaminant distribution; the contaminant plume extends in each direction away from the former OK Tool property.

Although there are a limited number of hydraulically-active fractures in the bedrock, the combination of the expansive overburden plume, downward gradients between the overburden and the bedrock, and the physical properties of the COCs, has resulted in the distribution of low level concentrations of COCs throughout the bedrock aquifer beneath the Site. Higher concentrations are typically observed in the shallow bedrock fractures in the downgradient areas of the Site and are inferred to be the result of the downward migration of contamination from the overburden aquifer because of the fact that the overburden aquifer has historically exhibited higher concentrations of COCs.

Although the extent of contamination in the deeper downgradient portions of the bedrock aquifer is limited as compared to the shallow bedrock and the overburden aquifers, it may be attributable to both lateral migration through the bedrock aquifer as well as downward migration from the overburden aquifer and shallow bedrock fractures that intercept the bedrock surface. The similarity of the areal extent of both the overburden and shallow bedrock plumes, and the minimal impact to the deep bedrock aquifer in the downgradient areas of OU2 is further evidence of the downward migration of impacted groundwater from the overburden into the bedrock. It is important to note that because of the competent bedrock, the actual number of hydraulically active fractures is limited; therefore, while the area of impact is large and

comparable to the overburden plume, the actual volume of impacted water, and the mass of contaminant, is relatively small when compared to that of the overburden aquifer.

3.3 CONTAMINANT FATE AND TRANSPORT

The 1991 and 2014 RIs evaluated the environmental fate and transport characteristics of the COCs at the Site. Fate and transport address the mobility, stability, volatility, and persistence of each substance. Mobility and persistence represent the potential for a chemical to migrate along a given pathway and how long a chemical will remain in the environment, and are therefore of primary importance.

Compounds identified in overburden and bedrock groundwater above the ROD ICLs in OU-1 include PCE, TCE, cis-1,2-DCE, and 1,1-DCE. Tetrachloroethene and to a lesser extent TCE are the most widely detected compounds in groundwater. The other two compounds are limited in extent and fall well within the boundary of the PCE plume. The physical properties of TCE, cis-1,2-DCE, and 1,1-DCE are similar in nature to PCE and they would behave in a similar fashion with regard to their fate and transport within groundwater. Therefore, any remedial action implemented to address PCE-impacted groundwater is inferred to result in a concomitant reduction of TCE, cis-1,2-DCE, and 1,1-DCE concentrations.

The compound 1,4-dioxane is an emerging contaminant that was not identified as a COC at the time the original RI/FS was conducted. It was commonly used with 1,1,1-TCA as a stabilizer and corrosion inhibitor (EPA, 2009b). No ICL for 1,4-dioxane was included in the ROD, but the current NHDES AGQS for that compound is 3 µg/L. Because 1,1,1-TCA was included as a COC for the Site, groundwater in OU1 and OU2 has been periodically tested for 1,4-dioxane during selected monitoring rounds beginning in 2003. To date, 1,4-dioxane has not been found in groundwater samples collected from either overburden or bedrock monitoring wells located within OU1, although only two sampling rounds (May and October 2009) had detection levels that were below the AGQS of 3 µg/L. Other OU1 sampling rounds in 2003 and 2010 had elevated detection levels ranging from 50 to as high as 12,500 µg/L in well PW-06D. In contrast to the lack of 1,4-dioxane detections in OU-1, 1,4-dioxane was detected in OU2 groundwater monitoring rounds conducted in 2003, 2009, and 2010 at concentrations as high as 7.54 µg/L.

Based on observations during drilling and groundwater monitoring activities at the Site, no non-aqueous phase liquid has been directly identified in soils or on bedrock fracture surfaces to date. The highest concentration of PCE in groundwater was identified in the bedrock monitoring well BR-6, at levels of 100 milligrams per liter (mg/L). This represents approximately 67% of the solubility of PCE in water, which is 150 mg/L at standard temperature and pressure (Montgomery and Welkom, 1990), and indicates a high potential for free-phase PCE to exist. Free-phase chlorinated ethenes such as PCE and TCE are often referred to as DNAPL, meaning that they will tend to sink within the saturated zone (because their specific gravity is greater than 1.0, the specific gravity of water) until they encounter a low-permeability layer in the formation at which point the DNAPL will flow laterally along the surface of the low-permeability formation and may diffuse into it.

The DNAPL constituents (which, if present, are likely primarily PCE at this Site) will slowly dissolve into groundwater and migrate as part of the groundwater flow system, thereby creating a dissolved contaminant plume. Once the dissolved-phase contaminants have entered the groundwater flow system, they will be carried in the direction of groundwater flow by advection. Mechanical dispersion will tend to spread the plume perpendicular to the groundwater flow direction in the unconsolidated aquifer but will play a lesser role in the downgradient migration of contaminants through the bedrock because of the limited mixing that occurs in flow through fractures. Some dilution of the contaminants will occur as a result of clean water from precipitation that enters the overburden aquifer from above and as the fractures containing contaminants receive water from fractures with clean water in bedrock. Molecular diffusion will tend to move contaminants into low-permeability layers in the overburden aquifer and smaller fractures transverse to the primary direction of contaminant movement in bedrock, many of which may be dead-end fractures. This is important because reverse diffusion from low-permeability layers in the overburden and dead-end/micro fractures in bedrock must be considered when evaluating potential remedial measures for the Site as they will tend to hold residual contamination long after the primary pathways have been cleaned up. In general, advection and dispersion are inferred to play the greatest roles in the transport of impacted groundwater at the Site. The distribution of the contaminant plume can be directly attributed to the effects from these two mechanisms.

3.4 CONTAMINANT MIGRATION PATHWAYS/ASSESSMENT OF SOURCE AREAS

Chlorinated VOCs are the principal concern at the Site in terms of potential for migration off-site and potential exposures that could affect human health and the environment. Although groundwater, surface water, soil, and air are all potential exposure pathways, groundwater is considered the only complete migration pathway in OU1. The other migration pathways were each evaluated in the 1991 RI for the Site, and the ROD included a determination that no risk was posed by pathways other than groundwater.

The configuration of the bedrock surface appears to have had a significant influence on the migration of contamination between the overburden and bedrock. The highest concentrations of PCE detected in the bedrock aquifer during the 2014 RI were in BR-6, a well installed downgradient from known overburden sources near the base of a slope in the bedrock surface within the OU1 slurry wall. It is likely that when DNAPL was released at the ground surface, it migrated down through the overburden under the force of both gravity and the hydraulic gradient, possibly encountering lower-permeability layers in the overburden and traveling laterally along them until able to resume downward migration. This process would continue until the DNAPL eventually reached the till/bedrock surface. The DNAPL then flowed eastward down along the top of the sloped till/bedrock surface, collecting in low spots. The variable thickness and discontinuous nature of the till at the Site and the nature of the bedrock surface facilitated infiltration of DNAPL into shallow bedrock fractures.

Upon infiltrating the bedrock surface through gaps in the till and via fractures, DNAPL would have continued to flow vertically downward under its own positive head (overcoming pore entry pressures) and hydraulic gradients through the steeply dipping, interconnected fracture network until ultimately dissolving in groundwater or collecting in small troughs in primary fractures or dead end fractures. Dissolved phase contamination would have been carried via groundwater flow in a tortuous “zig-zag” pattern through the bedrock fracture network as it followed the hydraulic gradients and flow along the bedrock strike. The natural hydraulic flow of groundwater has resulted in the migration of dissolved phase PCE and TCE away from the former OK Tool property through bedrock fractures. However, concentrations of dissolved phase PCE and TCE in the bedrock decrease by two to three orders of magnitude within approximately 500 ft laterally

beyond the OU1 slurry wall. The large decline over such a short distance suggests that lateral migration is curtailed in the competent bedrock, unlike the overlying unconsolidated aquifer where expansive lateral migration was observed and the plume extends over 5,000 ft.

Groundwater flow in bedrock at the Site is governed by two factors: 1) hydraulic gradients, which dictate bedrock groundwater flow must be to the east and east-northeast; and 2) the north-northeast orientation of the fractures in the bedrock which limit the direction groundwater can physically move. As a result of those two influences, as impacted groundwater attempts to follow the easterly hydraulic gradient, the orientation of the bedrock fractures tends to push the groundwater off to the north-northeast. The north-northeast oriented fractures are one of the primary lateral contaminant migration pathways in the bedrock.

The majority of fractures identified during the investigations conducted in support of the RI was moderate to steeply dipping and had a primary dip direction to the northwest with a lesser number of fractures dipping to the southeast. In general, the distribution of the heavily-contaminated groundwater within OU1 at the Site is primarily the result of the vertical migration of DNAPL flowing vertically downward under the force of gravity. That is evidenced by the highest contaminant concentrations present almost directly beneath known release areas at the former OK Tool property.

Current hydrogeologic conditions in the bedrock aquifer and historical contaminant levels in the overburden explain the migration of dissolved phase contamination to the east. The north-northeast orientation of the bedrock anisotropy explains migration of contaminants from the former OK Tool property toward the north and northeast. However, the steeply dipping nature and low transmissivity of most hydraulically active fractures and the relatively low horizontal hydraulic gradient have limited the lateral migration of grossly-impacted groundwater in bedrock away from the area directly beneath the former OK Tool area, when compared to the expansive lateral migration of contamination to the east that has occurred in the overburden aquifer.

3.5 CONCEPTUAL SITE MODEL SUMMARY

A CSM was developed for the bedrock aquifer to form the basis for the evaluation of potential remedial measures. The conceptual model was developed to represent how the hydrological cycle interacts with the local geology, describing the presumed migration of water through the

system, and its corresponding effects on the migration of contamination based on contaminant transport mechanisms, migration rates, and degradation pathways. The CSM explains the primary contaminant migration pathways and provides insight into the observed distribution of contaminants in site media. The CSM for the Site is depicted graphically on Figure 3-9.

Groundwater at the Site originates as precipitation falling onto the ground surface, collecting in river, streams, and impoundments such as the Souhegan River, and eventually infiltrating into the subsurface. Extensive groundwater elevation and river stage monitoring indicate a highly coupled flow system between the Souhegan River, the overburden aquifer, and the bedrock aquifer, as evidenced by a rapid coordinated response to precipitation events (Harte, 2006).

Groundwater flows from areas of higher hydraulic head to areas of lower hydraulic head. That results in flow to the east within the overburden aquifer and to the east-northeast within the bedrock aquifer. In addition to moving horizontally, groundwater will also move vertically based on hydraulic head differences between subsurface materials and between the overburden and bedrock aquifers. The construction of the slurry wall and the groundwater extraction and treatment system within OU1 has curtailed horizontal overburden flow within the slurry wall and has had a particularly significant effect on the interaction between the overburden and bedrock aquifers in the proximity of the wall. Unless removed from the system by pumping, surface recharge of precipitation will increase the hydraulic head inside the wall, resulting in downward hydraulic gradients that force overburden groundwater into the bedrock aquifer by way of shallow fractures that intercept the bedrock surface. Outside the slurry wall, vertical gradients are primarily static within the overburden aquifer suggesting near-horizontal flow (EPA, 2011a).

Groundwater movement in bedrock is restricted to open fractures. Therefore, groundwater follows those fractures as it moves downgradient in response to regional hydraulic head distributions. Groundwater movement in the bedrock aquifer at the Site is primarily controlled by two factors: 1) the east-northeast influence of hydraulic head differences and 2) the north-northeast anisotropy of the bedrock. The dynamics of those two forces strongly manipulate the distribution of contamination that is observed throughout the Site.

At the OU1 portion of the Site, contaminants were historically released to the ground surface, the former leach field, subsurface discharge features such as floor drains and drainage pipes, and to surface water between the 1940s and 1980s.

Once released to the ground at the former OK Tool area, DNAPL migrated vertically downward under gravity until eventually reaching the till/bedrock surface. The DNAPL that reached the till/bedrock surface interface then likely flowed downward along the top of the sloped surface of these units and collected in low spots. The variable thickness of the till and the nature of the bedrock surface facilitated infiltration of DNAPL into shallow bedrock fractures. Upon infiltrating the fractures at the bedrock surface, DNAPL continued to flow downward under gravity through the steeply dipping, interconnected fracture network in the bedrock formation until ultimately dissolving fully into groundwater or collecting in small troughs and dead end fractures. Residual DNAPL that likely remains in the bedrock fractures continues to dissolve into groundwater flowing through the bedrock fractures.

Dissolved-phase contamination is transported in groundwater as it flows through the overburden and the bedrock fracture network. The direction of bedrock groundwater flow is impacted by the groundwater gradients and by the orientation of fracture strike. This combination of hydraulic gradient and physical characteristics of the bedrock has resulted in some dissolved-phase contamination migrating in an easterly, downgradient direction, and some being directed to the north-northeast along the fracture strike. The fact that the hydraulic gradients and bedrock anisotropy are nearly perpendicular, combined with the very low fracture density and steep dip of the few fractures that do exist, has tended to severely restrict lateral migration of dissolved contamination in the deep bedrock. The steeply dipping fracture network combined with the higher density of DNAPL resulted in the significant downward migration of contamination within the bedrock near the source area. Overburden groundwater flow is driven by hydraulic gradients and the distribution of lower-permeability zones and as a result generally follows a much less tortuous path as it migrates downgradient.

The local hydraulic head distributions in fractured bedrock systems can be significantly altered by the withdrawal of groundwater such as the operation of water supply wells. In addition to altering the local hydraulic head distributions within the bedrock, pumping of bedrock wells can

increase the groundwater velocities through nearby fractures. A stress applied to the bedrock aquifer results in groundwater flowing from the overburden to the bedrock via the shallow steeply dipping fractures present at the Site. The stressed conditions also increase groundwater flow velocities dramatically, specifically along the orientation of fracture strike. In general, the data collected in support of the 2014 RI indicate that when a stress is placed upon the bedrock aquifer, the flow component in the north-northeast direction is greatly accentuated, and depending on the magnitude of the stress, becomes a significant contaminant migration pathway.

As groundwater flows to the east through the overburden, the vertical hydraulic gradient is impacted by precipitation. As a result, there is a propensity for downward contaminant migration of the overburden groundwater into deeper portions of the overburden aquifer and into the bedrock aquifer where shallow fractures intercept the bedrock surface. Downward migration of impacted groundwater from the overburden into the bedrock aquifer is believed to have played a significant role in influencing the distribution of contamination in the shallow bedrock downgradient of OU1. The configuration of the shallow bedrock PCE/TCE plume mimics the historical distribution of contamination in the deep overburden across both OU1 and OU2, and concentrations of these contaminants in the bedrock aquifer decrease with depth in areas east of OU1. Currently, operation of the OU2 groundwater extraction system intercepts the easterly migration of the overburden plume.

SECTION 4

EVALUATION OF RESTORATION POTENTIAL

4. EVALUATION OF RESTORATION POTENTIAL

In order to assess the potential of restoring the impacted groundwater to beneficial use (i.e. meeting the RGs), an estimate of the remaining contaminant mass was developed, as well as a basic understanding of its disposition in the subsurface. Once the baseline understanding has been established, restoration potential is best evaluated through performance evaluation of past remedial actions, when available and relevant.

There is a large amount of information regarding the effectiveness of remedial activities performed at OU1 (soil and overburden groundwater) including the installation of a slurry wall, permanganate injections to treat overburden groundwater, and a groundwater extraction and treatment system that has been operating since 1999. However, there have been no specific attempts to directly remediate the bedrock aquifer within OU1, although operation of the OU1 treatment system was designed to induce upward gradients inside the slurry wall to prevent further migration of contaminants into the underlying bedrock.

The evaluation of restoration potential for the groundwater within OU1 will rely on performance data from the existing remedial systems, current industry experience with respect to expected performance of remedial technologies on chlorinated solvents released to complex hydrogeologic settings such as those found in OU1 (DNAPL released to fractured bedrock), the ability (inability) to fully characterize the nature and extent of DNAPL present in the bedrock aquifer.

4.1 ASSESSMENT OF MASS RELEASED

A number of factors limit the accuracy of estimating the remaining mass of PCE in the bedrock. Records provided by OK Tool and documented in the 1991 RI Report suggest that at least 9,338 gallons of PCE (approximately equivalent to 126,000 lbs) were purchased between 1977 and 1985 (HMM 1991a). No records are available for the time period between when the facility first began operation in the 1940s and 1977 and as a result, the volume of chemicals used during the 35+ years prior to 1977 is unknown. In addition, no records for the volume of TCE purchased are available, so it is uncertain what percentage of TCE present is associated with the original discharge versus that created from the natural biodegradation of PCE.

The ultimate fate of the total volume of PCE purchased is unknown because insufficient data are available, particularly with respect to how much DNAPL was released into the ground versus what may have been properly disposed off-site. The fate of the total PCE purchased by the facility can be described generally as: disposed off-site, volatilized to the air during use, discharged to the Souhegan River along with process water, and discharged to the ground surface via spills, floor drains, and the septic system. No records of off-site disposal of spent PCE were discovered while conducting the 1991 RI (HMM, 1991a).

A portion of the PCE used at the facility may have been released directly to the Souhegan River during the years when OK Tool held a NPDES permit that allowed discharge of industrial wastewater; however, accurately estimating that quantity is not possible because neither the volume of the discharge nor the PCE concentration in the discharge are available. The mass of PCE released to the river was likely negligible given the very low discharge limit (copies of NPDES permits provided in 1991 RI that show a discharge limit of 9 parts per billion PCE). The total mass of PCE released directly to the river is likely to be no more than tens of lbs.

A portion of the PCE used at the facility would likely have volatilized during use. While there is no accurate way to estimate this volume, it is again likely to have been on the order of tens of lbs of PCE.

Therefore, the remaining volume is assumed to have been discharged to the ground, incidentally or intentionally. Without any reliable method to estimate the volume of PCE actually discharged into the ground, a range of volumes was developed. Ignoring any pre-1977 releases and assuming between 25% and 75% was actually discharged to the ground and entered groundwater either as dissolved phase or DNAPL (to account for the possibility of significant volatilization loss or undocumented off-site disposal), a mass of approximately 32,000 to 95,000 lbs is estimated.

4.2 MASS REMOVAL BY PREVIOUS REMEDIAL ACTIONS

Remedial actions designed to treat VOCs present in soil and groundwater at the Site have been ongoing since 1984. These remedial actions have focused largely on overburden soil and groundwater, although some may have had a minor impact on the shallow bedrock groundwater as well.

Remedial actions performed to date that included some mass removal/destruction include those listed below. Full coverage of remedial actions and the associated details can be found in the latest Five-Year Review Report (EPA, 2011a) and the series of Annual Reports for OU-1 available for this Site:

- Past Remedial Actions for Mass Removal Within OU-1:
  - Removal of 27 cy of highly-impacted soil from the vicinity of the OK Tool facility in 1984.
  - Installation of SVE/AS system in the OK Tool source area in May 1998 to treat groundwater via *in situ* air stripping and carbon adsorption. The SVE/AS system was operated intermittently until 2004 when the AS component was dismantled. The SVE system was operated intermittently from 2006 to 2008. The SVE system was permanently removed from operation in 2008.
  - Construction of slurry wall in October 1998 to prevent contamination migration in the overburden from the source area to the downgradient plume.
  - Groundwater extraction and treatment for the overburden groundwater inside the slurry wall to maintain hydraulic control and facilitate contaminant mass removal. Full operation of the system commenced in May 1999 and is still ongoing.
  - Excavation and ex situ oxidation of approximately 3,000 cy of soil from the vicinity of former leach field and PW-22 in 2009 to prevent continued leaching of contamination from the vadose zone. The treated soil was replaced in the original excavation as backfill.
  - An ISCO treatment to reduce contaminant mass, accelerate groundwater cleanup inside the slurry wall, and/or prevent impacted groundwater from entering the shallow bedrock aquifer. A pilot study was performed in 2003, and full-scale injections were performed in 2004, 2008, 2009, and 2010.

The treatment technologies listed above have removed or destroyed only a small portion of the PCE released into the environment. Remediation of overburden groundwater has been ongoing with AS and extraction and treatment for over 15 years. The SVE system removed approximately 800 lbs of VOCs between 1999 and 2007. An estimated 3,000 lbs of VOCs have been removed via the OU1 groundwater extraction and treatment system since 1999. Since 2009, the system has been removing approximately 50 lbs/year of total VOCs based on a pumping rate of roughly 25 gpm. Because only total VOC mass removal is typically provided in the annual reports for treatment plant operation, the PCE mass removal was estimated. A review of data in

the 2013 Annual Monitoring Report suggested that approximately 90% of the total VOC mass was attributable to PCE. Therefore, it is estimated that approximately 2,700 lbs of PCE have been removed by the OU1 treatment plant, with roughly 45 lbs of PCE currently being removed annually. Table 4-1 shows the annual mass removal by the OU1 groundwater extraction system.

Additionally, approximately 2,500 to 3,500 cy of soil containing PCE was excavated and disposed off-site, or treated on-site via chemical oxidation and replaced. These soil treatments have removed an additional 20 lbs of PCE, assuming an average concentration of 2 milligrams per kilogram.

The mass of PCE destroyed via ISCO treatments was approximated by estimating the volume of groundwater within the ISCO treatment zone (assumed the lower third of the area within the slurry wall) and applying the average reduction in PCE concentration that resulted from the treatment. Based on this, the total estimated mass destroyed via ISCO is approximately 125 lbs.

Based on the above estimates, less than 3,000 lbs of PCE have been removed from the Site through remedial actions over the last 28 years, representing less than 10% of the lower end of the range of the total mass potentially available for release. Subtracting the mass removed from the total estimated mass potentially released (see above) suggests that the potential remaining mass of PCE present in the overburden and bedrock within OU1 (as sorbed phase, dissolved phase, mobile DNAPL, and residual DNAPL), could be as high as 29,000 lbs to 92,000 lbs. This analysis demonstrates the limited success of the current remedy and its inability to attain RGs in a reasonable amount of time, if ever.

4.3 FACTORS LIMITING REMEDIATION

There are several factors specifically related to this Site that would limit the effectiveness of remedial technologies in achieving groundwater cleanup standards within the proposed TI Zone or at a minimum, require that remediation be conducted for an extended period of time[2]. Factors

[2] Remediation may still be warranted to reduce the mass of contamination, particularly the DNAPL, inside of the proposed TI Waiver Zone and to manage the migration of contaminated groundwater beyond the proposed TI Waiver Zone boundary.

that affect the groundwater restoration potential for this Site include hydrogeologic and contaminant-related factors. These factors include the following:

1. **The presence of DNAPL in fractured bedrock**: One of the most significant factors within the proposed TI Zone that severely limits the ability to restore groundwater to drinking water standards is that the extremely high observed concentrations in deep bedrock strongly suggests that DNAPL is present in fractured bedrock that underlies the Site. The presence of large amounts of chlorinated VOCs, such as PCE, in both non-aqueous and dissolved phases in fractured bedrock has been recognized by scientists, engineers, and regulatory agencies as the most difficult situation to remediate [National Research Council (NRC), 1994, Table 7-1; EPA, 1993, Figure 1]. There are several reasons for this conclusion. Fractured bedrock is typically much more heterogeneous than unconsolidated porous media and the fractures create extreme variability in permeability over very short lateral distances. In addition, the contaminant migration pathways in bedrock fractures are much less predictable than in porous media because they are controlled by fracture distribution and orientation. As a result, the pathways tend to be highly tortuous routes between two points rather than a straight line. This makes mapping of the pathways nearly impossible using the standard monitoring well approach. To further complicate matters at this Site, the extremely high concentrations observed at several deep bedrock wells (BR-2, BR-3, BR-5, and BR-6) suggest that DNAPL is likely present within fractures. The solubility limit for PCE is 150 mg/L. Dissolved concentrations as high as 100 mg/L have been measured in bedrock fractures at depths of 500 ft below grade. Current industry practice is that dissolved concentrations higher than 1% of the solubility limit are suggestive of potential DNAPL and that concentrations greater than 10% indicate that DNAPL is likely (probable DNAPL). The observed concentrations of PCE in deep bedrock at this Site are more than 65% of the solubility limit, meaning that the presence of DNAPL in the deep bedrock is nearly assured. Based on the estimated volume of PCE potentially spilled at the Site and the limited amount recovered, it is possible that a large volume of DNAPL could be present in the deep bedrock. If present, the DNAPL would be trapped within dead-end or micro fractures or at the bottom of hydraulically-active fractures where they pinch out at depth.

2. **The potential for slow back-diffusion of groundwater contaminants into the bedrock matrix and inaccessible fractures due to low matrix porosity and hydraulic conductivity**: Dissolved-phase contamination located in this type of setting (fractured bedrock of low permeability) can be hydraulically inaccessible and, therefore, not readily removed, causing a long-term source of groundwater contamination through slow back diffusion of the contaminant into the groundwater. For example, DNAPL in lower permeability overburden formations and bedrock fractures will likely be hydraulically inaccessible to water for groundwater extraction and treatment. While injected oxidants, microbes, nutrients, and other *in situ* remediation amendments will eventually diffuse into these dead-end and micro fractures, it is an extremely slow process that requires maintaining a high concentration of the amendment in the primary hydraulically-active fractures for a very long time. This can be nearly impossible because the high rate of groundwater

flow through the active fractures tends to quickly flush out the amendment. Back-diffusion of contamination from the rock matrix, dead-end fractures and micro-fractures will significantly increase the time it takes to remediate groundwater to drinking water levels, which is often measured in terms of 100s or even 1,000s of years (Lipson et al, 2005; Chapman et. al., 2005).

3. **The difficulty of characterizing DNAPL location and extent**: In order to effectively remediate the Site, it would be necessary to confirm the horizontal and vertical extent of the DNAPL zone. However, characterization of a DNAPL zone is very difficult, costly, and typically inconclusive in a heterogeneous environment such as fractured crystalline bedrock. Investigations are limited to drilled wells (more cost-effective direct push drilling methods cannot be applied to bedrock) that are very expensive due to the great thickness of the overburden at this Site (50 to 110 ft) and its highly conductive nature. This type of drilling and the extreme depth of the contamination (exceeds 600 ft) requires highly-specialized equipment. Further, information on the bedrock structure that may be gained from each well is highly limited because of the severe heterogeneity of the rock, requiring a very high density of wells. As a result, adequate characterization of DNAPL zones within bedrock for the purpose of aquifer restoration is highly problematic and may not be attainable. Cost-effective site characterization is difficult, owing to the difficulty in determining the location of subsurface fractures and predicting the transport of contaminants through fracture systems. These characterization problems make it difficult to design monitoring and cleanup strategies for complete restoration. (Geosyntec, 2004).

In summary, these hydrogeologic and contaminant characteristics within the proposed TI Zone combine to create what are known to be the most challenging conditions for remediation, given the current state of available technologies.

4.4 SPATIAL EXTENT OF THE TECHNICAL IMPRACTICABILITY ZONE

Previous remedial efforts in OU1 have had a range of results. As mentioned above, the OU1 slurry wall combined with the groundwater extraction and treatment system was highly effective in restoring the overburden aquifer outside the slurry wall. However, similar efforts inside the slurry wall (where residual DNAPL presumably exists based on dissolved PCE concentrations greater than 10% of the solubility limit) have not been very effective and nor have other technologies such as SVE/AS and ISCO, although they have reduced the contaminant mass to some extent. This variation in effectiveness suggests that restoration of bedrock groundwater to RGs may be achievable in areas outside of the potential DNAPL area. As a result, a TI Zone is proposed to include the area where it is interpreted to be technically impractical to achieve the RGs due to the potential or probable presence of DNAPL in overburden or bedrock. EPA recognizes that the presence of DNAPL, particularly in a complex geologic medium such as

fractured bedrock, frequently makes the restoration of groundwater technically impracticable (EPA, 2011b).

Determination of the extent of the proposed TI Zone is based on the extent of the potential and probable DNAPL zones, physical site boundaries including the Souhegan River and the existing overburden slurry wall, and the OU1 boundary. The borders of the proposed TI Zone to the east and north were generally based upon the results of solute transport modeling using the EPA "BIOCHLOR" software which concluded that dissolved concentrations less than approximately 175 µg/L will naturally attenuate within a time period of approximately 100 years assuming that the upgradient source has been removed. A detailed description of the modeling is presented in Appendix A. Based on the approximate location of the current bedrock groundwater concentrations of 175 µg/L within OU1, a TI Zone boundary is proposed that is located along the northern bank of the Souhegan River, the OU1 boundary to the east, and the slurry wall to the south and west as shown on Figure 4-1.

The proposed TI Zone encompasses the area shown in Figure 4-1 and extends vertically to include the full extent of the contamination (currently not defined, but extending to at least 600 ft below existing grade). The TI Zone corresponds to the area of highest bedrock contaminant concentrations and includes the area encompassed by the slurry wall as well as downgradient areas south of the Souhegan River, east to the OU1/OU2 boundary. The vertical extent of the TI Zone is depicted on Figure 4-2 (cross-section X-X') and Figure 4-3 (cross-section Y-Y'). The DNAPL within this zone will continue to dissolve slowly into the groundwater maintaining dissolved concentrations exceeding the RGs for the foreseeable future (i.e., hundreds of years).

4.5 EVALUATION OF POTENTIAL REMEDIAL TECHNOLOGIES

An evaluation of current and innovative remedial technologies was conducted to assess their applicability for treatment of groundwater within the TI Zone. To be successful, a candidate technology must be capable of reducing VOC contamination in fractured bedrock at depths greater than 600 ft by nearly five orders of magnitude (100,000 µg/L to <5 µg/L), within a reasonable timeframe[3]. Each technology was evaluated solely on its likely effectiveness in meeting the above criteria, its implementability, and timeliness. Cost was not considered for this evaluation because no technology was identified that could restore the groundwater within a reasonable timeframe at any cost.

The general response actions, remedial technologies, and associated process options that were evaluated for applicability within the TI Zone are presented in Table 4-2. A detailed description of the technologies is presented in Appendix B. Technologies that are not capable of reducing contaminant mass, such as physical containment and ICs, were not included in the assessment because achievement of the RGs cannot occur without reducing contaminant mass. The identified technologies included MNA, groundwater extraction and treatment, excavation, and a variety of *in situ* technologies that use various methods to destroy and/or remove the VOCs.

Table 4-3 provides the results of the screening of the technologies listed in Table 4-2. The technologies were evaluated with regard to effectiveness, implementability, and timeliness. As shown on Table 4-3, no remedial technology exists that can restore groundwater within a reasonable timeframe. Excavation and permeable reactive barriers were determined to not be implementable at this Site due to the vertical extent of contamination within the deep bedrock. While MNA, groundwater extraction, and the *in situ* technologies could conceivably reduce the observed concentrations to the RGs eventually, it would take hundreds if not thousands of years to do so. *In situ* technologies that require the injection of various amendments (ISCO, *in situ* chemical reduction (ISCR), and *in situ* biological treatment) are all limited by the likely presence

[3] EPA's TI guidance states: "…restoration timeframes may be an important consideration in remedy selection, no single timeframe can be specified during with restoration must be achieved to be considered technically practicable. However, very long restoration timeframes (e.g., longer than 100 years) may be indicative of hydrogeologic or contaminant-related constraints to remediation. While predictions of restoration timeframes may be useful in illustrating the effects of such constraints, EPA will base TI decisions on an overall demonstration of the extent of such physical constraints at a site, not on restoration timeframe analyses only.

of DNAPL in deep bedrock fractures. Because of the planar geometry of the fractures, a large volume of DNAPL would have only a very small surface area exposed to the amendments, thereby severely limiting the effectiveness of *in situ* technologies which require direct contact. Some of these technologies may be effective in achieving mass reduction of the DNAPL, but not eliminating it, as part of an effort to prevent contaminant migration. There is really only one technology that could possibly achieve the RGs within a reasonable timeframe and that is *in situ* thermal treatment (ISTT). However, ISTT has never been implemented on nearly the scale that would be required at this Site nor at the required depths. Even if the total volume of bedrock could be heated to the required temperature to facilitate volatilization of the DNAPL, it is not certain that an adequate extraction system could be designed to collect the vapors. As a result, the ISTT technology was determined to not be implementable on the scale required for this Site to achieve RGs, although it may be a practicable alternative for mass reduction of the DNAPL, as part of an effort to prevent contaminant migration.

4.6 PROTECTIVENESS UNDER A TI ARAR WAIVER

Under a TI Waiver scenario where chemical-specific ARARs and risk-based standards would be waived within the TI Zone, long-term protectiveness would be provided by:

- Implementing ICs to permanently prohibit use of groundwater within the TI Zone. Such ICs may include establishment of municipal restrictions associated with the groundwater protection district (e.g., a Town Ordinance), and/or individual deed restrictions or notices (as permitted under the regulations).
- Prevention of migration of dissolved-phase contamination across the TI Zone boundary.
- Treatment of the source area to reduce contaminant mass.
- Monitoring to verify that dissolved-phase contamination is not migrating beyond the TI Zone boundary.

The FS to which this TI Evaluation Report is attached further evaluates the technologies mentioned above with regard to their ability to meet these requirements. The FS identifies ISCO, ISCR, and hydraulic containment as the three most promising technologies for meeting the objectives within the TI Zone. The ISCO and ISCR alternatives would provide greater contaminant mass reduction in bedrock in addition to managing migration of contamination outside of the TI Zone. They would both rely on the existing slurry wall and a permeable reactive

barrier to manage migration in the overburden. Hydraulic containment is able to manage contaminant migration but has limited ability to reduce contaminant mass within the TI Zone. Under the hydraulic containment alternative, the existing remedy would be continued for the overburden zone where it has proved successful for managing migration. All alternatives include a robust monitoring network that will be used to evaluate remedy performance and assess compliance at the TI Zone boundary.

SECTION 5

CONCLUSIONS

5. CONCLUSIONS

This report provides an evaluation of the technical impracticability of restoring groundwater within the proposed TI Zone in OU1, based on available data. The conclusion of this report is that restoration of the overburden and bedrock aquifers to drinking water standards in a reasonable timeframe is not practical for the following reasons:

- The documented presence of PCE in bedrock fractures at a depth of 500 ft below grade at a concentration that is 67% of the solubility limit nearly assures that DNAPL is present in deep bedrock. The CSM is consistent with the release of a large volume of PCE (possibly on the order of 30,000 gallons or more) was discharged to the ground at the former OK Tool facility and that the PCE DNAPL migrated vertically through the overburden aquifer and gained access to the bedrock via a trough located near the east end of the slurry wall. Vertical migration of the DNAPL in bedrock was enhanced by the steeply dipping fracture system, allowing the DNAPL to extend to great depths.

- The presence of dissolved chlorinated solvent contamination in groundwater several hundred to several thousand feet from the release area demonstrates widespread bedrock and overburden contamination.

- The full extent of the depth of contamination in the deep bedrock has not been determined. Studies regarding fracture density and yield with depth for similar crystalline bedrock in Maine (Loiselle, et.al., 1995) suggest that there is little reduction in the frequency or yield of hydraulically-active fractures at depths as great as several thousand feet. This suggests that the contamination in bedrock could extend substantially deeper than currently mapped. A significant and costly investigation would be required to more fully characterize the vertical extent of contamination. Even with installation of numerous additional monitoring wells, the characterization of the source zone and extent of dissolved-phase contamination would likely still not be conclusive due to the heterogeneous nature of fractured bedrock. Without a firm grasp of the location and extent of the contamination (particularly the DNAPL), it is nearly impossible to design an effective remediation system to restore bedrock groundwater.

- Removal of DNAPL from fractured bedrock and restoration of groundwater to drinking water standards in DNAPL zones within a reasonable time-frame is extremely difficult, if not impossible, due to limited natural or induced flushing within bedrock fractures and the difficulty of delivering treatment reagents or processes to the bedrock matrix, particularly dead-end fractures. Also, back diffusion from the matrix will likely cause concentrations to persist above groundwater standards for many years. Current remedial technologies are not effective in restoring DNAPL zones in porous and fractured media, particularly in this type of complex setting.

- The overburden aquifer is a high-permeability sand and gravel aquifer with discontinuous low-permeability lenses overlying low permeability glacial till. Observed dissolved concentrations of PCE in overburden groundwater suggest that residual DNAPL exists in the low permeability lenses and/or glacial till. These characteristics limit the hydraulic accessibility of DNAPL and, coupled with the low permeability zones, make removal or treatment of the DNAPL and restoration of groundwater to RG within a reasonable time frame unlikely.

- Screening of available remedial technologies was unable to identify a technology that would be effective in restoring DNAPL zones in complex heterogeneous geologic environments to drinking water quality in a reasonable time frame. Various technologies implemented at the Site, over the last 30 years including excavation, SVE/AS, ISCO, and groundwater extraction, were all employed in an effort to remove contaminant mass from the overburden and shallow bedrock aquifers. , These technologies have removed less than 10% of the suspected mass of contamination (roughly 3,000 lbs of PCE) from the overburden and shallow bedrock aquifer within OU1, and are untested in the deep bedrock aquifer at the Site. This historic level of performance, combined with the uncertainty with implementability and effectiveness, further demonstrates the impracticability of restoring groundwater at this Site.

For these reasons, a TI Waiver of ARARs is warranted for the portion of OU1 where DNAPL is potentially present based on current data. This area, designated as the proposed TI Zone, is shown on Figure 4-1. The TI Zone consists of a specific area of the overburden and bedrock aquifers in OU1 that encompasses the presumed original release of contamination associated with the former OK Tool facility as well as the downgradient area where natural attenuation of COCs below the RGs would likely not occur within approximately 100 years following source zone treatment, based on preliminary modeling. The lateral boundaries of the TI Zone have been drawn on Figure 4-1 to extend to the OU2 boundary to the east and northeast, to the nearest bank of the Souhegan River to the north and northwest, and areas roughly corresponding to the OU1 boundary to the south and just west of the slurry wall. There is no limit to the depth of the TI Zone; however, based upon current data, DNAPL is suspected to extent to a depth greater than 500 ft (see Figures 4-2 and 4-3).

Protectiveness within the TI Zone can be achieved through the application of ICs to prevent human contact with groundwater. Contaminant migration across the TI Zone boundary will be managed to allow downgradient areas to attain RGs. Groundwater monitoring would be performed to verify compliance at the TI Zone boundary and in the downgradient areas to document the future recovery to drinking water standards.

Instituting a TI Waiver for the proposed TI Zone would allow subsequent remedial actions (as described in the FS) to focus on reducing Source Area concentrations to the extent practicable, while eliminating downgradient migration of contaminants, and facilitating restoration of the downgradient dilute plume in the Groundwater Cleanup Area for beneficial use within a reasonable timeframe.

These remedial actions, based upon a TI Waiver of chemical-specific ARARs within the TI Zone, coupled with monitoring of groundwater and implementation of ICs to prevent exposure to contaminated groundwater, will provide an appropriate remedial response for the TI Zone that will provide protection to human health.

SECTION 6

REFERENCES

6. REFERENCES

Chapman, S.W., and B.L Parker. 2005. *Plume Persistence Due to Aquitard Back Diffusion Following Dense Nonaqueous Phase Liquid Source Removal or Isolation.* Water Resources Research Vol. 41, No. 12: pages 4224-4235.

EPA (United States Environmental Protection Agency). 1991a. *EPA Proposed Cleanup Plan for the Savage Municipal Water Supply Site*. July 1991.

EPA. 1991b. *EPA Superfund Record of Decision: Savage Municipal Water Supply, EPA ID: NHD980671002, OU1, Milford, NH*. September 27.

EPA. 1993. *Guidance for Evaluating the Technical Impracticability of Ground-Water Restoration, Interim Final*, EPA Directive 9234.2-25. September.

EPA. 1996. *EPA Superfund Explanation of Significant Differences: Savage Municipal Water Supply, EPA ID: NHD980671002, OU1, Milford, NH*. December 19.

EPA. 2011a. *First Five-Year Review Report, Savage Municipal Water Supply Superfund Site, Milford, New Hampshire*. September.

EPA. 2011b. *Clarification of OSWER's 1995 Technical Impracticability Waiver Policy*. OSWER Directive #9355.5-32. September 19.

Federal District Court of New Hampshire (D.N.H.), 1994. *Consent Decree. United States of America and State of New Hampshire v. Conductron Corporation d/b/a Hendrix Wire and Cable et al.*, Civil Action No. 94-174L, 27 June.

Geosyntec (Geosyntec Consultants). 2004. *Assessing the Feasibility of DNAPL Source Zone Remediation: Review of Case Studies.* NAVFAC Contract Report CR-04-002-ENV. May.

Gradient (Gradient Corporation). 2011. *Draft Annual Groundwater Quality Monitoring Report – 2010, Savage Municipal Water Supply Well Superfund Site, Milford, New Hampshire*. June 2.

Gradient. 2013. *Final 2011 Annual Groundwater Quality Monitoring Report, Savage Well Superfund Site, Operable Unit 2*. January 31.

Harte, P.T., and Mack, T.J. 1992. *Geohydrology of, and Simulation of, Groundwater Flow in the Milford-Souhegan Glacial-Drift Aquifer, Milford, New Hampshire*: *U.S. Geological Survey Water-Resources Investigations Report 91-4177*.

Harte, P.T., 2006. *Effects of a Remedial System and its Operation on Volatile Organic Compound Contaminated Groundwater, Operable Unit 1, Savage Municipal Well Superfund Site, Milford, New Hampshire, 1998 – 2004*. U.S. Geological Survey SIR 2006-5083.

HMM Associates, Inc. 1991a. *Remedial Investigation, Savage Municipal Water Supply Site, Milford, New Hampshire.* June.

HMM Associates, Inc. 1991b. *Feasibility Study, Savage Municipal Water Supply Site, Milford, New Hampshire*. June.

Interstate Technology & Regulatory Council. 2011. *Integrated DNAPL Site Strategy Technical/Regulatory Guidance.* November.

Lipson, S.D., B.H. Kueper, and M.J. Gefell. 2005. *Matrix Diffusion-Derived Plume Attenuation in Fractured Bedrock.* Ground Water, Vol. 43, No. 1, January-February 2005 (pages 30-39).

Loiselle, M, and D. Evans. 1995. *Fracture Density Distributions and Well Yields in Coastal Maine*. Ground Water, Vol. 33, No. 2, March-April 1995 (pages 190-196).

Montgomery, J.H., and L.M. Welkom. 1990. *Groundwater Chemicals Desk Reference.* Lewis Publ., Chelsea, MI. 650 p.

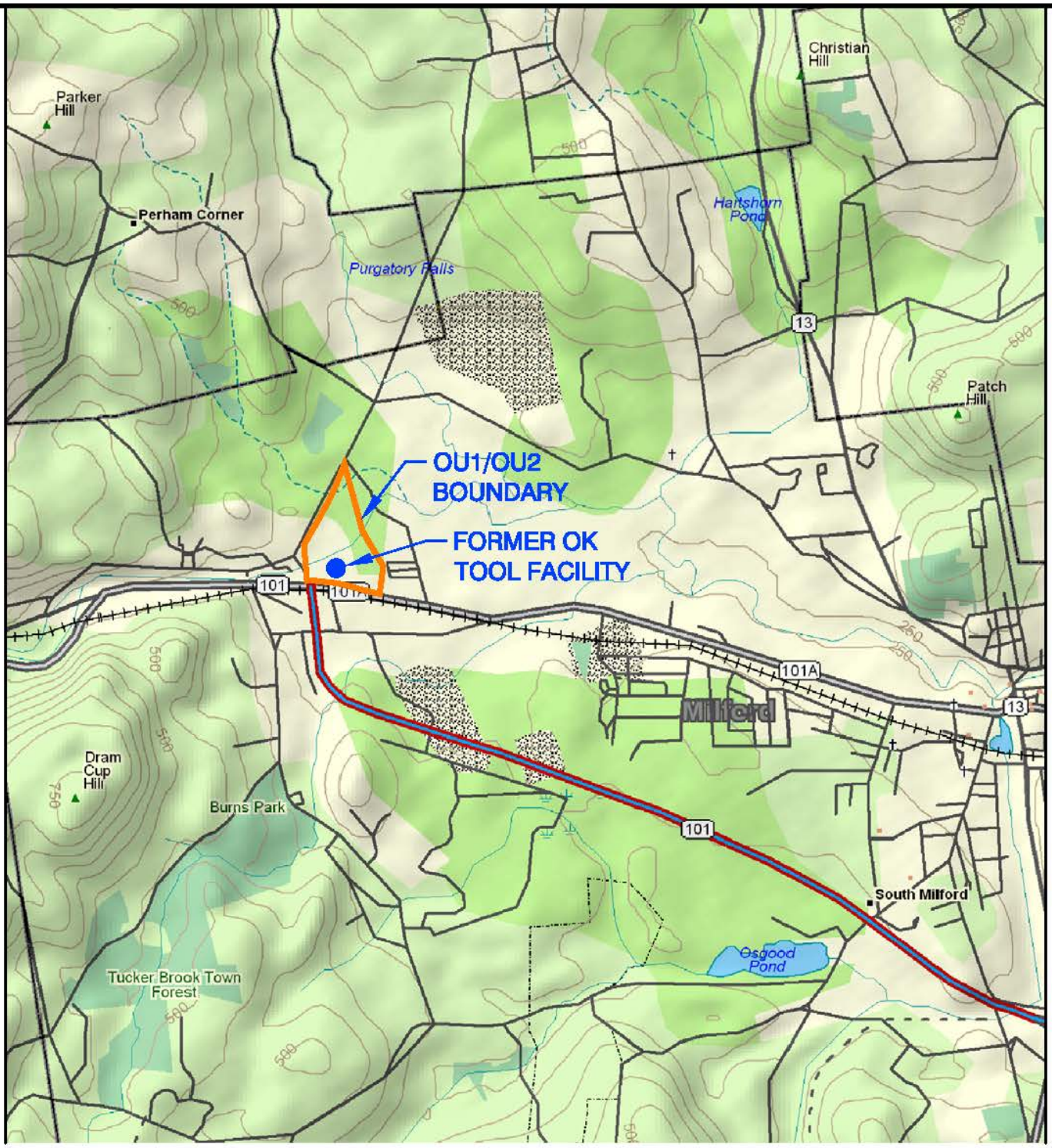
NRC (National Research Council). 1994. Alternatives for Groundwater Cleanup, Washington, D.C. 1994.

NRC. 1996. Rock Fractures and Fluid Flow – Contemporary Understanding and Applications, Washington, D.C. 1996

WESTON (Weston Solutions, Inc.). 2011. *Draft 2010 Annual Report, Savage Municipal Water Supply Superfund Site OU-1, 621 Elm Street, Milford, New Hampshire*. May.

WESTON. 2014. *Remedial Investigation Report, Savage Municipal Water Supply Superfund Site OU-3, 621 Elm Street, Milford, New Hampshire*. March.

FIGURES



SOURCE:
DELORME 3-D TOPOQUADS SOFTWARE; NEW HAMPSHIRE & VERMONT 3-D TOPOQUADS CD

QUADRANGLE LOCATION

GRAPHIC SCALE
3000 1500 0 1500 3000
APPROXIMATE SCALE IN FEET

SITE LOCATION MAP

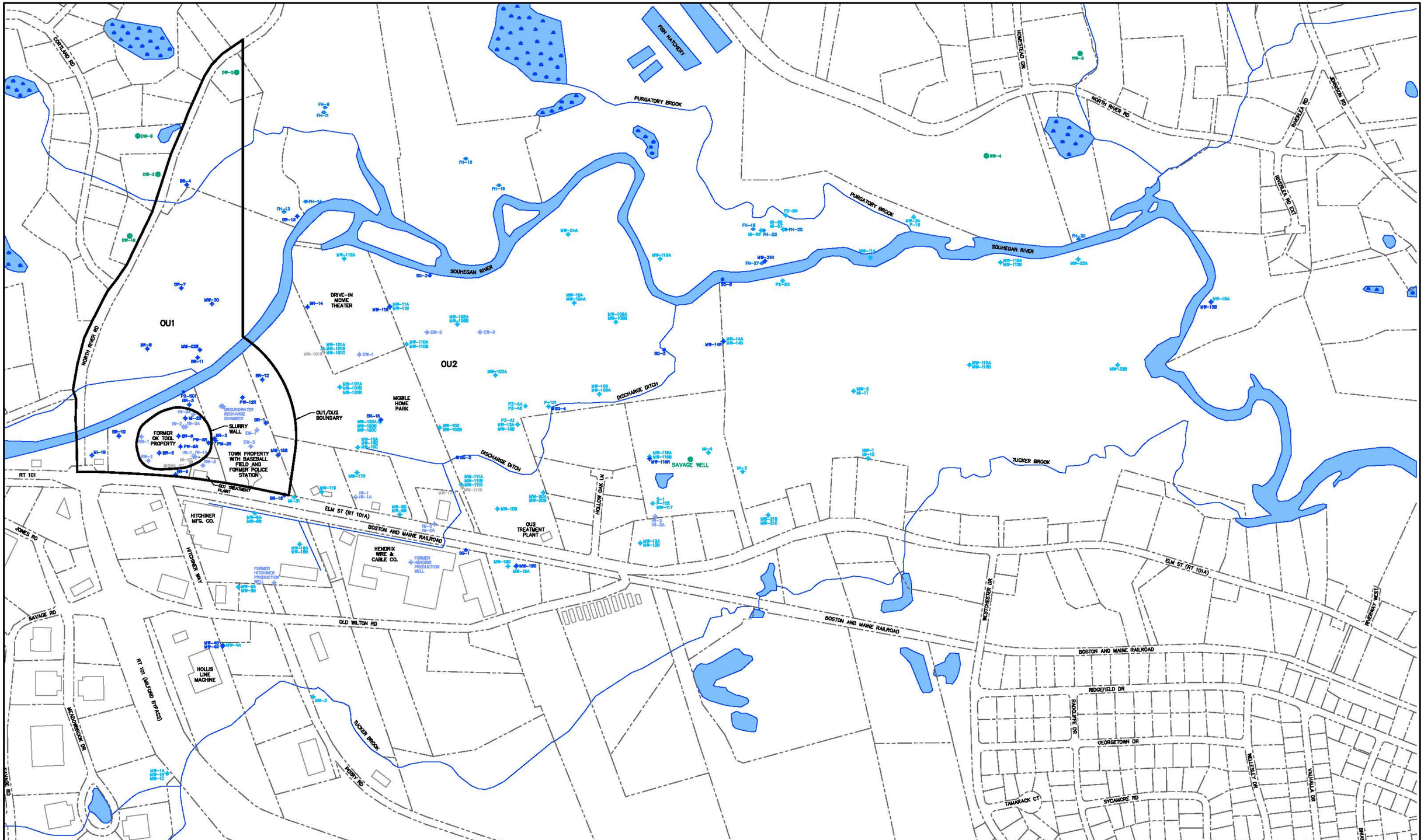
WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	DATE	DES. ENG.	DATE	W.O. NO.
TAC	MAR 2015			20118.016.001
CHECKED	**DATE**	**SCALE**	**REVISION**	**FIGURE NO.**
VD	MAR 2015	AS SHOWN		1-1

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 1-1.dwg, Layout1, 3/30/2015 9:22:37 AM, girardeb, 1:1



NOTE:
OU2 IS THE REMAINING PORTION OF THE CONCENTRATED PLUME NOT INCLUDED IN OU1 AND THE DISSOLVED PLUME.

SOURCE:
PROPERTY MAPS TAKEN FROM TOWN OF MILFORD, NH, TAX MAP DATABASE, MAP NUMBERS B2, B3, B4, C2, C3, C4, D2, D3, D4, E2, E3, & E4, DATED APRIL 2009.

GRAPHIC SCALE
300 150 0 150 300
APPROXIMATE SCALE IN FEET

LEGEND

- BR-9 BEDROCK MONITORING WELL
- MW-101R ABANDONED BEDROCK MONITORING WELL
- MW-102 OVERBURDEN MONITORING WELL
- IW-1A INJECTION, EXTRACTION, OR RECHARGE WELL
- SG-6 STREAM GAUGE
- FH-3 FISH HATCHERY WELL
- DW-16 DRINKING WATER WELL

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

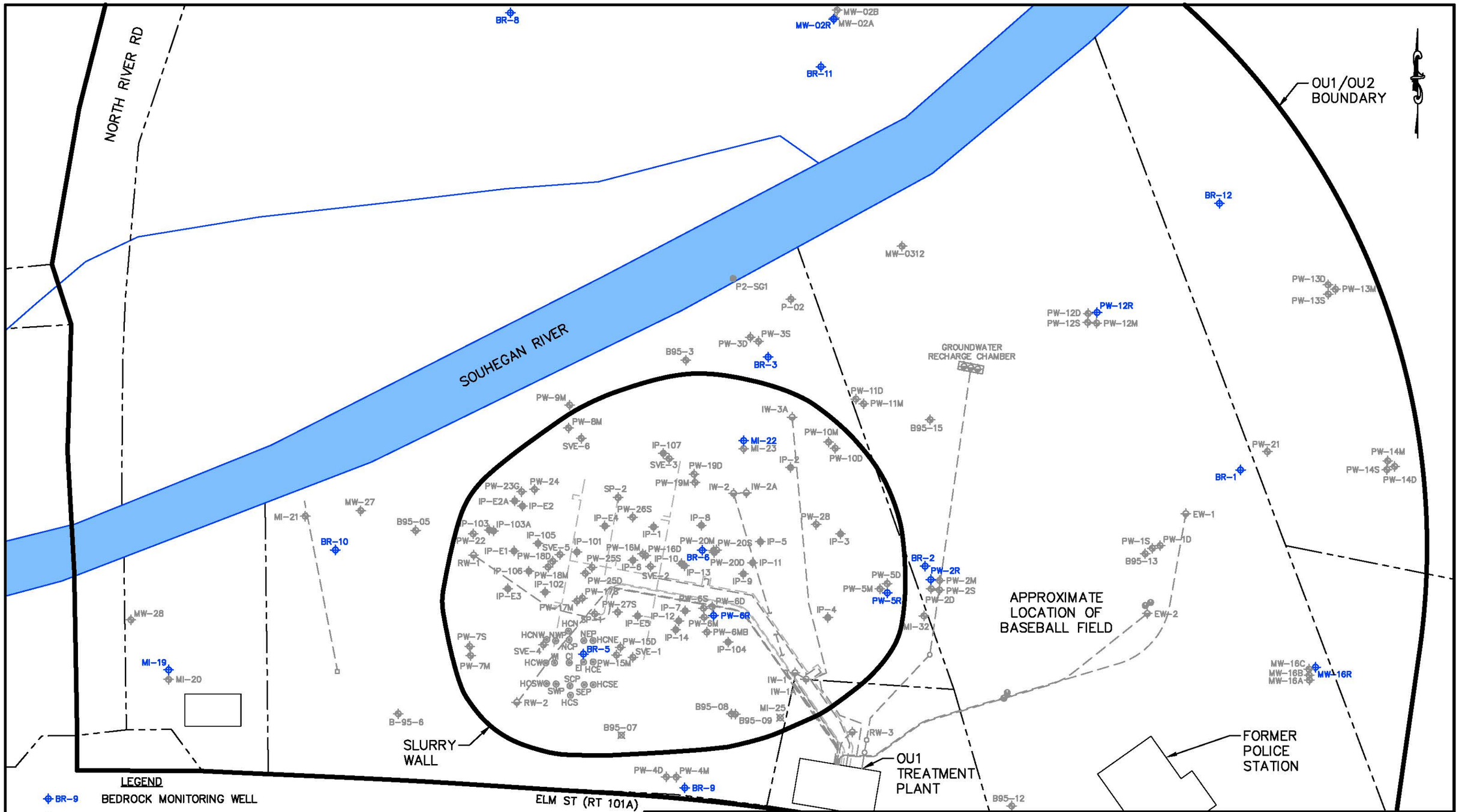
WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

SITE PLAN

DRAWN	DATE	FIGURE NO.	REV. NO.
BEG	MAR 2015	1–2	
SCALE	W.O. NO.	SHT. ___ OF ___	
AS SHOWN	20118.016.001		

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 1-2.dwg, Layout 1, 6/3/2015 3:06:40 PM, girardeb, 1:2



LEGEND

- BR-9 BEDROCK MONITORING WELL
- MW-102 OVERBURDEN MONITORING WELL
- IW-1A INJECTION, EXTRACTION, OR RECHARGE WELL
- SG-6 STREAM GAUGE
- HCNW INEEL WELL
- IP-7 INJECTION POINT
- PROPERTY BOUNDARY

GRAPHIC SCALE

100 50 0 50 100

APPROXIMATE SCALE IN FEET

SOURCE:
PROPERTY MAPS TAKEN FROM TOWN OF MILFORD, NH, TAX MAP DATABASE, MAP NUMBERS B2, B3, B4, C2, C3, C4, D2, D3, D4, E2, E3, & E4, DATED APRIL, 2009.

OU1 SITE DETAIL

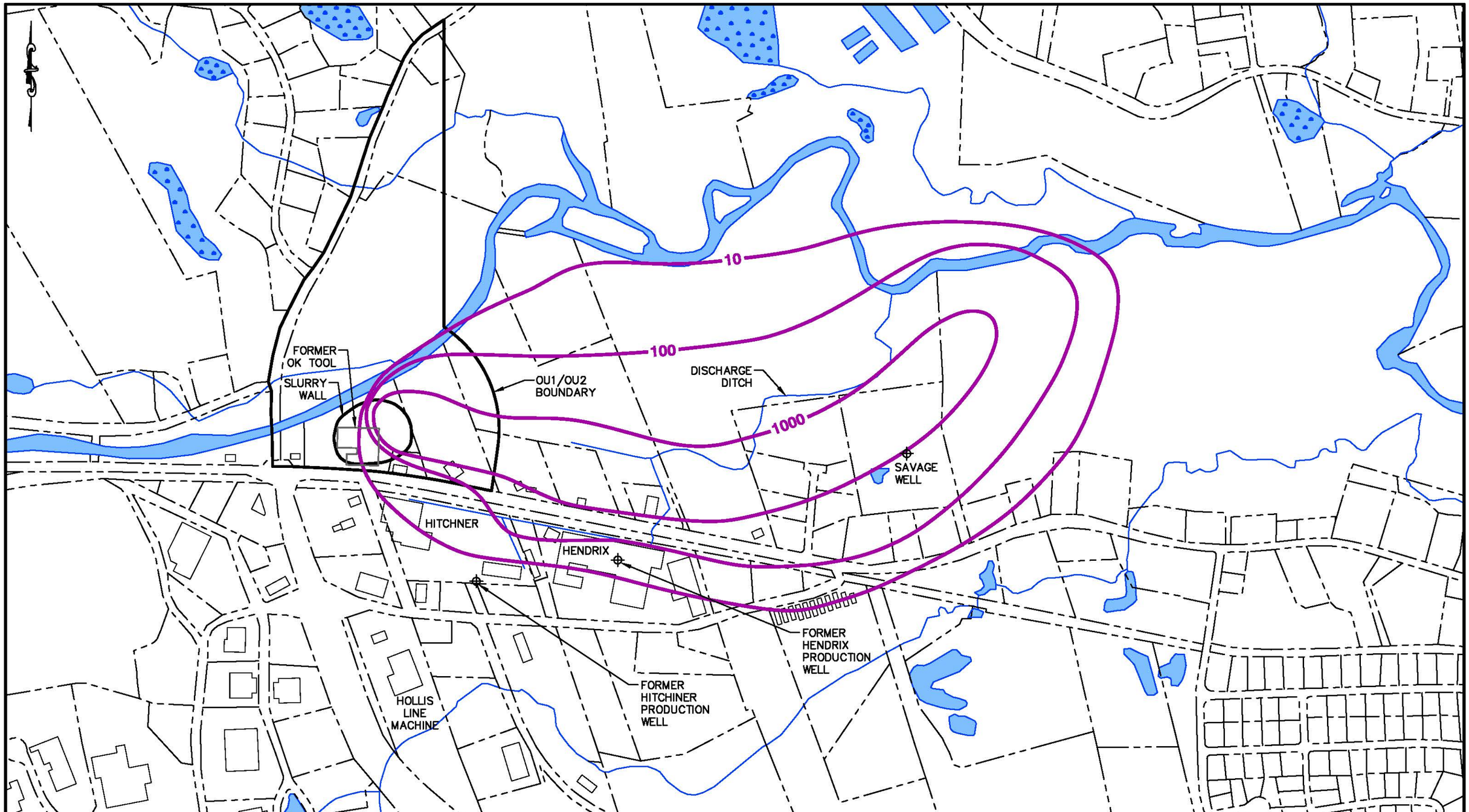
TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

DRAWN	DATE	DES. ENG.	DATE	W.O. NO.
BEG	MAR 2015			20118.016.001
CHECKED	DATE	SCALE	REVISION	FIGURE NO.
VD	MAR 2015	AS SHOWN		1-3

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 1-3.dwg, Layout 1, 3/30/2015 9:25:41 AM, girardeb, 1:1



SOURCE:
1) FIGURE 1, SITE LOCATION MAP SHOWING OU1, THE 1990 PCE PLUME, AND THE LOCATIONS OF THE POTENTIALLY RESPONSIBLE PARTIES (PRPs), BY US FILTER, DATED MAY 20, 2004.
2) PLUME CONTOURS DERIVED FROM THE 1991 REMEDIAL INVESTIGATION.

LEGEND
PCE CONTOUR IN PPB

GRAPHIC SCALE
600 300 0 300 600
APPROXIMATE SCALE IN FEET

HISTORIC 1990 PCE OVERBURDEN PLUME

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

DRAWN	DATE	DES. ENG.	DATE	W.O. NO.
BEG	MAR 2015			20118.016.001
CHECKED	DATE	SCALE	REVISION	FIGURE NO.
VD	MAR 2015	AS SHOWN		3-1

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 3-1.dwg, Layout 1, 3/30/2015 9:28:28 AM, girardeb, 1:1

Legend

Data Obtained from Cluster Wells

- Well Screened in the Shallow Zone Overburden
- Well Screened in the Medium Zone Overburden
- Well Screened in the Deep Zone Overburden
- Well Screened in the Shallow Zone Bedrock

WESTON SOLUTIONS CONCORD NEW HAMPSHIRE

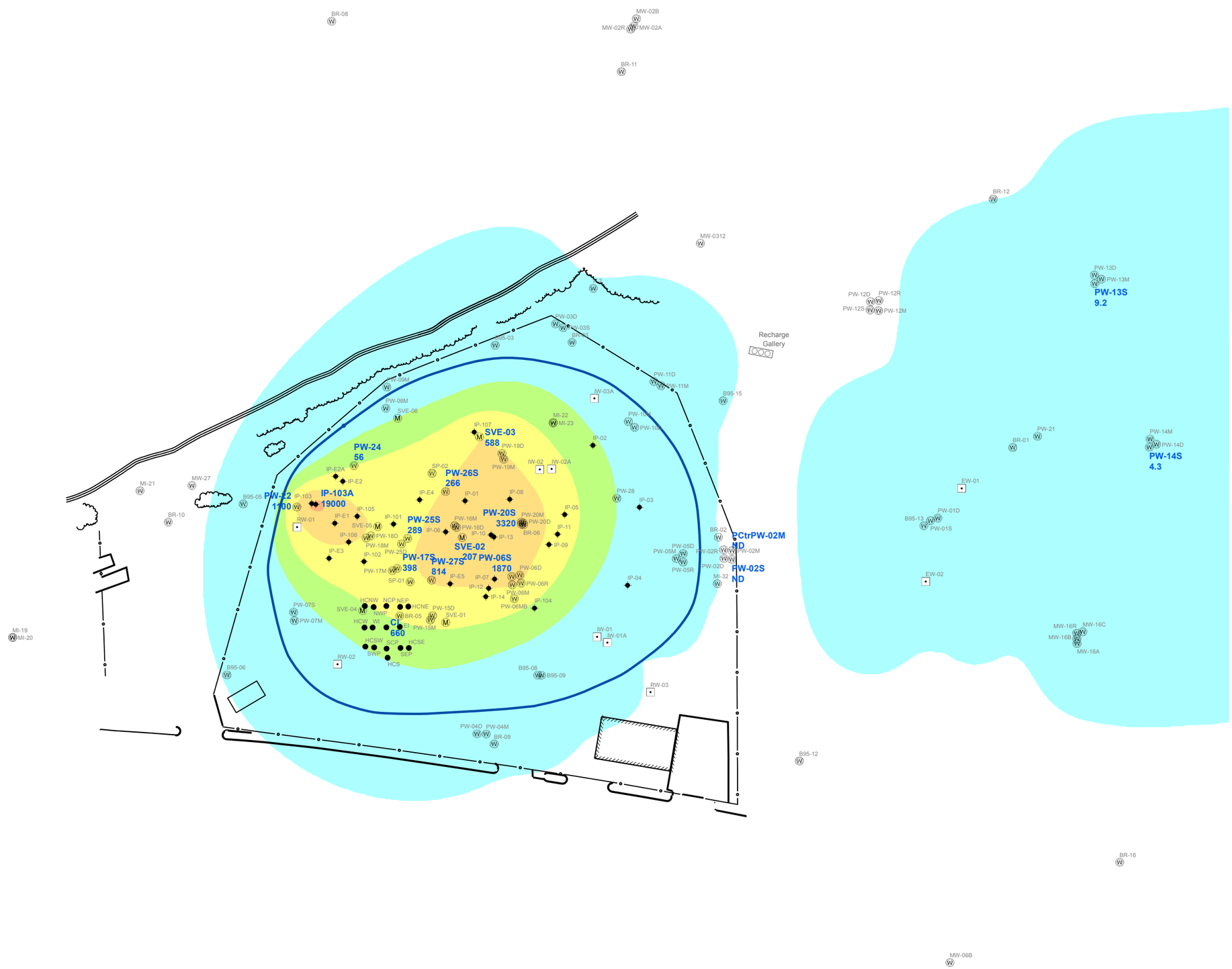
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

PCE TRENDS IN OU-1
OUTSIDE THE SLURRY WALL

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	CHECKED	DATE	W.O. NO.	FIGURE NO.
AL	JS	June 2015	20118.016.001	3-2

FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig3-2_PCE_conc_trend_outside_slurrywall_rev1.mxd 5:26:11 PM 06/04/2015 llua



N

Legend

Location Type

- INEEL
- ISCO Injection Point
- Monitoring Well
- SVE
- Extraction/Recharge Well

Detected Concentration

PW-25S
289

Model Concentration

- >5000ppb
- >500ppb
- >50ppb
- >5ppb (MCL)
- Detection

0 60 120 240
Feet

Note:

1. All Concentrations are in units of ug/L.
2. Only samples taken at the elevation that is within 240 ft and 260 ft amsl interval are shown with concentrations on the figure.
3. Monitoring locations and concentration data highlighted in blue on the figure do not necessarily represent concentrations at this 250 ft amsl elevation, but instead may represent data from a nearby higher or lower elevation that was used to interpolate the concentration at the 250 ft amsl elevation.
4. For illustration purpose, the hot spot at IP-103A is projected to this map as it lies within 2 ft of the 250 ft elevation plane.

ug/L - micrograms per liter
amsl - above mean sea level
MCL - maximum contaminant level
PCE - tetrachloroethylene

WESTON SOLUTIONS, CONCORD NEW HAMPSHIRE

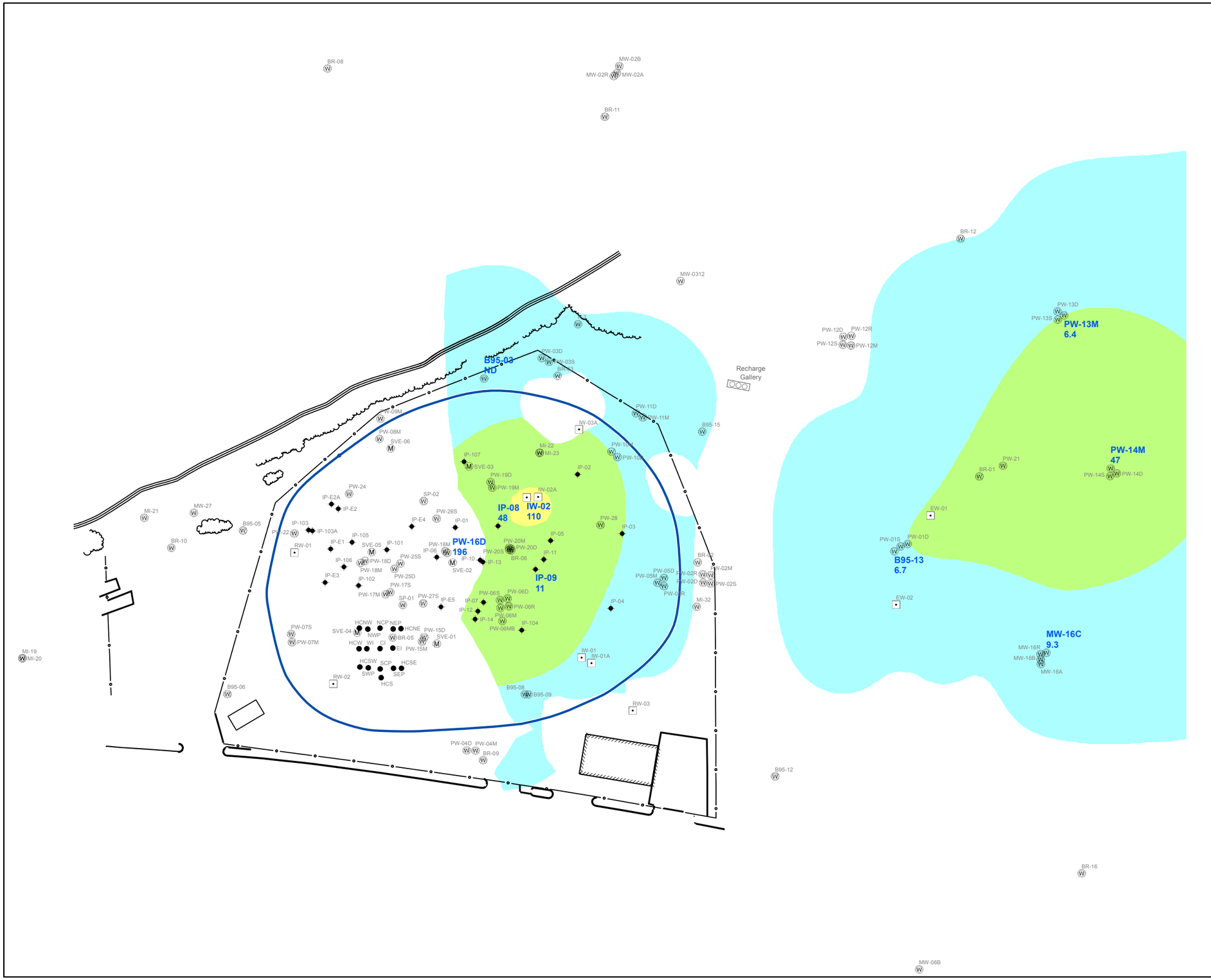
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

TI WAIVER EVALUATION REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

PCE Concentrations
In Shallow Overburden Groundwater
Fall 2014

DRAWN	DATE	DEG. ENG.	DATE	W.O. NO.
AL	Mar 2015			20118.016.001
CHECKED	**DATE**	**SCALE**	**REVISION**	**FIGURE NO.**
JS	Mar 2015	AS SHOWN		3-3

FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig3-3_PCE_ShallowOB_2014.mxd 4:43:01 PM 03/30/2015 liua



N

Legend

Location Type

- INEEL
- ISCO Injection Point
- Monitoring Well
- SVE
- Extraction/Recharge Well

Detected Concentration

PW-16D
123

Model Concentration

- >5000ppb
- >500ppb
- >50ppb
- >5ppb (MCL)
- Detection

0 60 120 240
Feet

Note:

1. All Concentrations are in units of ug/L.
2. Only samples taken at the elevation that is within 10 feet interval of the slicing plane are shown with concentrations on the figure.
3. The slicing plane at 195ft amsl cuts through the till and bedrock in the western portion of the area inside the slurry wall. Therefore, concentrations of PCE were not modeled within these geologic units.
4. Monitoring locations and concentration data highlighted in blue on the figure do not necessarily represent concentrations at the 195 ft amsl elevation, but instead may represent data from a nearby higher or lower elevation that was used to interpolate the concentration at the 195 ft amsl elevation.

WESTON SOLUTIONS CONCORD NEW HAMPSHIRE

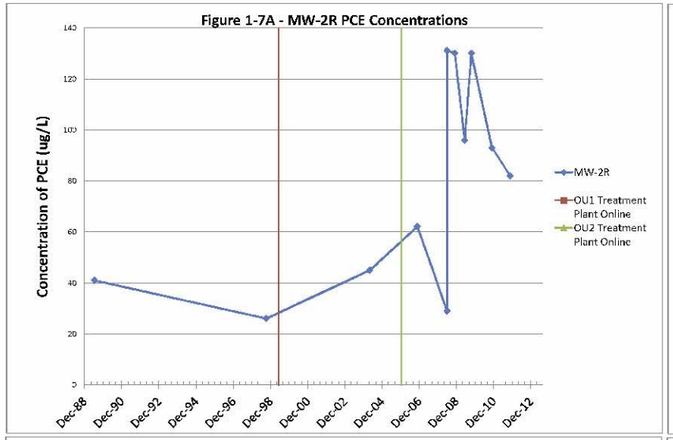
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

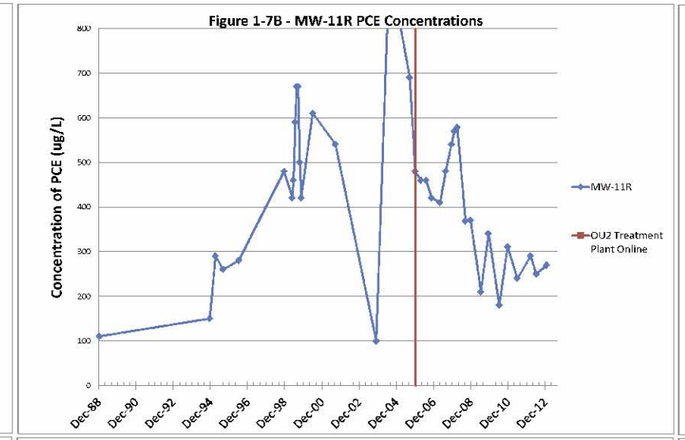
TI WAIVER EVALUATION REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

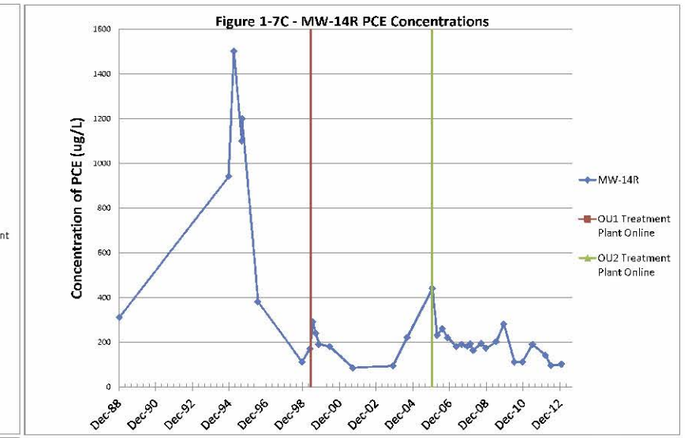
PCE CONCENTRATIONS
IN DEEP OVERBURDEN GROUNDWATER
FALL 2014

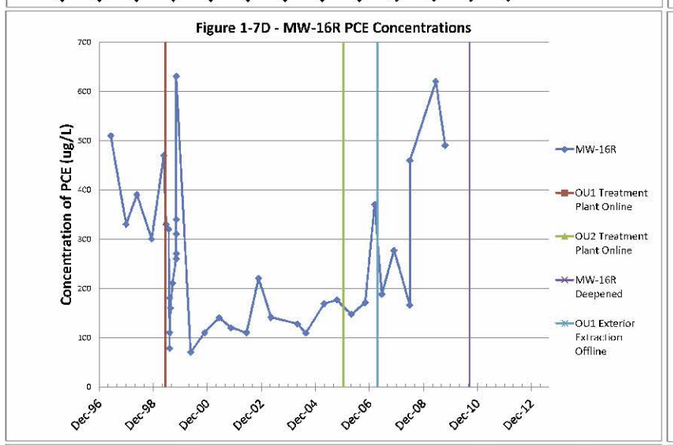
DRAWN	DATE	DEG. ENG.	DATE	W.O. NO.
AL	Apr 2015			20118.016.001
CHECKED	DATE	SCALE	REVISION	FIGURE NO.
JS	Apr 2015	AS SHOWN		3-4

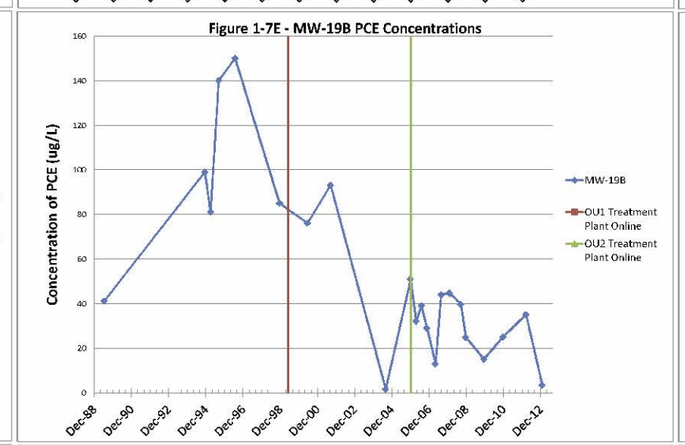
FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig3-4_PCE_Fall14_195ft.mxd 5:24:02 PM 04/01/2015 liua

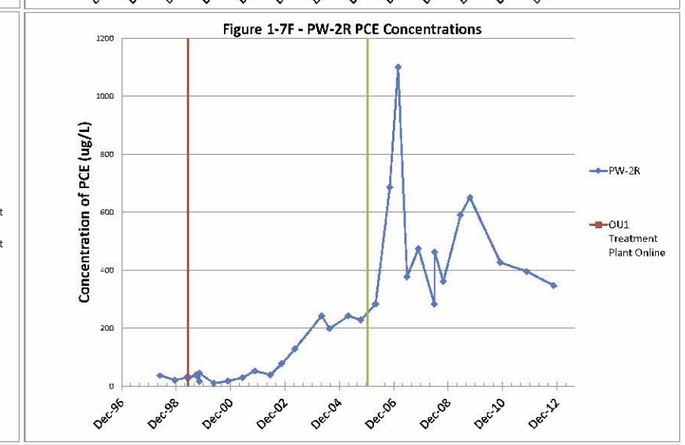


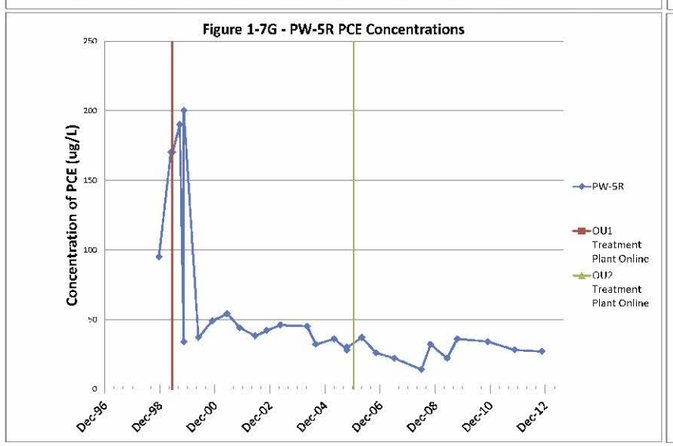


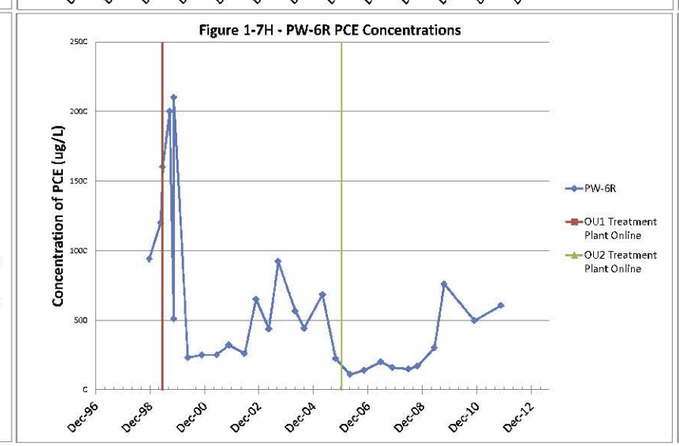


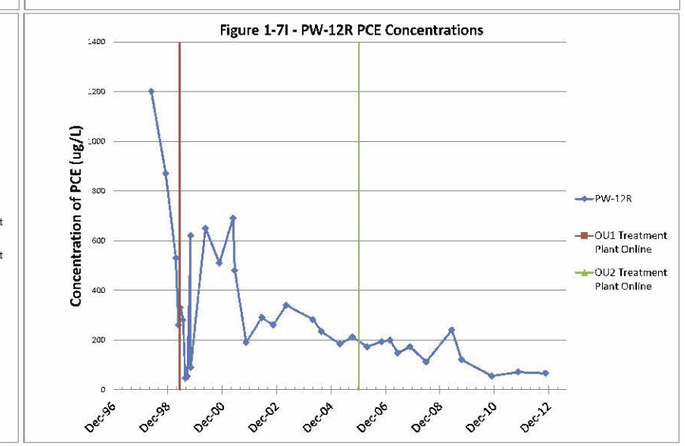








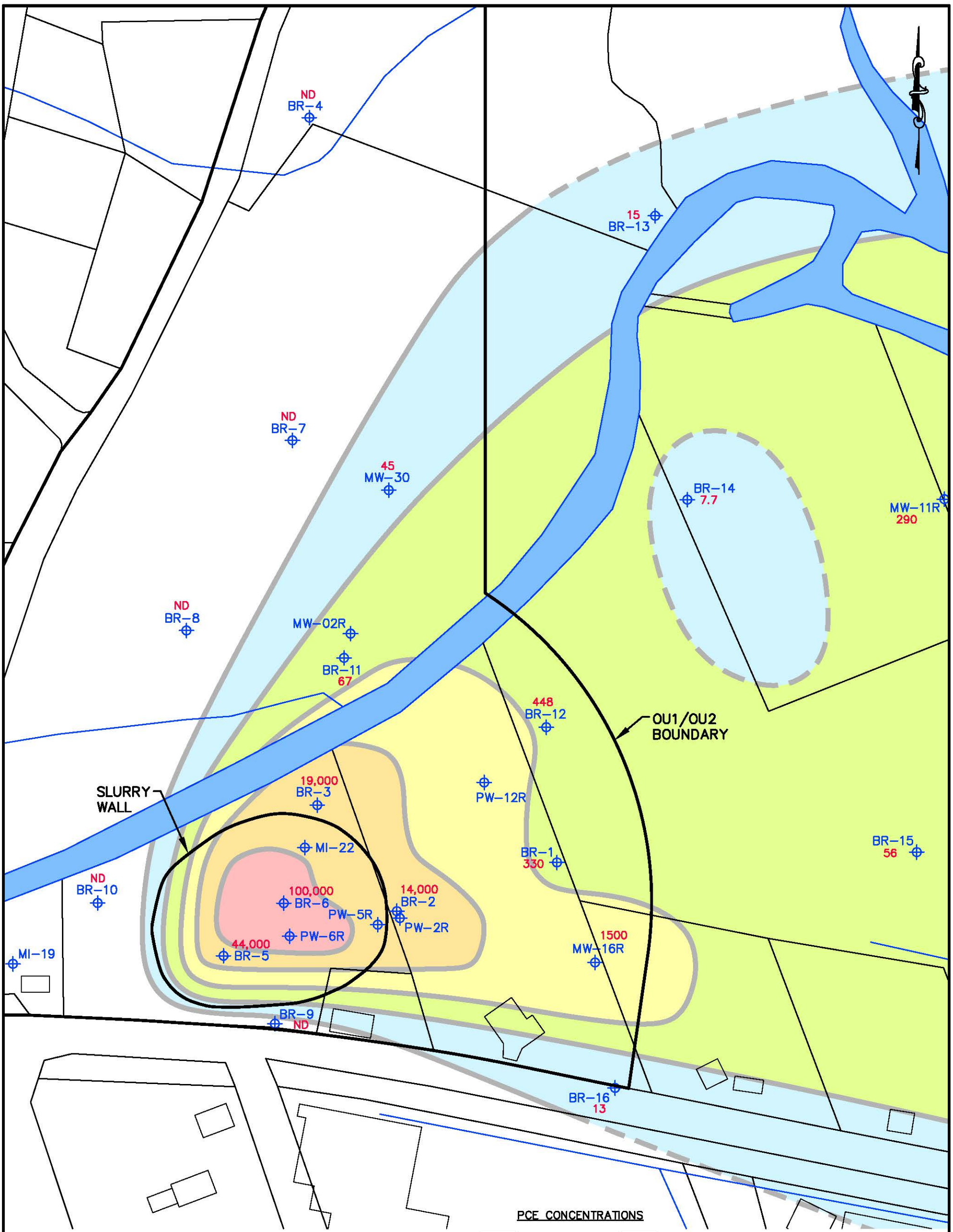




BEDROCK PCE CONCENTRATION TRENDS	WESTON SOLUTIONS CONCORD NEW HAMPSHIRE		NEW HAMPSHIRE DEPARTMENT OF Environmental Services		
TI WAIVER REPORT SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE MILFORD, NEW HAMPSHIRE	DRAWN BEG	DATE MAR 2015	DES. ENG.	DATE	W.O. NO. 20118.016.001
	CHECKED TD	DATE MAR 2015	SCALE AS SHOWN	REVISION	FIGURE NO. 3-5

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 3-5.dwg, Layout 1, 4/1/2015 3:58:41 PM, girardeb, 1:1

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 3-6.dwg, Layout 1, 4/1/2015 4:00:34 PM, girardeb, 1:1



LEGEND

BR-2 MONITORING WELL LOCATION

1500 PCE CONCENTRATION

PLUME IS INFERRED

NOTES:
1) 1590 = PCE DATA IN IN ug/L
2) ND = NON DETECT
3) PCE VALUES FOR EACH WELL ARE THE HIGHEST OBSERVED DURING THE RI (2012-2013)

PCE CONCENTRATIONS

> 50,000 ug/L

> 5000 ug/L

> 500 ug/L

> 50 ug/L

> 5 ug/L

GRAPHIC SCALE
200 100 0 100 200
APPROXIMATE SCALE IN FEET

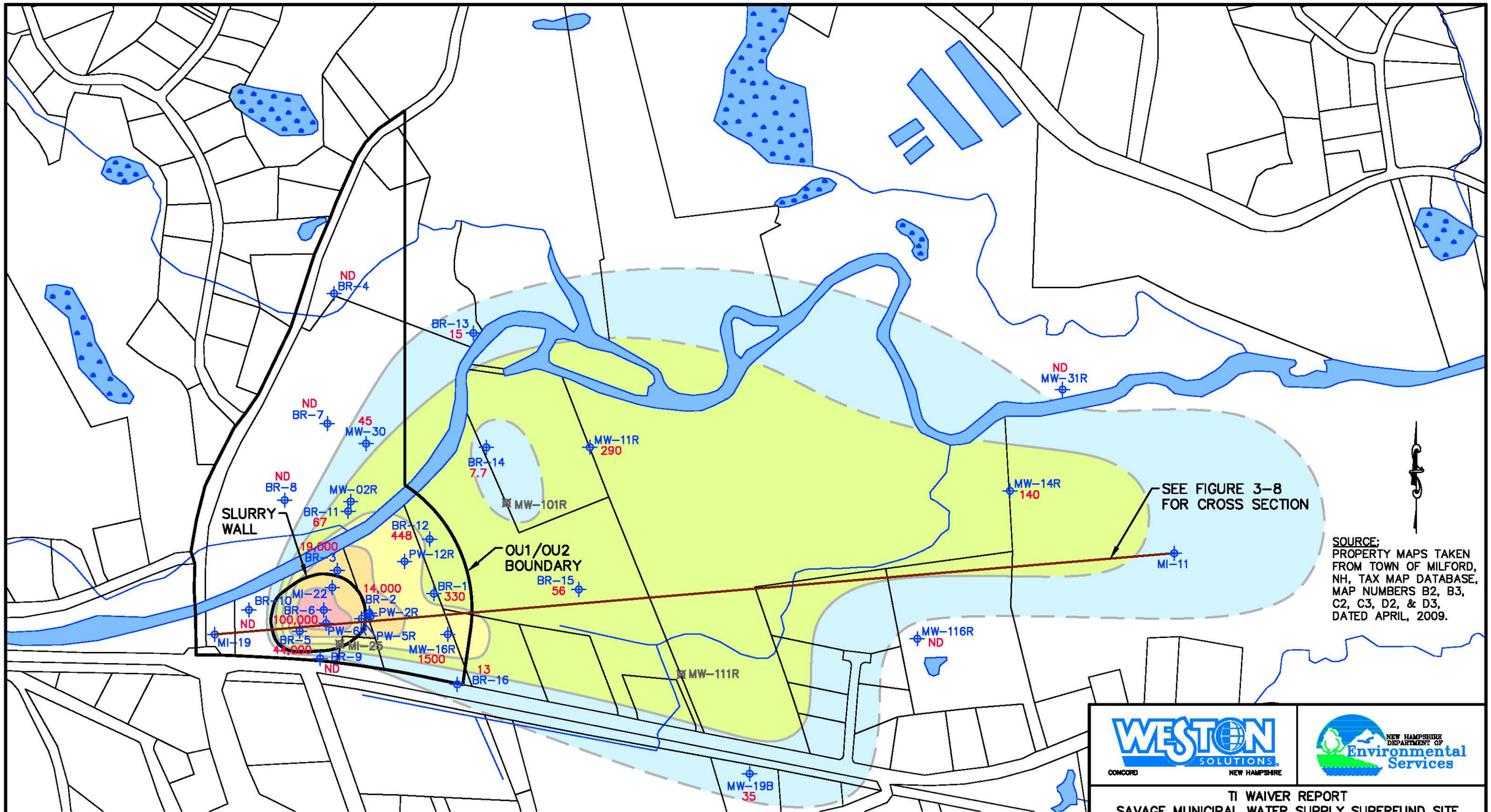
LATERAL EXTENT OF PCE IN OU1 BEDROCK

WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	DATE	DES. ENG.	DATE	W.O. NO.
BEG	MAR 2015			20118.016.001
CHECKED	DATE	SCALE	REVISION	FIGURE NO.
JJS	MAR 2015	AS SHOWN		3-6



WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

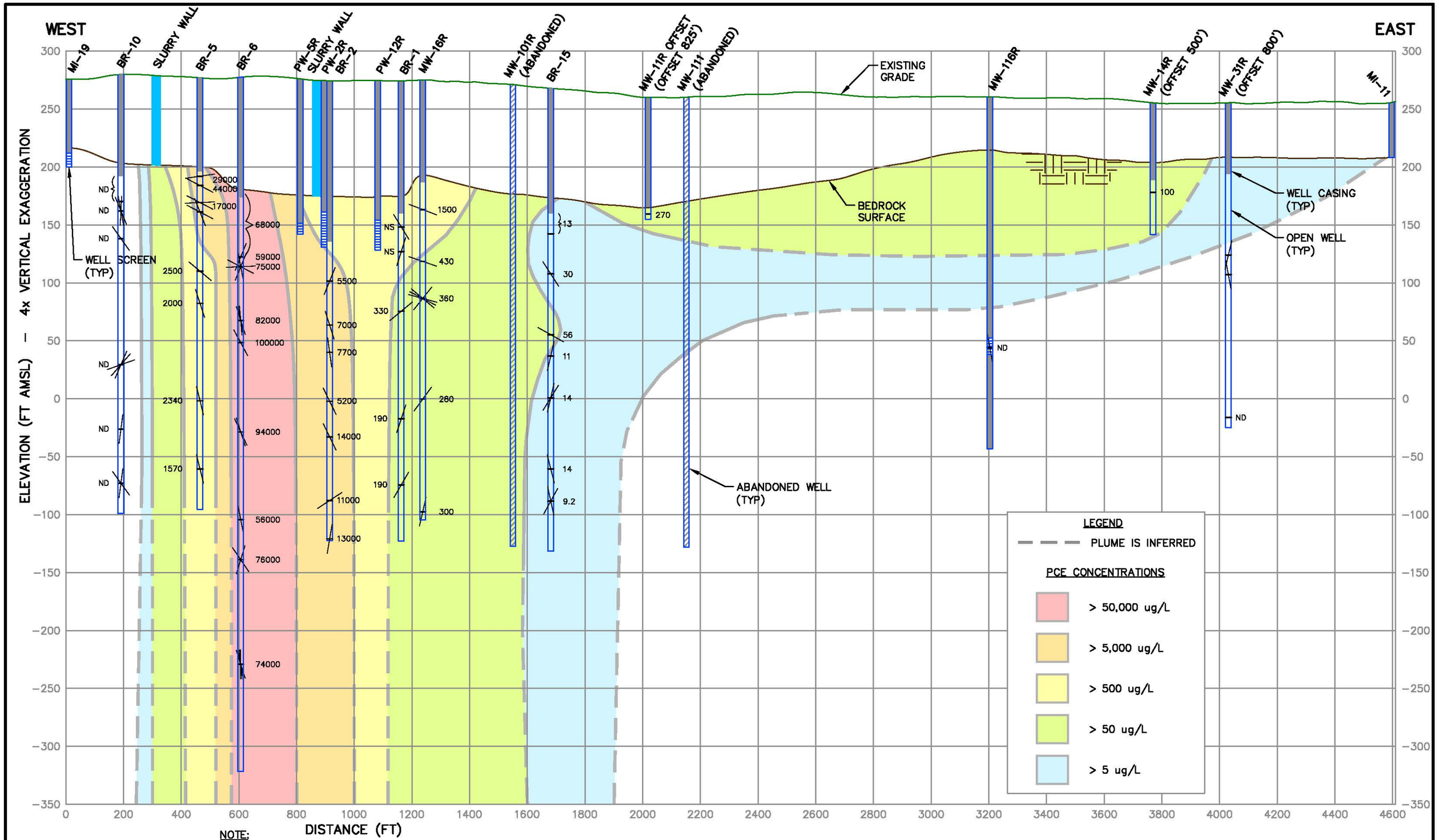
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

LATERAL EXTENT OF PCE IN BEDROCK
SITE-WIDE

DRAWN	DATE	DES. ENG.	DATE	W.O. NO.
BEG	MAR 2015			20118.016.001
CHECKED	DATE	SCALE	REVISION	FIGURE NO.
JJS	MAR 2015	AS SHOWN		3-7

M:\Design\DWG\NHDES\savage\2015\TI Waiver Report\figure 3-7.dwg, Layout 1, 6/3/2015 3:13:05 PM, girardeb, 1:1



HORIZONTAL GRAPHIC SCALE

300 150 0 150 300

APPROXIMATE SCALE IN FEET

NOTE:
1) 5500 = PCE DATA IN ug/L
2) NS = NO SAMPLE
3) ND = NON DETECT
4) PCE VALUES FOR EACH INTERVAL ARE THE HIGHEST OBSERVED DURING THE RI (2010–2013)

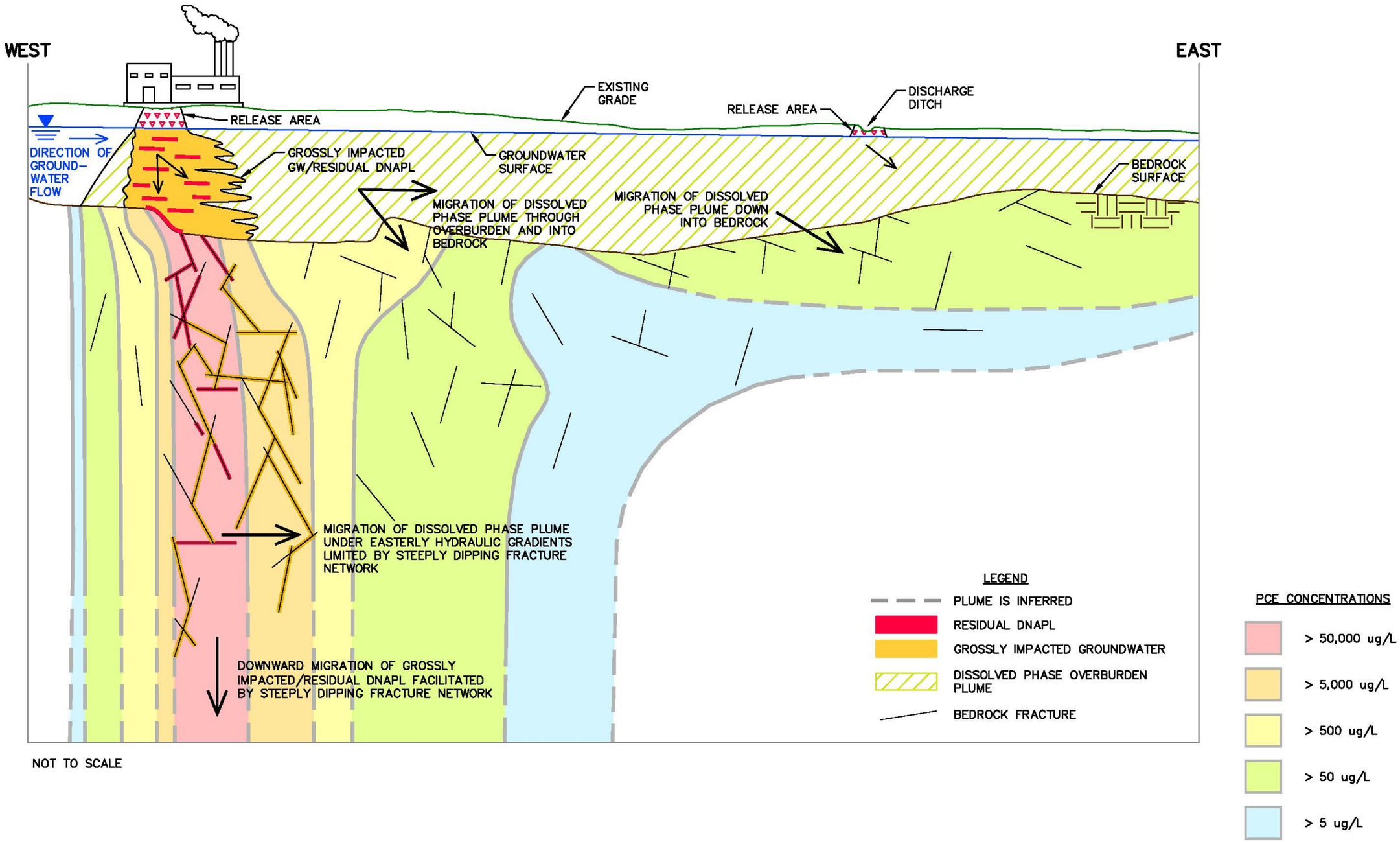
VERTICAL DISTRIBUTION OF PCE IN BEDROCK

TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

BEG	MAR 2015
FIGURE 3–8	



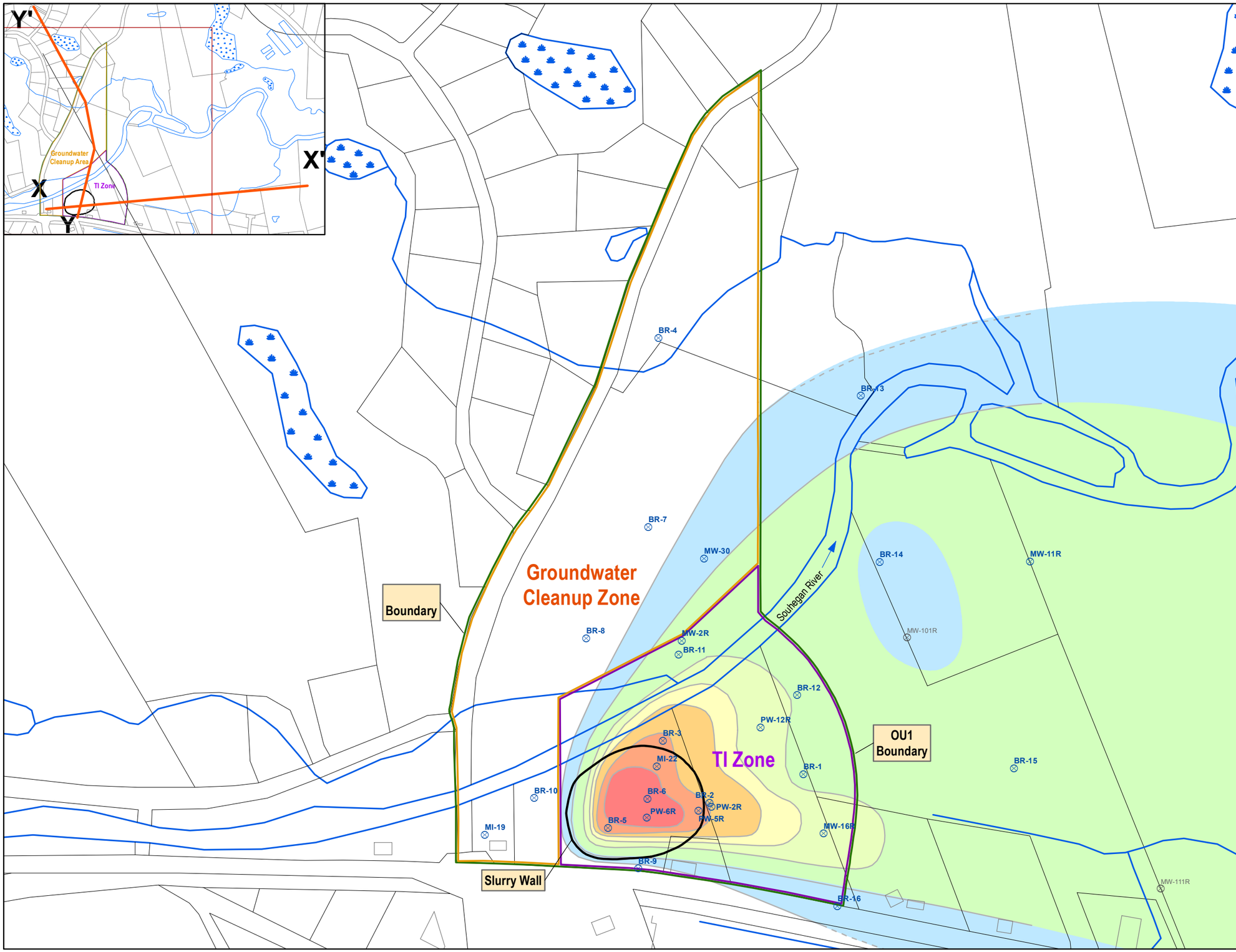
NOT TO SCALE

CONCEPTUAL SITE MODEL
TI WAIVER REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

WESTON SOLUTIONS CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

BEG	MAR 2015
FIGURE 3-9	



N

LEGEND

Bedrock Wells
- Well
- Abandoned

Above Ground Features
- Slurry Wall
- Buildings
- Property boundary
- Waterway & Wetland
- Cross-Section Line
- OU1 Boundary
- TI Zone
- Groundwater Cleanup Zone

PCE Concentrations in Bedrock Groundwater
- > 50,000 ug/L
- 15,000-50,000 ug/L
- 5,000-15,000 ug/L
- 1,500-5,000 ug/L
- 500-1,500 ug/L
- 50-500 ug/L
- 5-50 ug/L

Notes:
1. The PCE plume is configured based on Figure 7-2 of the Remedial Investigation Report, Savage Municipal Water Supply Superfund Site OU3, Milford, NH, 2014, which was based on the highest observed PCE values for each well during the Remedial Investigation (2010 to 2013).

ug/L - micrograms per liter
mg/L - milligrams per liter
DNAPL - Dense Non-Aqueous Phase Liquid
PBL - Physical Bound Level
PCE - tetrachloroethene
TI - Technical Impracticability

0 175 350 700
Feet

WESTON SOLUTIONS
CONCORD NEW HAMPSHIRE

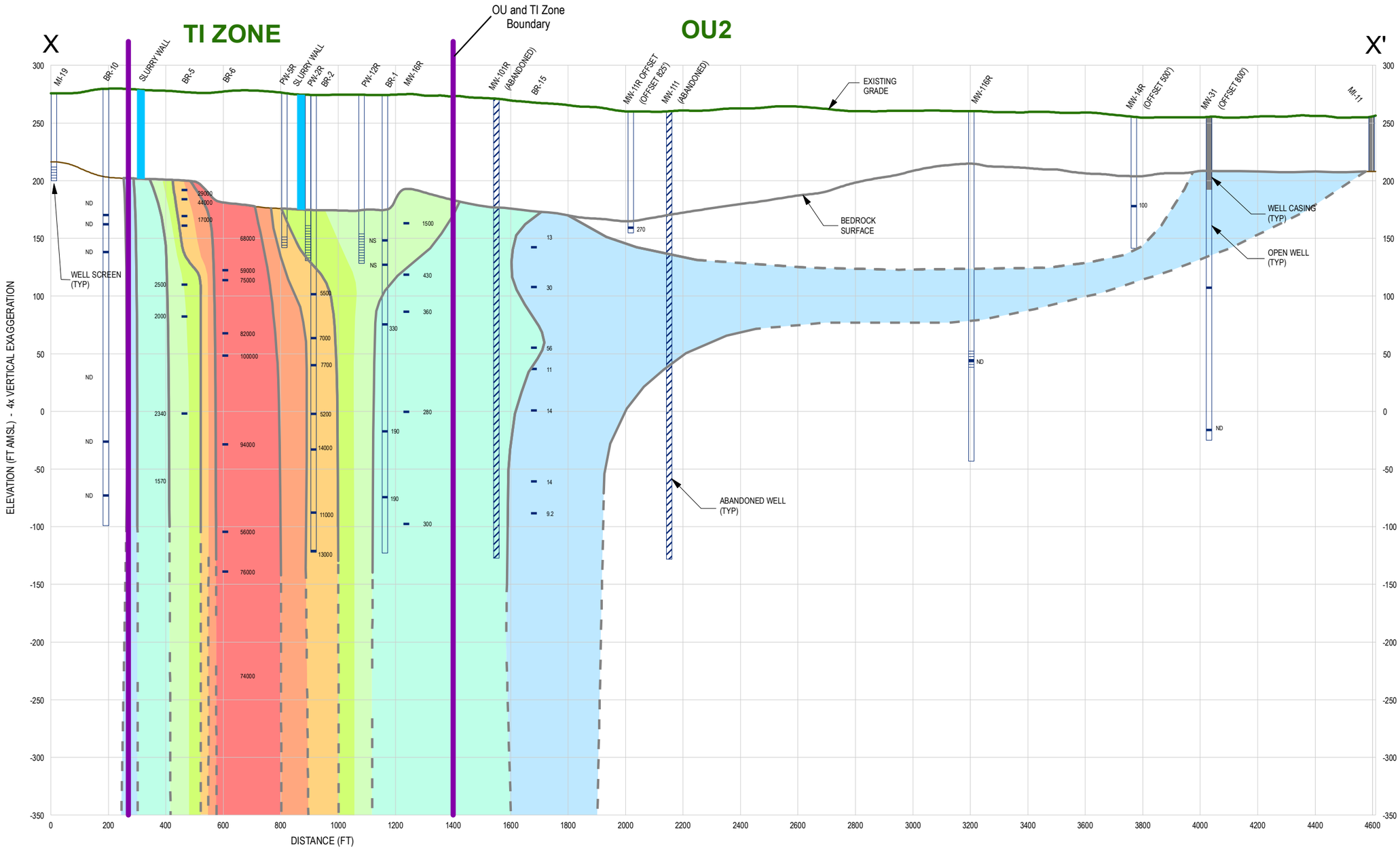
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

LATERAL EXTENT OF TI ZONE

TI WAIVER EVALUATION REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	DATE	W.O. NO.
AL	June 2015	20118.016.001
CHECKED	DATE	FIGURE NO.
JS	June 2015	4-1

FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig4-1_TI_Zone_plan_DRAFT_rev1.mxd 3:26:03 PM 06/04/2015 liua



LEGEND

PCE Concentrations

- > 50,000 ug/L
- 15,000-50,000 ug/L
- 5,000-15,000 ug/L
- 1,500-5,000 ug/L
- 500-1,500 ug/L
- 50-500 ug/L
- 5-50 ug/L

Features

- Slurry Wall
- Well Casing
- Abandoned Well

Boundaries

- TI Zone Boundary

Notes:
1. PCE values for each interval are the highest observed during the RI (2010-2013).
2. Values in the figure (e.g. 74,000) represent PCE concentrations in ug/L.
3. Plume is inferred where dashed lines are used.

ug/L - micrograms per liter
mg/L - milligrams per liter
DNAPL - Dense Non-Aqueous Phase Liquid
ND - Non-Detect
NS - No Sample
PBL - Physical Bound Levels
PCE - tetrachloroethene
TI - Technical Impracticability

WESTON SOLUTIONS CONCORD NEW HAMPSHIRE

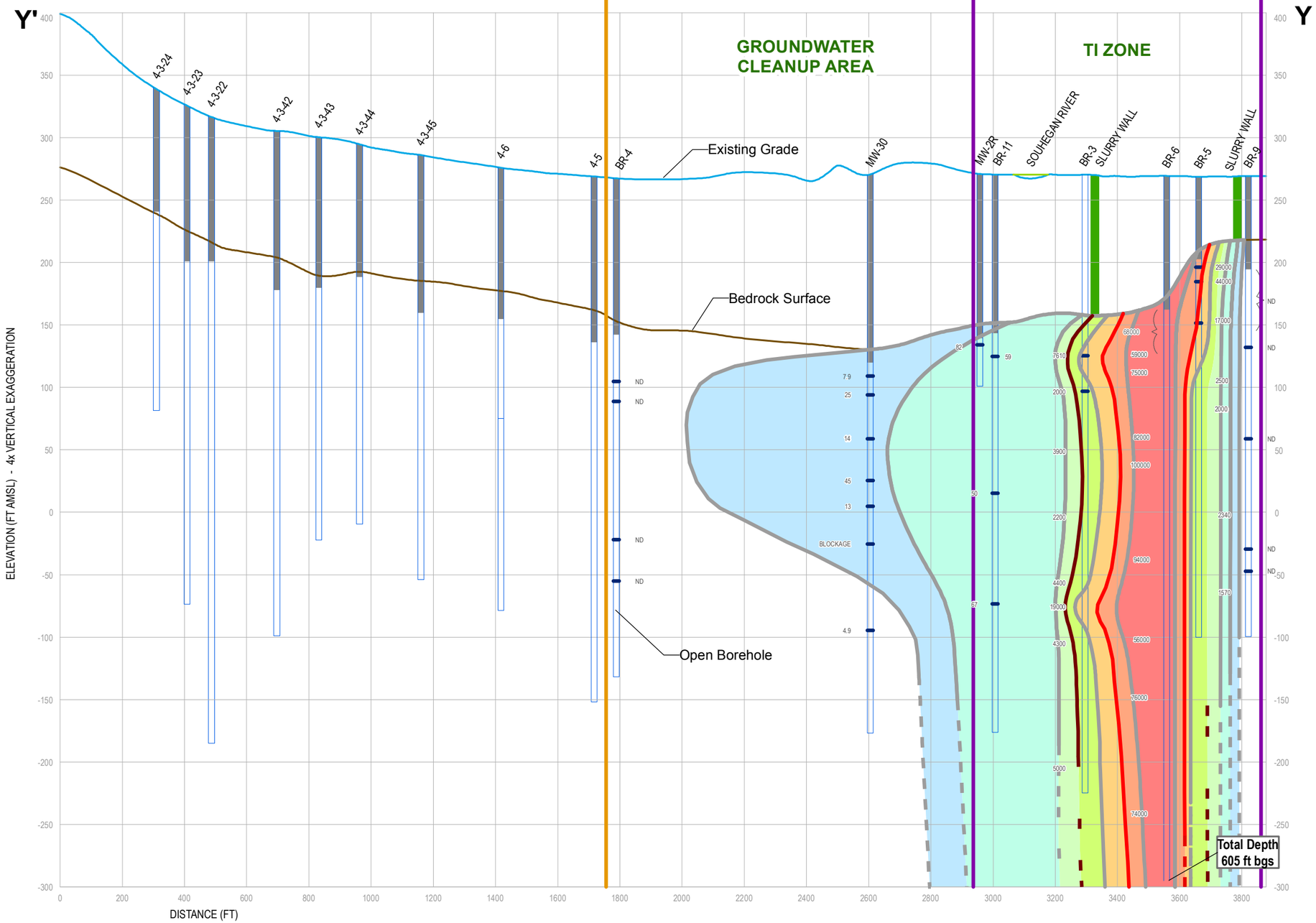
NEW HAMPSHIRE DEPARTMENT OF Environmental Services

VERTICAL EXTENT OF TI ZONE

TI WAIVER EVALUATION REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	CHECKED	DATE	W.O. NO.	FIGURE NO.
AL	JS	June 2015	20118.016.001	4-2

FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig4-2_BR_Xsec_X_PCE_2013_rev1.mxd 3:21:57 PM 06/04/2015 liua



LEGEND

PCE Concentrations in Bedrock Groundwater
- > 50,000 ug/L
- 15,000-50,000 ug/L
- 5,000-15,000 ug/L
- 1,500-5,000 ug/L
- 500-1,500 ug/L
- 50-500 ug/L
- 5-50 ug/L

Features
- SLURRY WALL
- Well Casing
- Open Boreholes

Boundaries
- TI Zone Boundary
- Groundwater Cleanup Area Boundary

Notes:
1. PCE values for each interval are the highest observed during the RI (2010-2013)
2. Values in the figure (e.g. 74,000) represent PCE concentrations in ug/L;
3. Plume is inferred where dashed lines are used.

ug/L - micrograms per liter
mg/L - milligram per liter
DNAPL - Dense Non-Aqueous Phase Liquid
ft bgs - feet below ground surface
ND - Non-Detect
NS - No Sample
PBL - Physical Bound Levels
PCE - tetrachloroethene
TI - Technical Impracticability

WESTON SOLUTIONS CONCORD NEW HAMPSHIRE

NEW HAMPSHIRE DEPARTMENT OF Environmental Services

VERTICAL EXTENT OF TI ZONE

TI WAIVER EVALUATION REPORT
SAVAGE MUNICIPAL WATER SUPPLY SUPERFUND SITE
MILFORD, NEW HAMPSHIRE

DRAWN	CHECKED	DATE	W.O. NO.	FIGURE NO.
AL	JS	Apr 2015	20118.016.001	4-3

FILE: C:\gis\GIS Project Files\SavageWell\TI_2015\MXD\Fig4-3_BR_Xsec_Y_PCE_2013_REV1.mxd 3:39:10 PM 06/04/2015 ltua

TABLES

Table 1-1

Groundwater Chemicals of Concern and Associated Interim Cleanup Levels and Ambient Groundwater Quality Standards

Savage Municipal Water Supply Well Superfund Site – OU1 & OU2
Milford, New Hampshire

Compound	AGQS/MCL µg/L (ppb)	ROD ICLs µg/L (ppb)
1,1-Dichloroethane	81 (1)	3,500
Trans 1,2-Dichoroethene (2)	100	100
Cis 1,2-Dichloroethene	70	Not Established
1,1-Dichloroethylene	7	7
Benzene	5	5
1,1,1-Trichloroethane	200	200
Methylene Chloride (4)	5	5
Tetrachloroethene	5	5
Trichloroethene	5	5
Antimony	6	3
Arsenic	10(3)	50
Beryllium	4	1
Chromium	100	100
Lead	15	15
Nickel	100	100

Notes:

(1) A Maximum Contaminant Level (MCL) has not been established for 1,1-dichloroethane. The concentration listed is the State Ambient Groundwater Quality Standard (AGQS) based on cancer potency factor of 9.1×10^{-2} (milligrams per kilograms per day)$^{-1}$ derived by State.

(2) Using the more restrictive MCL Goals for cis-1,2-dichloroethene (Cis = 70, Trans = 100).

(3) The arsenic standard was changed from 50 parts per billion (ppb) to 10 ppb by the United States Environmental Protection Agency in 2001. The New Hampshire Department of Environmental Services has also changed the AGQS (MCL) to 10 ppb per RSA 485 C:6 (MCL) and Env-Ws 316.01.

(4) Methylene Chloride is listed under the AGQS as dichloromethane and has a limit of 5 µg/L.

ROD = Record of Decision
µg/L = micrograms per liter
ICL = Interim Cleanup Levels

Table 4-1

Summary of OU-1 Groundwater Treatment Plant Volatile Organic Compound Mass Removal Savage Municipal Water Supply Well Superfund Site Milford, New Hampshire

	Mass VOC removed (lbs)		
Year	**Groundwater**	**SVE**	**Total (lbs)**
1999	194	0.4	194.4
2000	215	470	685
2001*	176	37	213
2002	256	22	278
2003	249	0	249
2004	172	35	207
2005	110	0	110
2006	151	112	263
2007	220	136	356
2008	101	0	101
2009	56	0	56
2010	36.1	0	36.1
2011	28.9	0	28.9
2012	49.9	0	49.9
2013	50.7	0	50.7
2014	47.9	0	47.9
Totals	2113.5	812	2925.9

Notes:

* Data missing for SVE from May 2001 to Oct 2001

VOC - volatile organic compounds

gal - gallons

lbs - pounds

SVE - soil vapor extraction

Table 4-2

General Response Actions and Technologies
Savage Municipal Water Supply Well Superfund Site
Technical Impracticability Evaluation Report

General Response Action	Technology	Process Options
Natural Attenuation	Monitored Natural Attentuation	Monitoring Wells/Analysis of Groundwater Samples for Contaminants and Parameters indicative of Degradation of Contaminants
Hydraulic Containment	Groundwater Extraction and Treatment	Air Stripping/Carbon Vapor Treatment
		Air Stripping/Cat-Ox Vapor Treatment
		Advanced Oxidation
Physical Removal	Excavation	Excavation of overburden; blasting and removal of bedrock
In Situ Treatment	In Situ Chemical Oxidation (ISCO)	Sodium Permanganate
		Potassium Permanganate
		Peroxide/Ozone
		Sodium Persulfate
	In Situ Chemical Reduction (ISCR)	Nano or Micro-Scale Zero Valent Iron
		Calcium Polysulfate
	In Situ Biological Treatment (ISB)	Commercially available oil or carbohydrate based amendments.
	In Situ Thermal Treatment	Thermal Conductivity Heating (TCH)
		Electrical Resistivity Heating
		Steam Injection
		Radio Frequency Heating (RFH)
	Permeable Reactive Barrier	ZVI PRB
		BioWall - mulch
		ZVI + carbon

Table 4-3

Remedial Technology Screening Results
Savage Municipal Water Supply Well Superfund Site
Technical Impracticability Evaluation Report

Technology	Description	Effectiveness	Implementability	Timeliness
Monitored Natural Attenuation (MNA)	Rely on natural processes to reduce contaminant mass.	Natural processes would eventually be able to reduce the observed levels of contamination to below the PRGs/PSs.	This technology can be easily implemented at the Site.	Preliminary modeling performed using EPA's BIOCHLOR software suggests that natural attenuation of the observed PCE concentrations down to PRGs/PSs would require several thousand years.
Pump and Treat (P&T)	Extraction of contaminated groundwater via recovery wells followed by ex situ treatment.	Extraction would be an effective technology to capture and remove the contaminated groundwater, although it would not be highly effective at removing the presumed DNAPL. Because DNAPL in fractured bedrock is nearly impossible to locate, this technology would have to rely on dissolution of the DNAPL into the groundwater following removal/treatment of the dissolved component.	This technology is already being implemented in the overburden aquifer in OU1 and the existing system could be modified to contain the bedrock contamination as well.	Based on a constant mass removal rate of roughly 50 lbs of PCE per year, and the minimum estimated remaining mass, this technology would require at least 600 years to achieve the PRGs/PSs, and likely much longer because the mass removal rate will slowly decrease over time.
Physical Removal	Excavation of overburden soil; blast and remove impacted bedrock.	Excavation of the overburden and bedrock skeleton of the impacted aquifer would be unable to remove all of the presumed DNAPL in the bedrock because the blasting needed would open new fractures and allow the DNAPL to migrate away from the excavation area.	It is not currently feasible to remove 100 ft of highly-permeable overburden and then blast/remove an additional 600 ft (or greater) of competent crystalline bedrock.	Because this technology is not implementable, timeliness is not assessed.
In Situ Chemical Oxidation (ISCO)	Injection of chemical oxidant into the aquifer to destroy the contaminants in place.	ISCO relies on direct contact between the oxidant and the contamination. The inferred presence of DNAPL within dead-end and micro fractures in the deep bedrock limits the ability of the oxidant to contact (and thus destroy) the contamination. The oxidant can only be injected into larger hydraulically-active fractures but the majority of the contaminant mass is believed to be located within the non-active fractures. This incongruence is characteristic of bedrock remediation.	The technology exists to drill injection wells in the overburden and bedrock to the depths required. Oxidants capable of destroying the observed contaminants are readily available.	The heterogeneity of the fractured bedrock aquifer will tend to isolate DNAPL and dissolved contamination from the ISCO reagent and thereby limit the amount of contact between the two. The effect of this will be that rebound will occur after each injection event, although slightly less each time, and the total time required to attain the PRGs/PSs will be extensive. This technology would not be able to restore the groundwater within a reasonable timeframe.

Table 4-3

Remedial Technology Screening Results
Savage Municipal Water Supply Well Superfund Site
Technical Impracticability Evaluation Report

Technology	Description	Effectiveness	Implementability	Timeliness
In Situ Chemical Reduction (ISCR)	Injection of chemical reductant into the aquifer to destroy the contaminants in place.	ISCR relies on direct contact between the reducing agent and the contamination. The inferred presence of DNAPL within dead-end and micro fractures in the deep bedrock limits the ability of the reducing agent to contact (and thus destroy) the contamination. The reducing agent can only be injected into larger hydraulically-active fractures but the majority of the contaminant mass is believed to be located within the non-active fractures. This incongruence is characteristic of bedrock remediation.	The technology exists to drill injection wells in the overburden and bedrock to the depths required. Chemical reductant capable of destroying the observed contaminants are readily available.	The heterogeneity of the fractured bedrock aquifer will tend to isolate DNAPL and dissolved contamination from the ISCR reagent and thereby limit the amount of contact between the two. The effect of this will be that rebound will occur after each injection event, although slightly less each time, and the total time required to attain the PRGs/PSs will be extensive. This technology would not be able to restore the groundwater within a reasonable timeframe.
In Situ Biological Treatment (ISB)	Injection of amendments to stimulate natural biological degradation of contaminants and accelerate the cleanup.	The inferred presence of DNAPL within dead-end and micro fractures in the deep bedrock limits the ability of ISB to destroy the contamination. Biodegradation of DNAPL in fractures has not been observed. As a result, this technology would have to rely on dissolution of the DNAPL into the groundwater before it can be treated, which is an extemely slow process. Further, the amendments can only be injected into larger hydraulically-active fractures but the majority of the contaminant mass is located within the non-active fractures. This incongruence is characteristic of bedrock remediation.	The technology exists to drill injection wells in the overburden and bedrock to the depths required. Amendments capable of stimulating natural biodegradation of the observed contaminants are readily available.	The heterogeneity of the fractured bedrock aquifer will tend to isolate DNAPL and dissolved contamination from the ISB amendment and thereby limit the amount of contact between the two. The effect of this will be that rebound will occur after each injection event, although slightly less each time, and the time required to attain the PRGs/PSs will be extensive. This technology would not be able to restore the groundwater within a reasonable timeframe.
In Situ Thermal Treatment (ISTT)	Groundwater and soil/bedrock are heated to volatilize contaminants so they can be collected and destroyed.	If it could be implemented at the depths needed and over the entire area, ISTT would possibly be able to achieve the PRGs/PSs.	Thermal treatment of a large volume of bedrock extending to a depth of more than 500 ft has never been performed. It is unknown whether it is feasible to heat such a large volume of material to the termperatures needed to volatilize the DNAPL. Further, it is unclear whether it would be possible to collect the vapors from the observed depths that are impacted at this Site.	If implementable, this technology would likely be the fastest way to achieve the PRGs/PSs, possibly within as little as 5 years.
Permeable Reactive Barrier (PRB)	As contaminated groundwater flows through the PRB, contaminants are removed by stimulated biological degradation or abiotic reactions with the reactive materials inside the treatment barrier.	Although a PRB will remove or destroy contaminant mass, it relies on groundwater flow to deliver contamination to it. As a result it is more commonly used to manage migration of contamination to a sensitive receptor and not strictly for contaminant mass reduction.	A PRB installed in deep bedrock would need to consist of injected amendments of the type described above. As such, it would be subject to the same constraints as the other injection technologies. The inherent heterogeniety of the bedrock would present a challenge to installing a continuous PRB without gaps.	Because a PRB is a management of migration technology that would rely on groundwater flow to transport dissolved contamination to the barrier for treatment, it would be limited by dissolution of the DNAPL into hydraulically-active fractures which is an extremely slow process. This technology would not be able to restore the groundwater within a reasonable timeframe.

APPENDIX A

GROUNDWATER MODELING USING BIOCHLOR

Appendix A

Groundwater Modelling Using BIOCHLOR

Savage Municipal Water Supply Well Superfund Site OU-1

I. Model Description

Groundwater modeling was conducted using BIOCHLOR v2.2, a USEPA natural attenuation decision support system. BIOCHLOR is a screening model that simulates remediation by natural attenuation of dissolved solvents at chlorinated solvent release sites.

BIOCHLOR can be used to simulate solute transport without decay and solute transport with biodegradation modeled as a sequential first-order process within one or two different reaction zones. The software, programmed in the Microsoft Excel spreadsheet environment and based on the Domenico analytical solute transport model, has the ability to simulate 1-D advection, 3-D dispersion, linear adsorption, and biotransformation via reductive dechlorination (the dominant biotransformation process at most chlorinated solvent sites). Reductive dechlorination is assumed to occur under anaerobic conditions and dissolved solvent degradation is assumed to follow a sequential first-order decay process. BIOCHLOR v.2.2 (June 2002) was downloaded from the U.S. EPA website (http://www2.epa.gov/water-research/biochlor-natural-attenuation-decision-support-system).

II. Modeling Objective

The objective of the modeling is to obtain an order of magnitude (OOM) estimate of cleanup timeframe for the downgradient bedrock groundwater plume when an upgradient source is cut off, to assist in assessing whether Monitored Natural Attenuation (MNA) would be a reasonable remedial alternative for portions of OU1 at the Savage Municipal Water Supply Well Superfund Site in Milford, NH (Site).

BIOCHLOR takes into account dilution, dispersion, adsorption/matrix diffusion, and degradation of PCE and a sequential first-order decay process to produce daughter products including TCE and cis-DCE. Important assumptions that serve as the basis of using the BIOCHLOR model include homogeneous matrix, steady and uniform groundwater flow, instantaneous linear sorption equilibration, continuous source or source truncation, and first order decay kinetics. Although some of these assumptions (primarily homogeneous matrix, uniform flow, and instantaneous sorption equilibration) are not satisfied substantially at this site, thus leading to significant uncertainty in the interpretation of the modeling results, the model covers the fundamental physical mechanisms governing the fate and transport of contaminants and approximates them with simplified processes that can generate reasonable results of OOM precision with relatively minor effort. It was deemed appropriate to serve as

one “line of evidence” to evaluate the groundwater plume migration and cleanup process in the bedrock for this Site

III. Modeling Approach and Considerations

Model Setup

The model was set up for 1-D simulation of the fate and transport of chlorinated solvents along the centerline of the plume in the deep bedrock at the Site. The hydrology of the site has been described in Section 1 of the 2015 FS. The groundwater flow direction in the deep bedrock is interpreted to be primarily to the north-northeast, along strike of predominate fracture set. Four bedrock wells located in this direction (BR-06, BR-03, BR-11, and MW-30) were interpreted to be located on the plume centerline and selected to be used as observation locations for the model.

Three compounds (PCE, TCE, and total DCE) were modeled by BIOCHLOR. The observed concentrations of cis-1,2-DCE were used in the model to represent total DCE concentrations because trans-1,2-DCE was not detected and 1,1-DCE was detected only once and at low level in the four wells. VC was not detected in these wells and therefore was not included in the model. The average concentration for each compound (PCE, TCE, and DCE) in each monitoring well based on not more than three sampling rounds was used to serve as the current condition. The model was then calibrated to match this current condition, assuming that the mass release from the DNAPL source began at an estimated starting time and continued to the present. Various model input parameters were varied until a good fit with the current condition was achieved. The ‘least square residual method’ was applied to determine the best fit of the simulation results to the field data. Once it was calibrated, the model was rerun using the current condition as the initial condition, but with complete cutoff of the source to simulate implementation of a source-control remedy in OU1. The calibrated model was used to estimate the timeframe for the downgradient plume to be cleaned up under natural attenuation.

Source Setup

Because the PCE concentration detected at BR-06 was very close to its water solubility, it was assumed that a continuous source (with DNAPL likely) was present for the bedrock plume in the vicinity of this location. The concentrations of PCE, TCE, and DCE at BR-06 were used as the source concentrations. The source width and thickness were estimated based on data presented in the 2014 RI.

When a TI Waiver Zone is established (and the downgradient area is the groundwater cleanup zone (GCZ)), no mass flux is assumed crossing the TI Waiver Zone boundary to release into the GCZ, which is the basis of the cleanup condition to be modeled. Under the cleanup condition, no source and no mass flux from upgradient was assumed.

Biotransformation

First order decay kinetics were assumed to be present based on the observation that while the concentrations of PCE, TCE, and DCE (denoted as [PCE], [TCE], and [DCE]) decrease along the distance away from the source, the ratios of daughter products to parent products (i.e. [TCE]/[PCE] and [1,2-DCE]/[TCE]) increase. Because no site-specific degradation rates are available, the range of the suggested decay rates provided in the BIOCHLOR manual was used as the limits when calibrating the model to fit the observed data.

Retardation Modeling

In fractured bedrock where the presence of organic matters is at minimum, the retardation of the contaminants is primarily controlled by mass retained in the secondary fractures and rock matrix and the mass exchange rate (through contaminant diffusion) between immobile groundwater in secondary fractures/rock matrix and mobile groundwater in primary fractures. This retardation mechanism is similar to the sorption mechanism in unconsolidated media where contaminants would be retained on the solid phase (organic carbon and surrounding water membrane) and exchanged between mobile groundwater and the immobile phase. The sorption function included in BIOCHLOR was used for this Site to simulate the retardation phenomenon in fractured bedrock. Because BIOCHLOR uses a single retardation factor for simulation of multiple compounds, the value of the retardation factor was adjusted within a reasonable range and applied to all compounds rather than using a unique value for each compound.

Model Calibration

The BIOCHLOR model was calibrated to fit the data at the observation locations BR-06, BR-03, BR-11, and MW-30. Calibration and initial sensitivity runs determined that:

(1) Groundwater flow velocity and the retardation factor were found to be coupled in a proportional manner, i.e. high groundwater velocity required a large retardation factor to accurately match the observed conditions; and low velocity simulations required a small retardation factor to get the model results to fit the observed data.
(2) The coupled groundwater flow velocity and retardation factor were the most sensitive parameters for model calibration and variation of their values significantly affects the estimated cleanup timeframe. The model results are highly sensitive to these two parameters.
(3) Multiple solutions with different groundwater flow velocity/retardation factors were used to calibrate the model.
(4) The other input parameters, including dispersitivity and decay rate (within a reasonable range) were much less sensitive for model calibration and variation of their values had a much smaller impact on the estimated cleanup timeframe.

To demonstrate the impact of the model sensitivity to groundwater velocity/contaminant retardation, three scenarios are presented that all resulted in an acceptable model calibration. The three scenarios correspond to (S1) high velocity/high retardation, (S2) medium velocity/medium retardation, and (S3)

low velocity/low retardation conditions. To gain model calibration under these scenarios, the decay rate was varied slightly within the referenced range and a single value was used for dispersitivity. In deep fractured bedrock, the true groundwater flow velocity under ambient flow conditions is highly heterogeneous; with high velocity in larger water-bearing fractures and very low velocity in secondary fractures. Therefore, there is not one clear scenario that best approximates actual conditions. However, it is likely that the actual condition falls within the limits described by these scenarios.

IV. Groundwater Model Input Parameters

BIOCHLOR model requires general input parameters about the model domain, simulation time, and calibration field data as well as scenario-specific parameters to describe the fate and transport of contaminants. The general input parameters are provided in Table 1 and scenario-specific input parameters are provided in Table 2. When applicable, the values of the parameter, the data source or reference for the value selection, and application notes are provided.

V. Groundwater Model Outputs

Concentrations of PCE, TCE, and DCE were calculated for regular locations along the 1-D transect by BIOCHLOR based on the inputs provided. The spacing of the selected locations for these simulations was 150 ft. The concentrations were interpolated from the regular locations (150-ft intervals along the transect) to the observation locations at BR-03, BR-11, and MW-30.

Table 3 presents the model-calculated concentrations of PCE, TCE, and DCE at the three observation wells (BR-3, BR-11, and MW-30) over the 400 years simulated. The results for the three scenarios are shown together for comparison. It should be noted that the data for the first 40 years were the results generated during calibration and represent the expansion of the plume from the release area. Year 40 represents the near-present conditions. The concentrations after Year 40 represent the predicted natural attenuation of the plume assuming that the source has been isolated and the contribution to downgradient areas has been eliminated.

VI. Conclusion

The change in the simulated PCE concentrations over time at BR-03 under the three scenarios is illustrated in Figure 1. BR-03 is the location with the highest starting concentration after source isolation and therefore represents the worst-case cleanup scenario.

Based on the results presented in Table 3 and Figure 1, the following conclusions are drawn with regard to the cleanup timeframe for the plume.

1. The current plume has not reached steady state and may continue to expand downgradient if the source is not removed or contained.
2. The cleanup timeframe for PCE in the downgradient plume after the source has been removed or isolated is likely on the order of 240 years (under Scenario 1) or greater (Scenarios 2 and 3);

3. After about 100 years from source truncation, the downgradient peak PCE concentration would be reduced by approximately 35-times under Scenario 1, the most favorable scenario among the three scenarios evaluated. The results suggested that in order to get downgradient PCE concentrations below cleanup standards within 100 years (considering this a "reasonable" timeframe), the source control remedy needs to include all PCE concentrations that are about 35 times the cleanup goal (5 ug/L), so above roughly 175 ug/L.

It should be noted that, as described above, the modeling approach used to simulate the observed conditions and the selection of the modeling input parameters contains significant uncertainty. As a consequence of this uncertainty, the modeling results represent only a rough OOM estimate.

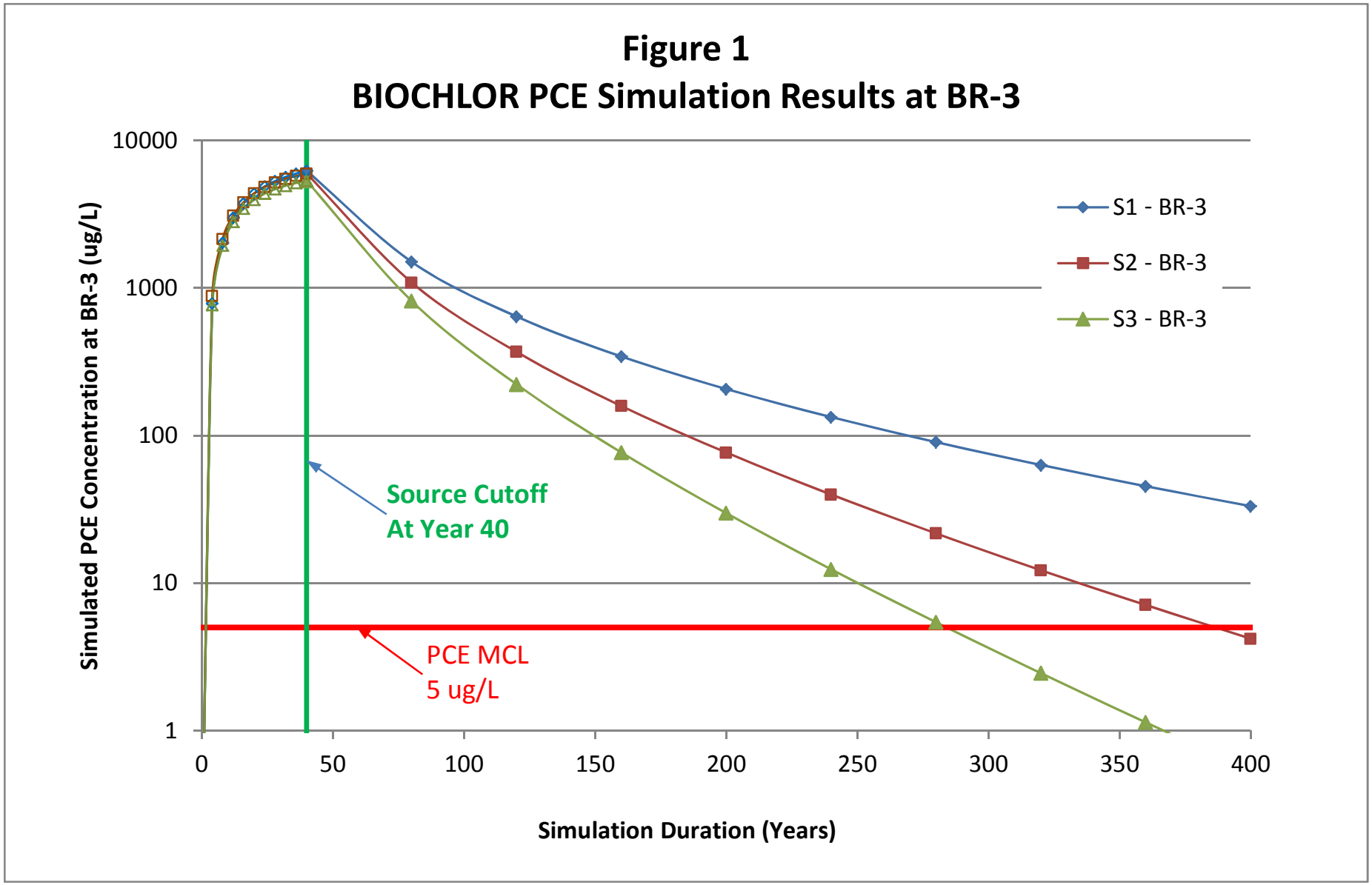


Figure 1
BIOCHLOR PCE Simulation Results at BR-3
Simulated PCE Concentration at BR-3 (ug/L)
10000
1000
100
10
1
0
50
100
150
200
250
300
350
400
Simulation Duration (Years)
S1 - BR-3
S2 - BR-3
S3 - BR-3
Source Cutoff
At Year 40
PCE MCL
5 ug/L

Table 1
General Model Setup Parameters

Model Parameter	Values	Units	Site Specific	Data Source/Reference
Source Area Dimensions				
Width	200	ft	Y	2014 Remedial Investigation
Height	500	ft	Y	2014 Remedial Investigation
Simulation Area				
Modeled Area Width	2000	ft		Set large to accommodate 1D simulation
Modeled Area Length	1500	ft	Y	2014 Remedial Investigation
Simulation Time				
Calibration	40	years	Y	Assumed
Prediction	360	years	Y	
Source Concentration (BR-06)				
PCE	67.6	mg/L	Y	2014 Remedial Investigation
TCE	3.47	mg/L	Y	2014 Remedial Investigation
DCE	0.058	mg/L	Y	2014 Remedial Investigation
Field Data For Calibration (BR-03)				
PCE	4.07	mg/L	Y	2014 Remedial Investigation
TCE	0.79	mg/L	Y	2014 Remedial Investigation
DCE	0.019	mg/L	Y	2014 Remedial Investigation
Field Data For Calibration (BR-11)				
PCE	0.069	mg/L	Y	2014 Remedial Investigation
TCE	0.044	mg/L	Y	2014 Remedial Investigation
DCE	0.005	mg/L	Y	2014 Remedial Investigation
Field Data For Calibration (MW-30)				
PCE	0.013	mg/L	Y	2014 Remedial Investigation
TCE	0.01	mg/L	Y	2014 Remedial Investigation
DCE		mg/L	Y	2014 Remedial Investigation

Table 2
Scenario Specific Input Parameters

Model Parameter	Scenario-1 Values	Scenario-2 Values	Scenario-3 Values	Units	Site Specific	Data Source/Reference	Notes
Hydraulics							
Hydraulic Gradient	0.006	0.006	0.006	ft/ft	Y	2014 Remedial Investigation	
Hydraulic Conductivity	1.10E-04	2.80E-05	1.00E-05	cm/s	Y	2014 Remedial Investigation	
Effective Porosity	0.002	0.002	0.005	--	Y	2014 Remedial Investigation	
Seepage Velocity	334	87	12			BIOCHLOR calculation	Primary factor for sensitivity analysis
Dispersivity alpha x	1000	1000	1000	--	N	Lipson, Kueper, Gefell; 2005	High dispersivity to fit field data;
Dispersivity alpha x/alpha y	1	1	1	--	N	Lipson, Kueper, Gefell; 2006	High dispersivity to fit field data;
Adsorption/Matrix Diffusion							
Retardation Factor	333	87	12		N		Adjusted to fit field data;
Biodegradation Rates							
PCE to TCE	1.16	0.693	0.173	1/yr	N	BIOCHLOR manual	Adjusted to fit field data;
TCE to DCE	0.866	0.139	0.087	1/yr	N	BIOCHLOR manual	Adjusted to fit field data;
DCE to VC	0.693	0.347	2.31	1/yr	N	BIOCHLOR manual	Adjusted to fit field data;

Table 3
BIOCHLOR Modeling Results at Observation Locations

Scenarios		Scenario 1 High Velocity, High Retardation			Scenario 2 Medium Velocity, Medium Retardation			Scenario 3 Low Velocity, Low Retardation		
Observation Locations		BR-03	BR-11	MW-30	BR-03	BR-11	MW-30	BR-03	BR-11	MW-30
Distance (ft from source)		223	547	915	223	547	915	223	547	915
chemicals	Years	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
PCE	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	4	7.85E+02	2.49E-03	0.00E+00	8.82E+02	6.33E-03	0.00E+00	7.67E+02	2.50E-03	0.00E+00
	8	2.03E+03	1.63E+00	1.73E-08	2.15E+03	2.59E+00	1.08E-07	1.93E+03	1.55E+00	1.74E-08
	12	2.99E+03	1.57E+01	7.80E-05	3.09E+03	2.12E+01	2.59E-04	2.80E+03	1.43E+01	7.31E-05
	16	3.75E+03	5.21E+01	5.40E-03	3.81E+03	6.44E+01	1.29E-02	3.45E+03	4.58E+01	4.81E-03
	20	4.36E+03	1.12E+02	7.00E-02	4.37E+03	1.30E+02	1.37E-01	3.96E+03	9.51E+01	5.95E-02
	24	4.86E+03	1.90E+02	3.91E-01	4.82E+03	2.13E+02	6.65E-01	4.35E+03	1.57E+02	3.18E-01
	28	5.28E+03	2.83E+02	1.34E+00	5.18E+03	3.06E+02	2.06E+00	4.67E+03	2.28E+02	1.05E+00
	32	5.64E+03	3.85E+02	3.41E+00	5.48E+03	4.04E+02	4.82E+00	4.93E+03	3.02E+02	2.57E+00
	36	5.95E+03	4.92E+02	7.07E+00	5.74E+03	5.03E+02	9.34E+00	5.14E+03	3.76E+02	5.14E+00
	40	6.21E+03	6.02E+02	1.27E+01	5.95E+03	6.02E+02	1.58E+01	5.31E+03	4.49E+02	8.93E+00
	80	1.50E+03	9.81E+02	1.72E+02	1.09E+03	7.56E+02	1.50E+02	8.16E+02	5.22E+02	8.73E+01
	120	6.40E+02	6.46E+02	2.69E+02	3.69E+02	3.87E+02	1.76E+02	2.22E+02	2.23E+02	9.16E+01
	160	3.43E+02	4.14E+02	2.53E+02	1.58E+02	1.97E+02	1.28E+02	7.67E+01	9.23E+01	5.60E+01
	200	2.06E+02	2.75E+02	2.07E+02	7.66E+01	1.05E+02	8.29E+01	2.98E+01	3.96E+01	2.97E+01
	240	1.33E+02	1.89E+02	1.62E+02	3.98E+01	5.75E+01	5.16E+01	1.24E+01	1.76E+01	1.51E+01
	280	9.02E+01	1.33E+02	1.26E+02	2.17E+01	3.26E+01	3.19E+01	5.42E+00	8.01E+00	7.55E+00
	320	6.31E+01	9.62E+01	9.71E+01	1.22E+01	1.89E+01	1.97E+01	2.45E+00	3.74E+00	3.77E+00
	360	4.53E+01	7.07E+01	7.51E+01	7.10E+00	1.12E+01	1.22E+01	1.14E+00	1.78E+00	1.89E+00
	400	3.32E+01	5.27E+01	5.83E+01	4.20E+00	6.70E+00	7.60E+00	5.39E-01	8.57E-01	9.48E-01

Table 3
BIOCHLOR Modeling Results at Observation Locations

Scenarios		Scenario 1 High Velocity, High Retardation			Scenario 2 Medium Velocity, Medium Retardation			Scenario 3 Low Velocity, Low Retardation		
Observation Locations		BR-03	BR-11	MW-30	BR-03	BR-11	MW-30	BR-03	BR-11	MW-30
Distance (ft from source)		223	547	915	223	547	915	223	547	915
chemicals	Years	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
TCE	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	4	4.64E+01	1.54E-04	0.00E+00	6.31E+01	5.01E-04	0.00E+00	6.45E+01	2.40E-04	0.00E+00
	8	1.31E+02	1.16E-01	1.26E-09	1.84E+02	2.70E-01	1.18E-08	2.07E+02	2.11E-01	2.52E-09
	12	2.07E+02	1.25E+00	6.49E-06	3.04E+02	2.71E+00	3.58E-05	3.57E+02	2.50E+00	1.39E-05
	16	2.76E+02	4.58E+00	5.04E-04	4.19E+02	9.70E+00	2.16E-03	5.03E+02	9.67E+00	1.14E-03
	20	3.38E+02	1.07E+01	7.22E-03	5.26E+02	2.25E+01	2.67E-02	6.42E+02	2.34E+01	1.68E-02
	24	3.95E+02	1.96E+01	4.40E-02	6.28E+02	4.12E+01	1.48E-01	7.73E+02	4.42E+01	1.04E-01
	28	4.48E+02	3.12E+01	1.64E-01	7.23E+02	6.56E+01	5.17E-01	8.96E+02	7.14E+01	3.92E-01
	32	4.96E+02	4.51E+01	4.48E-01	8.13E+02	9.48E+01	1.34E+00	1.01E+03	1.04E+02	1.07E+00
	36	5.42E+02	6.10E+01	9.91E-01	8.97E+02	1.28E+02	2.86E+00	1.12E+03	1.42E+02	2.38E+00
	40	5.84E+02	7.86E+01	1.89E+00	9.77E+02	1.65E+02	5.30E+00	1.22E+03	1.83E+02	4.53E+00
	80	3.22E+02	2.17E+02	4.03E+01	6.12E+02	4.43E+02	9.54E+01	7.00E+02	4.69E+02	8.59E+01
	120	2.16E+02	2.19E+02	9.28E+01	4.00E+02	4.25E+02	1.98E+02	3.77E+02	3.82E+02	1.61E+02
	160	1.59E+02	1.92E+02	1.18E+02	2.81E+02	3.51E+02	2.31E+02	2.15E+02	2.60E+02	1.59E+02
	200	1.22E+02	1.63E+02	1.23E+02	2.06E+02	2.81E+02	2.24E+02	1.27E+02	1.70E+02	1.28E+02
	240	9.67E+01	1.37E+02	1.18E+02	1.55E+02	2.24E+02	2.02E+02	7.74E+01	1.10E+02	9.46E+01
	280	7.79E+01	1.15E+02	1.09E+02	1.19E+02	1.79E+02	1.76E+02	4.80E+01	7.10E+01	6.70E+01
	320	6.35E+01	9.69E+01	9.78E+01	9.31E+01	1.44E+02	1.50E+02	3.02E+01	4.61E+01	4.66E+01
	360	5.23E+01	8.17E+01	8.68E+01	7.37E+01	1.17E+02	1.28E+02	1.93E+01	3.02E+01	3.21E+01
	400	4.34E+01	6.90E+01	7.64E+01	5.91E+01	9.51E+01	1.08E+02	1.25E+01	1.98E+01	2.20E+01

Table 3
BIOCHLOR Modeling Results at Observation Locations

Scenarios		Scenario 1 High Velocity, High Retardation			Scenario 2 Medium Velocity, Medium Retardation			Scenario 3 Low Velocity, Low Retardation		
Observation Locations		BR-03	BR-11	MW-30	BR-03	BR-11	MW-30	BR-03	BR-11	MW-30
Distance (ft from source)		223	547	915	223	547	915	223	547	915
chemicals	Years	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
DCE	0	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	4	9.08E-01	3.16E-06	0.00E+00	9.61E-01	7.53E-06	0.00E+00	1.02E+00	3.85E-06	0.00E+00
	8	2.83E+00	2.78E-03	1.00E-02	2.81E+00	4.15E-03	1.00E-02	3.48E+00	3.76E-03	4.35E-11
	12	4.88E+00	3.38E-02	1.83E-07	4.76E+00	4.43E-02	6.00E-07	6.36E+00	4.83E-02	2.77E-07
	16	6.99E+00	1.37E-01	1.60E-05	6.81E+00	1.71E-01	3.97E-05	9.42E+00	1.99E-01	2.42E-05
	20	9.16E+00	3.50E-01	2.54E-04	8.96E+00	4.27E-01	5.38E-04	1.25E+01	5.06E-01	3.75E-04
	24	1.14E+01	6.98E-01	1.70E-03	1.12E+01	8.43E-01	3.26E-03	1.56E+01	9.88E-01	2.41E-03
	28	1.36E+01	1.19E+00	6.87E-03	1.35E+01	1.44E+00	1.23E-02	1.85E+01	1.64E+00	9.33E-03
	32	1.59E+01	1.85E+00	2.02E-02	1.59E+01	2.22E+00	3.46E-02	2.13E+01	2.45E+00	2.61E-02
	36	1.82E+01	2.65E+00	4.79E-02	1.84E+01	3.20E+00	7.91E-02	2.40E+01	3.40E+00	5.89E-02
	40	2.05E+01	3.62E+00	9.74E-02	2.09E+01	4.36E+00	1.57E-01	2.65E+01	4.45E+00	1.14E-01
	80	2.42E+01	1.68E+01	3.30E+00	2.94E+01	2.19E+01	5.00E+00	1.86E+01	1.25E+01	2.31E+00
	120	2.51E+01	2.58E+01	1.11E+01	3.19E+01	3.41E+01	1.61E+01	1.04E+01	1.06E+01	4.45E+00
	160	2.52E+01	3.06E+01	1.89E+01	3.20E+01	4.01E+01	2.65E+01	6.04E+00	7.30E+00	4.47E+00
	200	2.47E+01	3.29E+01	2.50E+01	3.11E+01	4.25E+01	3.40E+01	3.60E+00	4.80E+00	3.63E+00
	240	2.38E+01	3.37E+01	2.91E+01	2.95E+01	4.27E+01	3.85E+01	2.20E+00	3.12E+00	2.69E+00
	280	2.27E+01	3.35E+01	3.17E+01	2.77E+01	4.17E+01	4.09E+01	1.37E+00	2.02E+00	1.91E+00
	320	2.14E+01	3.27E+01	3.30E+01	2.57E+01	3.99E+01	4.16E+01	8.64E-01	1.32E+00	1.33E+00
	360	2.01E+01	3.14E+01	3.34E+01	2.38E+01	3.77E+01	4.12E+01	5.53E-01	8.63E-01	9.18E-01
	400	1.88E+01	2.99E+01	3.31E+01	2.19E+01	3.53E+01	4.01E+01	3.57E-01	5.69E-01	6.29E-01

Notes:
1. The data of the first 40 years show the result of calibration when there was a continuous source;
2. The data of the later 60 years show the result of cleanup progress through natural attenuation when the source is cut off;

APPENDIX B

DESCRIPTION OF REMEDIAL TECHNOLOGIES

APPENDIX B

Description of Remedial Technologies

I. INTRODUCTION

This summary identifies remedial technologies and process options that may be applicable and potentially effective to achieve the remedial action objectives at OU1 of the Savage Municipal Water Supply Well Superfund Site.

II. DESCRIPTION OF TREATMENT AND DISPOSAL TECHNOLOGIES

Based on the site conditions, the following technologies are not considered implementable and not evaluated further for the reasons specified.

- Permeable Reactive Barrier: the deep bedrock trenching necessary to construct a barrier is not practical;
- Excavation and Disposal of contaminated media: it is not practical to excavate a large volume of bedrock.

The following technologies are retained for further evaluation.

- Pump and Treat (also serving as hydraulic control or recirculation to combine with other technologies)
- Physical Barrier
- Natural Attenuation
- In situ Thermal Treatment
- In situ Chemical Treatment
- In situ Biological Treatment

These technologies and their process options are described below. A few representative processes for each technology type are discussed.

III. DESCRIPTION OF REMEDIAL TECHNOLOGIES AND PROCESS OPTIONS

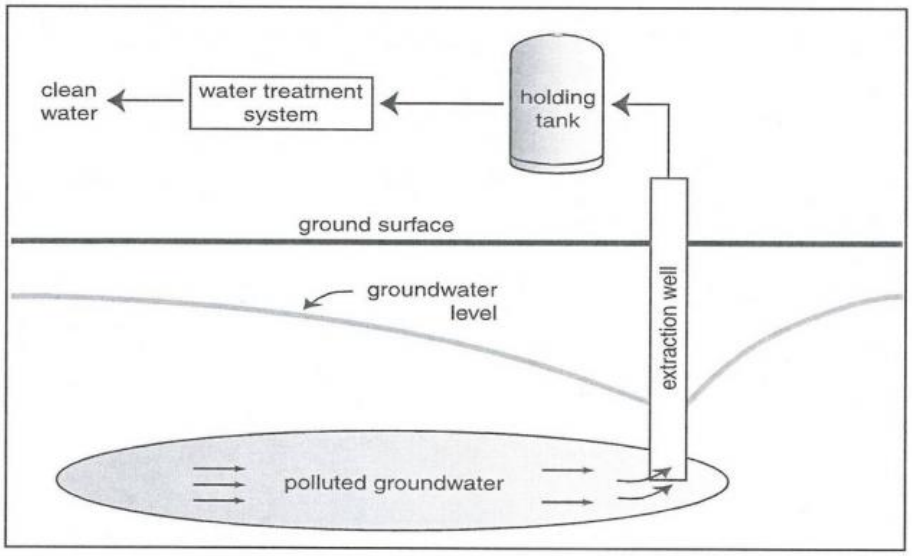
1. Pump and Treat

1.1 Technology Description

Pump-and-treat (P&T) is one of the most widely used groundwater remediation technologies and has been used at many Superfund sites where groundwater is contaminated. Conventional P&T methods involve pumping contaminated water to ground surface for treatment.

Groundwater extraction can create a cone of depression in groundwater that can then be used to manipulate the hydraulic gradient and thereby intercept contaminants that have migrated from upgradient, or to prevent contaminant plumes from moving beyond the hydraulically contained area. For bedrock that has a relatively low porosity, pumping from the center of the source area to achieve hydraulic containment can also result in removal of a significant volume of the dissolved contaminant mass when contaminated groundwater in the major water bearing fractures is extracted. Long-term P&T operation allows the removal of the contaminants that take time to slowly diffuse out from the secondary fractures or bedrock matrix.

P&T can be combined with other technologies to achieve a variety of remedial objectives, such as in when P&T is used for recirculation of substrate during in situ chemical or biological treatment, and where P&T serves as downgradient plume interception or migration control to allow upgradient source area treatment.



http://blog.augustmack.com/blog/august-mack-news/ex-situ-pump-and-treat

1.2 Process Options

P&T would include the following two major components:

Hydraulic containment

Pump and treat would be used to manipulate the movement of contaminated groundwater to control migration of contaminants and prevent continued expansion of the contaminated zone. Three major configurations for accomplishing hydraulic containment are: (1) pumping from extraction wells; (2) pumping from subsurface drain; or (3) pumping with a flow guiding system (funnel). The latter two

configurations are not considered practical for deep water-bearing bedrock. Therefore, pumping from bedrock extraction wells is the only applicable process option for this site.

Pumping tests and/or groundwater modeling are often needed to design groundwater extraction systems, perform capture zone analyses and to optimize operating P&T systems.

Treatment

Contaminated groundwater that is extracted from the subsurface typically requires treatment in order to remove the dissolved (and occasionally separate phase) contaminants prior to discharge or beneficial use. Commercially available treatment technologies for extracted groundwater contaminated with VOCs include liquid-phase carbon adsorption, air stripping/air-phase treatment, advanced oxidation (UV-H_2O_2, ozone- H_2O_2, etc.) .and biological treatment. Brief descriptions of those treatment processes are provided below.

- Liquid-phase carbon adsorption transfers contaminants from water to activated carbon. As the carbon becomes partially saturated with organic compounds, the amount of organic compounds sorbed by the carbon decreases, and the concentration of organic compounds in the effluent from the carbon bed increases. When the concentrations of contaminants in the effluent from the carbon adsorption unit reach unacceptable levels, the car bon must be disposed or regenerated. Contaminants are not destroyed by the carbon adsorption process. Regeneration facilities often include a unit process that destroys the organic contaminants after desorbtion from the carbon. The carbon adsorption process is relatively simple and can be easily implemented at the site. If the spent activated carbon is a RCRA hazardous waste, special handling, transportation, and regeneration/disposal of the carbon is required.
- Air stripping transfers VOCs from water to air. Once in the vapor phase, the VOCs can be captured using vapor phase carbon adsorption (which is often more efficient than liquid phase carbon adsorption) or destroyed using thermal/catalytic oxidation. In cases where the concentrations of VOCs in the air stripper exhaust are less than regulatory or risk-based criteria, then the air stripper exhaust can be discharged to the atmosphere untreated. As with liquid-phase activated carbon, vapor-phase carbon must be regenerated or disposed when VOCs break through the carbon bed at unacceptable levels. Vapor-phase carbon can be regenerated on-site using a solvent-recovery system. Thermal/catalytic oxidizers are often used when the VOC concentrations in the vapor stream are very high. Thermal/catalytic oxidizers can use significant amounts of fuel (typically natural gas or propane). Therefore oxidizers are generally not cost-effective unless the usage rate of vapor-phase carbon would be relatively high.
- Advanced oxidation technologies (AOT) use oxidizers such as ultraviolet (UV) light, hydrogen peroxide (H_2O_2), and ozone to destroy organic contaminants. The AOT process typically generates highly reactive hydroxyl radicals that rapidly oxidize contaminants into innocuous inorganic compounds such as carbon dioxide, water, and chloride. However, AOT processes are generally energy intensive and may require significant inputs of H_2O_2, Therefore, AOT processes are typically not used unless complete on-site destruction of contaminants is required, or if contaminants are present that cannot be removed by less costly treatment processes.

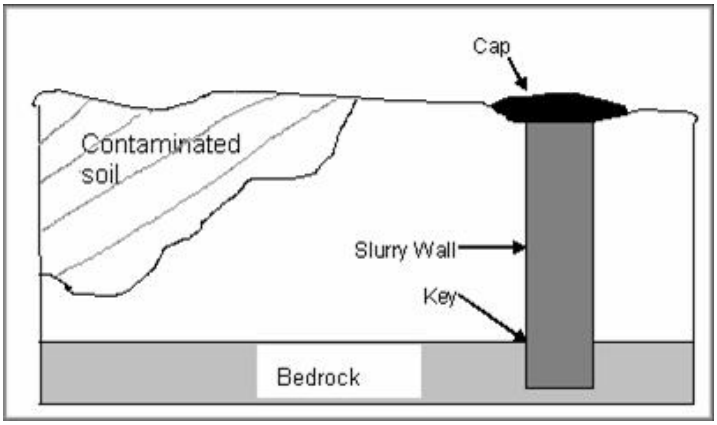
Air stripping/air-phase treatment using activated carbon has been used in the existing groundwater treatment facility (GWTF) at the site to remove VOCs and has potential to be upgraded for additional capacity.

2. Physical Barrier Containment

2.1 Technology Description

Physical barrier containment is a technology that can be used to prevent or reduce groundwater flow by applying physical blockage measures. The main advantage is that physical measures are often permanent once constructed and, if designed and constructed properly, can require minimal O&M effort while providing containment of contaminant source areas. Barriers are often used at sites where the source is incompletely characterized, inaccessible, or where long term remedial actions are being developed.

Physical barrier technology has demonstrated its effectiveness in containing contaminated groundwater in unconsolidated soil formations. Successful applications of physical barriers for containment of contaminated groundwater in bedrock have not been reported.



http://www.eugris.info/FurtherDescription.asp?e=24&Ca=2&Cy=0&T=Passive%20and%20hydraulicContainment

2.2 Process Options

Physical barriers may take a few forms depending on site-specific conditions. These include:

- Vertical trench(es) excavated and filled with low-permeability materials in order to intercept or reduce groundwater flow, such as a clay slurry wall.
- Pre-manufactured sheetpile keyed in vertically into a very low permeability layer that is below the contaminated zone order to intercept groundwater flow.
- Grouting with cement to solidify the subsurface matrix in order to prevent contact of clean groundwater with contamination.

Slurry walls and sheetpiles are common forms of physical barriers for overburden unconsolidated soil formations. However, these two processes are not practical for installation in fractured crystalline bedrock at bedrock depths that occur at the site, and will be not further evaluated.

Grouting with cement material to create impermeable curtain may be a practical approach to constructing a physical barrier in bedrock, and has been used to minimize the flow of water through bedrock below the foundations of river dams. Cement material is forced into the primary fractures and solidifies with time to completely plug the fracture. The contamination in groundwater in the secondary fracture matrix would be isolated and immobilized. Ultra high pressurized grouting through bedrock injection wells can be used to deliver the cement materials to the target fractures.

Bedrock grouting with great depth and distribution of fill materials into fractures through limited locations is very challenging. Confirmation drilling would be performed to verify the quality of fracture grouting. Additional rows of grouting points may be needed to reduce the chance of missing fractures. The bottom of the contained area cannot not be practicably grouted. Therefore, if there are hydraulic connections to the zone outside of the containment area, contamination may escape and spread to downgradient locations.

3. In-situ Thermal Treatment

3.1 Technology Description

In situ thermal treatment has been demonstrated to be effective for remediation of DNAPL in weathered bedrock .This technology is commonly combined with a soil vapor extraction (SVE) system when treating thermally non-degradable organic chemicals.

Thermal treatment is effective for removal of VOCs that are in separate, dissolved, and adsorbed phases, relying on mechanisms including desorption, evaporation, and boiling. When heated to boiling, groundwater can generate fine bubbles, and eventually pressurized steam flow. Boiling occurring at DNAPL-water interfaces causes direct evaporation of DNAPL. Higher temperatures increase the dissolution of DNAPL into groundwater. Bubbling facilitates evaporation of contaminants from groundwater and the pore matrix into the gas flow.

Steam that is injected or that results from boiling of groundwater can sweep towards locations with lower pressure. Locally generated steam flow can be effective at mobilizing contaminants to capture points (extraction wells, SVE wells) where they are removed from the subsurface. The contaminant vapors that are collected by the SVE system are treated ex situ. Vacuum extraction points are typically installed throughout the treatment area to maximize contaminant removal, and are also installed outside the perimeter of the treatment area to prevent contaminant migration away from the site.

To be effective, the thermal treatment of fractured bedrock sites needs to:

- Achieve thorough heating of the zone targeted for treatment;
- Prevent unwanted condensation of contaminant-laden steam and vapors; and
- Capture and remove contaminant mass liberated from the bedrock fractures.

- Prevent migration of volatilized or liquid contaminants away from the treatment area.

The effectiveness of thermal treatment for fractured bedrock has yet to be proven with full-scale applications.

3.2 Process Options

Heating can be accomplished by injecting steam into the contaminated zone or by installing subsurface heaters powered by electricity or a combustible fuel source. The injection points/heaters are installed at regular intervals throughout the treatment area in a manner that will provide a sufficient level of heating to the zone targeted for treatment. Use of in situ thermal treatment for the bedrock at the Savage Well site be extremely costly for installation of numerous heater and SVE wells, and for the large amount of energy required to heat the bedrock treatment zone to the target temperature.

Four types of thermal process options that may be used to heat the groundwater matrix are discussed below.

- Thermal Conductance Heating
- Electric Resistance Heating
- Radio Frequency Heating
- Steam Injection

Thermal Conductance Heating

Thermal conduction is the process of heat flowing from an object of higher temperature to another of lower temperature. In rock, heat flows from heater wells out into the formation across solid objects (rocks) or by grain-to-grain contact (in soil). The fluids (water, air, NAPL) in contact with the solids also heat up at the same time. The heat moves out radially from each thermal well until the heat fronts overlap. Compared to fluid injection processes, the conductive heating process can be uniform in its vertical and horizontal sweep.

Because the porosity in the bedrock matrix is very low (often <1%), heating the rock to very high temperature in order to pass the energy to groundwater (in the portion of porosity) to boil out is not energy efficient. However, TCH can rapidly raise the temperature of the matrix with intensive heating, thereby reducing the heat loss attributed to groundwater outflow from the treatment area, and improving the overall operation efficiency.

Electric Resistance Heating

When electric current passes through the bedrock matrix, the resistance, when at the right level, causes the rock, and the groundwater within the rock, to heat up. As most bedrock types have a low electric conductivity, electric current flows between electrodes primarily through groundwater to heat it up. When the water starts to boil off, electric current may be weakened or disconnected and heating would be interrupted. In fractured bedrock, the fractures are not evenly distributed in the matrix, and may not be well connected. During ERH treatment, a portion of the fractures with groundwater may not have

electric current passing through, causing uneven heating. Heating may not occur in the desired fractures or be interrupted before a temperature sufficient for volatilization and extraction of the contaminants is achieved. Because of its limitations, the ERH is not considered an effective option for deep fractured bedrock treatment and will not be carried through for further evaluation.

Radio Frequency Heating

RFH system heats up water by passing electromagnetic energy through water such as in a microwave oven. The RF generator creates an alternating electromagnetic field between two electrodes, causing polar molecules such as water to continuously reorient them to face opposite electrodes. Friction resulting from this molecular movement causes the material to rapidly heat almost uniformly throughout its entire mass. Physical connection of the material is not necessary for heating.

Because rocks in the bedrock matrix do not contain polar molecules which would be able to reorient in the electromagnetic field, the rocks will not be heated by RFH, and only the groundwater would absorb the energy and be heated, resulting in a potentially high efficiency of energy use.

Heating the bedrock groundwater to generate steam, complemented with SVE to remove the steam for treatment above ground surface, has not been in full-scale application. While theory indicates that RF heating may be effective for treatment of contaminated water in bedrock, RF heating technology for treatment of bedrock has not been proven in pilot or full-scale applications.

Steam Injection

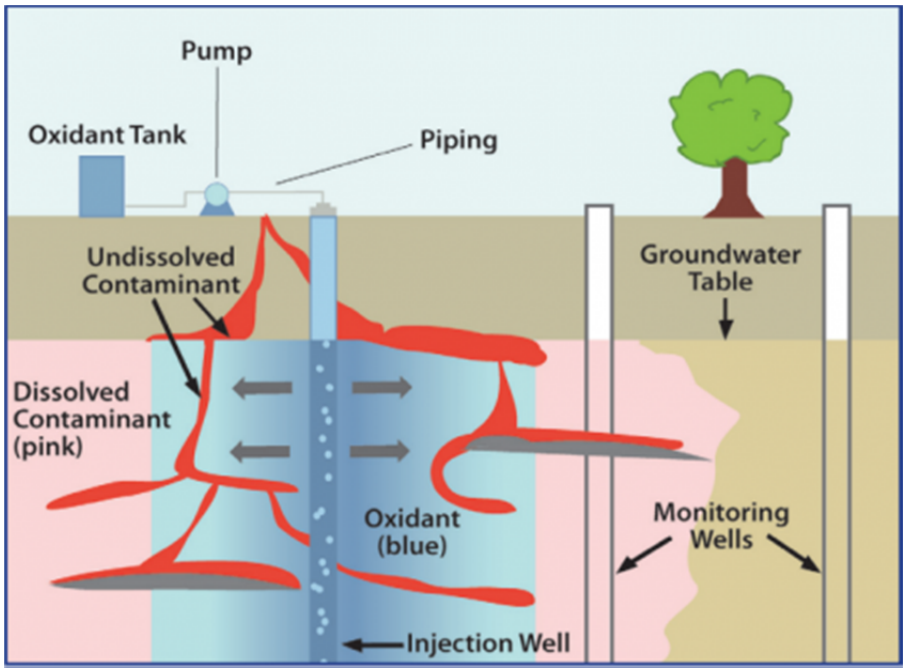
Steam injection is accomplished by processing steam that generated above-ground into contaminated zones. For bedrock, steam may be effective where injection into larger fractures can be accomplished and where those fractures are intercepted by heater and vapor/multi-phase extraction wells. For the Savage Well Site, high pressure injection of steam would be needed because of the greater than 300-foot injection depths. Management and injection of high pressure steam would require extensive safety precautions and an effective containment network to eliminate the risk of injected steam causing migration of contaminants to areas where containment/extraction was not provided.

4. IN-SITU CHEMICAL TREATMENT TECHNOLOGIES

4.1 Technology Description

Chemical treatment technologies employ aqueous chemical reduction-oxidation reactions to transform the toxic VOC contaminants into environment benign or less toxic compounds. For in situ bedrock applications, strong chemical amendments would be injected into the bedrock fractures to achieve contact with organic contaminants. Although the final products are different, either chemical oxidation or chemical reduction may be used to treat PCE and its daughter products at this site. For chemical reactions to sustain until the concentration of contaminants reduced to desirable level, adequate amount of reagents should be provided through injection.

A desired property of the selected chemical amendments for bedrock DNAPL remediation is the ability to follow DNAPL migration pathways and find high contaminant concentrations for treatment. Although the chemical reagents would unlikely react with DNAPL directly, rapid degradation of the dissolved contaminants will accelerate the dissolution of DNAPL and shorten the time frame for complete site cleanup.



http://www.groundsure.com/blogs/persulfox

4.2 Process Options

Chemical treatment in general includes two process options, chemical oxidation and chemical reduction.

In Situ Chemical Oxidation

In Situ Chemical Oxidation (ISCO) employs strong oxidants to mineralize organic compounds such as chlorinated ethenes at this site (i.e., PCE and daughter products) to non-toxic inorganic products including carbon dioxide, water, and chloride ions. Permanganate is often used for remediation of chlorinated ethenes because of its recognized advantages in the treatment, such as high reaction rate, relatively low soil oxidant demand, persistence in treatment matrix, and no pH adjustment or catalyst/activator addition. Sodium permanganate can be applied with concentration as high as 20%, which is significantly more dense than water and has the potential to follow migration pathways and thereby contact residual DNAPL and associated high concentrations of dissolve contaminants. Other oxidants including persulfate, ozone, and hydrogen peroxide are also commercially available for consideration.

It is challenging to effectively deliver oxidants in bedrock to achieve contact with contaminants because a significant portion of contamination mass is present in geometrically complex secondary fractures.

Soluble chemical oxidants migrate in primary fractures with groundwater flow. Injected oxidants may not be sufficiently persistent in the fractures to rely on counter-diffusion of chlorinated solvents resident in stagnant fractures and the rock matrix into mobile groundwater for reaction.

In Situ Chemical Reduction (ISCR)

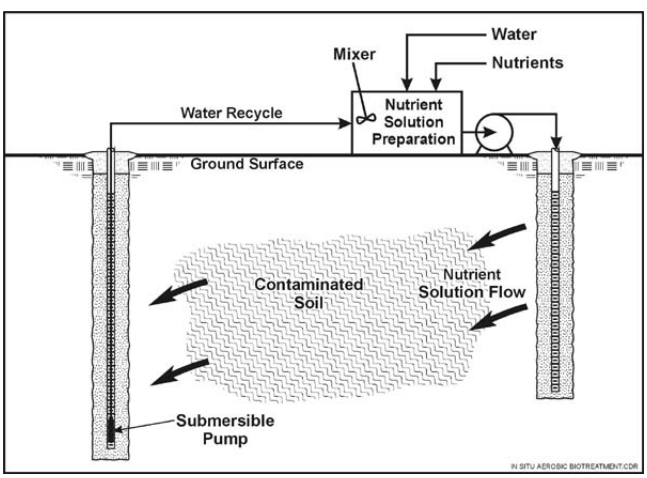
In Situ Chemical Reduction (ISCR) employs strong reducing reagent to reduce chlorinated ethenes to non-toxic products including chloride, ethene, ethane, or acetylene. Granular zero-valent iron (ZVI) has been widely used to treat chlorinated ethenes in ISCR applications, such as permeable reactive barriers (PRBs). ZVI particles of microscale or nanoscale sizes can be fluidized and injected into subsurface matrix for active and targeted remediation and accelerated soil and groundwater treatment because of their high reactivity in contaminant removal. The chemical reactions are surface mediated, i.e. contaminants need to contact and attach to the ZVI particle surface for dechlorination reactions to occur. Because ZVI suspension solution is heavier than water, ZVI has a potential to follow DNAPL migration pathways to attack trapped residual DNAPL and associated high concentrations of contaminants. Once ZVI particles are settled, they will serve as PRB to control contamination migration with groundwater.

In addition, strongly negative redox conditions and slow release of dissolved hydrogen gas within the treatment zone and downgradient create favorable conditions for anaerobic microbial growth and enhance bioremediation of CVOCs.

5. IN-SITU BIOLOGICAL REMEDIATION

5.1 Technology Description

In Situ bioremediation (ISB) relies on capable bacteria in the aquifer to degrade dissolved organic contaminants to non-toxic products. ISB for chlorinated ethenes such as PCE may be achieved by amending the groundwater with necessary substrates including organic carbon, nutrients, and minerals to stimulate growth of indigenous bacteria capable of degradation and sustain biodegradation of the contaminants. In some cases microbial media may be injected to the subsurface when indigenous microorganisms are incapable of dechlorination. The amendments would be injected via delivery systems such as injection wells.



https://portal.navfac.navy.mil/portal/page/portal/navfac/navfac_ww_pp/navfac_nfesc_pp/environmental/erb/imageserb

5.2 Process Options

ISB in general has two process options, anaerobic bioremediation and aerobic bioremediation.

Anaerobic bioremediation

In situ anaerobic bioremediation requires anaerobic conditions for the dechlorination bacteria to survive and grow. Anaerobic conditions are can be achieved by injecting a sufficient quantity of electron donors (i.e. food material) to consume excessive content of undesired electron acceptors, other than the target contaminants, such as oxygen and nitrate, and provide carbon sources for the growth of dechlornation microbes. Nutrients and minerals are also necessary substrate for microbial healthy growth. During ISB, the chlorinated ethenes are removed as electron acceptors during metabolite processes when bacteria can gain energy for their growth. Chlorine atoms in chlorinated ethenes are sequentially removed from the chlorinated contaminant and replaced with hydrogen atoms, producing final products ethene.

Several full-scale ISB anaerobic applications to treat chlorinated solvents in bedrock matrix have been reported. Commercial products serving as long-term carbon and nutrient sources are available in the market. As for other in situ treatment technologies, successful delivery of sufficient amount of amendments for contact with the contaminants, particularly to target the contaminants in the secondary fractures, is crucial for the ISB application at this site. Microbial study to ensure the presence and sustainable growth of dechlorination bacteria in the bedrock fractures is necessary.

Aerobic bioremediation

In situ aerobic bioremediation typically entails maintaining a high level of organic substrates and dissolved oxygen in groundwater to sustain the healthy growth of aerobic microorganisms for contaminants degradation. Under aerobic conditions, microorganisms use dissolved oxygen in groundwater as electron acceptors and grow in a faster pace than that under typical anaerobic conditions, can completely oxidize many organic chemicals to carbon dioxide and water. TCE and DCEs in the groundwater may be biodegraded via cometabolism under aerobic conditions in the presence of a suitable substrate (e.g. methane, propane, or butane).

Because of its already high oxidation state of carbon atoms, degradation of PCE under aerobic conditions is more difficult than degradation of TCE and DCEs, and full-scale applications have not been reported. Aerobic option will not be further considered at this site.

6. MONITORED NATURAL ATTENUATION (MNA)

6.1 Technology Description

MNA, monitored natural attenuation, refers to the remediation relying on natural attenuation processes to achieve the groundwater regulatory standards. MNA is a common component of remedies selected for Superfund sites with groundwater contaminated by dilute concentrations of chlorinated solvents. Natural attenuation often uses a variety of processes that can reduce the mass or concentration of contaminants in groundwater, without enhancement of engineered measures. These processes include biodegradation, dispersion, dilution, sorption, volatilization, and chemical and/or biological destruction of contaminants. The groundwater needs to be periodically monitored to quantify the effectiveness of the natural attenuation.

Because of the suspected DNAPL presence in the bedrock fractures and already detected high contaminant concentrations at the site, MNA alone without active remedial measures for the source area is not expected to effectively in prevent contaminant migration. Although the presence of PCE daughter products in the plume suggests that microbes may be active at the site for PCE mass reduction, the effectiveness and mechanisms of natural attenuation should be further evaluated if MNA is retained to develop remedy alternatives. Additionally, some natural attenuation processes may form degradation products that are more mobile and/or more toxic than the parent contaminants, the presence of such degradation process and products must be assessed.

6.2 Process Options

Source control and long term monitoring are in general fundamental components of MNA remedies. Following source control measures, MNA could be used in combination with or without other active remedial options, depending on whether MNA alone is sufficient to achieve remediation objectives. Phased remedial strategies are often considered with implementation of source control during initial phases and MNA as the polishing phase to remediate slower release and more diluted groundwater contamination.

MNA can be accomplished in general using conventional equipment and analytical methods; therefore, it is readily implementable. Existing monitoring wells may be used or additional wells may be installed to collect from specific areas that do not currently have monitoring wells.