



Conceptual Site Model Site Investigation Report: Bennington, Vermont

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Conceptual Site Model Site Investigation Report

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Acronyms and Abbreviations

Abbreviation/ Acronym	Description
AHC	Agglomerative Hierarchical Cluster analysis
bgs	below ground surface
CAAI	Corrective Action Area II
CAAs	Corrective Action Areas
CSM	conceptual site model
ELLE	Eurofins Lancaster Laboratories Environmental, LLC
EPA	U.S. Environmental Protection Agency
ES	Environmental Standards
f_{oc}	fraction of organic carbon
FSP	Field Sampling Plan
gpm	gallons per minute
IDW	investigation derived waste
K	hydraulic conductivity
LCS	laboratory control sample
MDL	method detection limit
mg/kg	milligrams per kilogram
MSL	mean sea level
MVA	multivariate analysis
NAVD88	North American Vertical Datum of 1988
ng/g	nanograms per gram
ng/L	nanograms per liter
PAH	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
PFAS	per- and polyfluoroalkyl substances
PFBA	perfluorobutanoic acid
PFBS	perfluorobutanesulfonic acid
PFHpA	perfluoroheptanoic acid
PFHxA	perfluorohexanoic acid
PFHxS	perfluorohexanesulfonic acid
PFNA	perfluorononanoic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctanesulfonic acid
PFPeA	perfluoropentanoic acid
PFTeDA	perfluorotetradecanoic acid
PTFE	polytetrafluoroethylene
PID	photoionization detector
POET	point-of-entry treatment system

ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
PVC	polyvinyl chloride
QA/QC	quality assurance/quality control
QAPP	Quality Assurance Project Plan
RPD	relative percent difference
SVOC	semivolatile organic compound
TAL	target-analyte-list
TOC	total organic carbon
TOP	total oxidizable precursor
VGS	Vermont Geological Survey
VOC	volatile organic compound
VTANR	Vermont Agency of Natural Resources
VTDEC	Vermont Department of Environmental Conservation
WWTP	wastewater treatment plant

Executive Summary

An investigation was conducted on behalf of Saint-Gobain Performance Plastics (Saint-Gobain) to further assess the distribution of per- and polyfluoroalkyl substances (PFAS) in soil and groundwater in an investigation area in and near the Town of Bennington and the Village of North Bennington, Vermont. Specific investigation objectives included:

- assessment of site hydrogeologic conditions;
- evaluation of the relationships and trends in PFAS concentrations;
- evaluation of potential impacts from the former Bennington Landfill;
- identification of some of the other potential sources of PFAS in the investigation area and measurement of background levels of PFAS present in soil and groundwater within the Corrective Action Areas (CAAs); and
- assessment of the measured distribution of PFAS in soil and groundwater against the distribution based upon modeling of air emissions at the former Chemfab facilities.

The investigation was completed in accordance with the requirements of the Consent Order between the State of Vermont Agency of Natural Resources (VTANR) and Saint-Gobain dated October 2, 2017 and the *Revised Work Plan – CSM Site Investigation: Bennington, Vermont* (Work Plan; Barr, 2017a) approved by the Vermont Department of Environmental Conservation (VTDEC) as amended (VTDEC, 2017a; Barr, 2017b).

The investigation area includes two CAAs. As identified in the Consent Order, Corrective Action Area I generally consists of a bounded area west of Route 7 (Harwood Hill Road), and Corrective Action Area II (CAAI) generally consists of a bounded area east of Route 7. PFAS has been detected in groundwater at varying concentrations and in various locations, including some private water wells, within both CAAs. Investigation activities described in this report were performed at locations primarily within the CAAs, but also at some locations beyond the CAA borders (e.g., background sample locations).

The investigation activities included drilling shallow and deep soil borings in unconsolidated materials located along transects between Bennington Landfill and the two former Chemfab facilities and in each cardinal direction from the former Chemfab on Water Street facility (former Water Street facility). Four bedrock borings also were advanced into bedrock at or near Bennington Landfill, and were completed as permanent monitoring wells. The locations of the wells were selected based on the results of a bedrock outcrop and lineament study of the landfill area. A total of 568 soil samples were collected from the borings and analyzed for PFAS, total organic carbon, soil pH, and moisture content. Soil samples from the deep soil borings and bedrock monitoring wells were also analyzed for metals and cations. Temporary wells were installed in 16 shallow soil borings and screened at or across the water table and permanent monitoring wells were completed in the unconsolidated aquifer at eight deep soil borings. All groundwater samples were analyzed for PFAS, metal/cations (excluding cyanide and mercury), anions, ammonia, orthophosphate, and total Kjeldahl nitrogen. Groundwater samples collected within the Bennington Landfill area were also analyzed for cyanide, mercury, volatile organic compounds, and 1,4-dioxane to assess the presence of landfill leachate impacts. Bedrock groundwater elevations were monitored by installing data-logging pressure transducers in previously existing bedrock monitoring wells

at Bennington Landfill and in seven private water wells completed in bedrock near the landfill. Slug testing was conducted to obtain estimates of hydraulic conductivity in seven permanent monitoring wells installed for the investigation. Hydraulic conductivity estimates were also obtained from analysis of grain-size distribution data for samples collected from investigation soil borings and from specific capacity values for the new bedrock wells. Borehole characterization of the new bedrock wells was performed using geophysical and flow logging methods to evaluate the potential for discrete groundwater flow through structures identified from the bedrock outcrop and lineament study.

Findings associated with the investigation objectives include the following:

- hydrogeologic conditions are similar to those described in the conceptual site model (CSM);
- relationships and trends in PFAS concentrations indicate that current concentrations in CAII soil are not indicative of historical impacts from the former Chemfab facilities;
- statistical analysis of available groundwater data from this investigation suggest different (multiple) PFAS sources, including the Bennington Landfill, as reflected by groundwater of different PFAS composition;
- concentrations of PFAS compounds in the bedrock aquifer at Bennington Landfill are highest directly downgradient (southwest) of the landfill cap area; are at much lower concentrations upgradient of this area; and are consistent with bedrock flow through fractures and zones of bedrock alteration forming discrete flow paths; and
- measured soil concentrations across the investigation area are generally consistent with background concentrations and may also be indicative of localized sources of PFAS.

After accounting for background and local sources other than the former Chemfab facilities, such as the Bennington Landfill, the data collected as part of this investigation are consistent with the CSM. Measured PFAS concentrations were higher in some areas of CAII than the modeled values, which is indicative of the fact that there are background local, regional and global sources of perfluorooctanoic acid (PFOA) other than the former Chemfab facilities that were not accounted for in the CSM and that have contributed to PFAS concentrations in CAII. Some of these sources are identified in this report. For the purposes of assessing the effects of air emissions from the former Chemfab facilities, an evaluation of the site data indicates that additional investigations within the CAAs are not warranted. However, additional investigation by the appropriate regulatory agency to evaluate other potential local or regional sources, such as those identified in this report, may be appropriate.

1.0 Introduction

The investigation activities described herein were conducted on behalf of Saint-Gobain Performance Plastics (Saint-Gobain) and were completed in accordance with the requirements of the Consent Order between the State of Vermont, Agency of Natural Resources (VTANR) and Saint-Gobain dated October 2, 2017 (Consent Order) and the *Revised Work Plan – CSM Site Investigation: Bennington, Vermont* (Work Plan; Barr, 2017a) approved by the Vermont Department of Environmental Conservation (VTDEC) as amended (VTDEC, 2017a; Barr, 2017b). The work was performed by consulting staff from Barr Engineering Co., C.T. Male Associates, and Golder Associates Inc.

The purpose of this investigation was to further assess the distribution of per- and polyfluoroalkyl substances (PFAS) in soil and groundwater in an investigation area encompassing the Village of North Bennington and portions of the Towns of Bennington and Shaftsbury, Vermont (Figure 1.1) to meet the following site-specific objectives:

- assess hydrogeologic conditions;
- evaluate relationships and trends in PFAS distribution, specifically in relation to perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS);
- evaluate potential impacts from the former Bennington Landfill;
- identify some of the other potential sources of PFAS within the investigation area and measure background levels of PFAS present in soil and groundwater within the Corrective Action Areas (CAAs); and
- assess the measured distribution of PFAS concentrations in soil and groundwater against the modeled distribution.

Data collected as part of this investigation were used to further refine the conceptual site model (CSM) of PFAS fate and transport developed for the area. The CSM is described in the *Draft Conceptual Modeling of PFOA Fate and Transport: North Bennington, Vermont* (Draft CSM Report; Barr, 2017c), and was developed using site data available prior to January 2017; chemical characteristics of PFAS; and modeling studies of processes important to PFAS fate and transport.

For the purposes of this investigation, the specific areas of interest within the investigation area included the former Bennington Landfill and the former Chemfab facilities on Water Street in North Bennington and on Northside Drive in Bennington (Figure 1.1). While there are other likely sources that have contributed to PFAS concentrations within the CAAs, some of which are identified in this report, a detailed investigation of such potential sources was beyond the scope of this report.

Investigation activities at the former Bennington Landfill were completed as part of this investigation. However, investigation activities at the former Chemfab on Northside Drive facility (former Northside Drive facility) were not completed because access to the former facility was not available at the time of Work Plan development and approval. Investigation of soil and groundwater at the former Water Street facility was conducted in 2016 in accordance with the *Initial Site Characterization Investigation Work Plan* (Water Street Work Plan, C.T. Male, 2016a). Results from that investigation are discussed in this report.

Additional investigation activities at the former Water Street facility are currently being conducted in accordance with the approved *Supplemental Site Investigation Work Plan – Former Chemfab Facility Water Street* (Supplemental Water Street Work Plan, C.T. Male, 2018). Additional investigations will be completed at the former Northside Drive facility, pending access and following the development and approval of an investigation work plan. Results from additional investigation activities at the former Chemfab facilities will be reported separately, and submitted to VTDEC in accordance with the schedule presented in the Consent Order.

1.1 Investigation Objectives

As stated above, the purpose of this investigation was to further assess the distribution of PFAS in soil and groundwater at the site. These data were collected to meet the following site-specific investigation objectives identified in the Work Plan:

- **Assess site hydrogeologic conditions.** Additional information regarding the hydrogeologic conditions was collected in areas with limited or no available data, specifically for bedrock in the Bennington Landfill area.
- **Evaluate relationships and trends in PFAS concentrations.** Soil analytical results were evaluated to enhance the understanding of geochemical relationships for PFAS and determine what, if any, relationship exists between total organic carbon (TOC), metals, and PFAS concentrations. Soil and groundwater analytical results were evaluated for distributional relationships, such as spatial (depth or distance) or lithological (unconsolidated or bedrock).
- **Evaluate potential impacts from Bennington Landfill.** Monitoring wells were installed at the landfill to characterize groundwater flow directions in this area relative to nearby private wells and to further assess the role of Bennington Landfill as a potential source of PFAS.
- **Identify other potential sources of PFAS within the investigation area and evaluate background conditions.** A preliminary evaluation of other potential PFAS sources in the investigation area was performed using available public records and data available prior to this investigation. In addition, samples were collected from areas not expected to have been impacted by air deposition of PFAS from the former Chemfab facilities for the purpose of establishing background concentrations of PFAS in soil and groundwater resulting from regional and global sources.
- **Assess measured distribution of PFAS in soil and groundwater against the expected distribution.** For purposes of this report, soil analytical results from the upper unsaturated zone soils (O horizon and C horizon mineral zone beneath the root zone) and at depths to bedrock or the water table were used for comparison against the air deposition and unsaturated zone models. Similarly, groundwater results were evaluated against the modeled concentrations.

1.2 Report Organization

The remainder of this report is organized into the following sections:

- **Section 2.0 Background** – discussion of the site history; physical setting; characteristics of PFOA, PFOS, and associated compounds; and the PFOA fate and transport conceptual modeling approach and results.
- **Section 3.0 Work Plan Implementation Activities and Results** – summary of the investigation activities and methods, including sample collection and analyses and deviations from the Work Plan; and discussion of the results of soil and groundwater sampling (including Quality Assurance/Quality Control (QA/QC), hydrogeologic investigation activities, and bedrock evaluation, including results from drilling, desktop and outcrop studies, and geophysical logging.
- **Section 4.0 Analysis** – summary of the interpretation and analysis of the investigation data with respect to hydrogeologic conditions, geochemical relationships of PFAS in the soil and groundwater, potential additional PFAS sources, and the observed distribution of PFAS in soil and groundwater.
- **Section 5.0 Summary** – discussion of findings as related to the objectives of the investigation.

2.0 Background

The following discussion of background information relevant to this investigation is drawn primarily from the Draft CSM Report (Barr, 2017c), with the exception of 2016 soil and groundwater data collected from property adjacent to the former Water Street facility as part of the Water Street Work Plan implementation, as described in Section 2.1.1 below.

2.1 Site Description and History

For the purposes of this investigation, the site is an area of approximately 35 square miles in southwestern Vermont (Figure 1.1), encompassing the Village of North Bennington and portions of the Towns of Bennington and Shaftsbury. While there are other potential PFAS sources within the investigation area, as described in Section 4.4, this investigation primarily focused upon the two former Chemfab facilities and Bennington Landfill. Background information and history for each of these is included below.

2.1.1 Former Chemfab Facilities

The former Chemfab facilities are located on Northside Drive in Bennington and on Water Street in North Bennington (Figure 1.1). The former Northside Drive facility operated from 1969 until mid-1978 when operations were transferred to the Water Street facility. The former Water Street facility operated from mid-1978 until February 2002, when the plant closed. The facilities applied polytetrafluoroethylene (PTFE) coatings to some fiberglass fabrics. During drying and curing of certain fabrics, a small portion of the PFOA driven off the fabric may have been emitted from the facilities' stacks. Modeling results presented in the Draft CSM Report (Barr, 2017c) suggest that air emissions of PFOA from the former Chemfab facilities contributed some PFOA, in varying amounts, to groundwater in certain, but not all, portions of the investigation area.

A shallow soil investigation was completed in 2016 at and around the former Water Street facility. Of the 103 soil samples collected and analyzed for PFAS compounds, 16 were collected adjacent to the former Water Street facility. The highest PFOA detection in soil was 20 nanograms per gram (ng/g) in a sample collected within 15 feet of the facility. In contrast, the highest PFOS concentration in soil (2.8 ng/g) was detected over 3 miles southeast of the facility. Additional information from the shallow soil investigation completed in 2016 is available in the *Draft Shallow Soil Sampling Report, Former Chem Fab Site & Surrounding Areas* (C.T. Male, 2016b).

Soil and groundwater investigation activities were also completed adjacent to the former Water Street facility as part of the Water Street Work Plan investigation (C.T. Male, 2016a). In a total of 54 soil samples collected and analyzed for PFAS as part of this investigation, PFOA was detected above the method detection limit (MDL) in 46 samples (at concentrations ranging from 0.49 ng/g to 30 ng/g). Again, by contrast, PFOS was detected in only four soil samples (at qualified concentrations ranging from 0.76 ng/g to 1.1 ng/g). Two rounds of groundwater samples from eleven wells were also collected as part of this investigation. Whereas PFOA was detected in all samples, PFOS was only detected in six samples. In those groundwater samples in which PFOS was detected, it was present at concentrations that were one to three orders of magnitude lower than PFOA concentrations in the same samples. The results of this

investigation are discussed briefly in this report (Section 4.4), and a comprehensive summary will be included with results from the Supplemental Water Street Work Plan (C.T. Male, 2018).

2.1.2 Bennington Landfill

Bennington Landfill, shown on Figure 1.1, is a closed landfill that received municipal sanitary waste from the Town of Bennington from 1969 until 1987 (U.S. EPA, 2004). Historical records indicate that polychlorinated biphenyls (PCBs), volatile organic compounds (VOCs), and lead were disposed of in the landfill. Several industries in the Bennington area dumped liquid wastes into an unlined lagoon at the landfill from 1969 to 1975. An underdrain was installed in 1976 to dewater the unlined lagoon (McLaren/Hart, 1997). This underdrain was extended to the north in 1979-1980 as waste was disposed of in that area. A treatment system was eventually installed to treat leachate collected by the underdrain. The treated leachate was infiltrated on site in an infiltration gallery east of the leachate vault. The landfill was graded and capped by 1999 and other measures were taken to reduce production of leachate. A drain structure, referred to as the Upgradient Groundwater Isolation Trench, was installed as part of the capping activities to intercept shallow groundwater and divert that water around the landfill (TRC, 1998). The leachate treatment system was decommissioned in 2008. Since 2008, untreated leachate has been collected in a storage vault, pumped out of the vault when leachate levels are high, and treated and disposed of offsite (VT ANR, 2016).

Leachate from Bennington Landfill has been shown to have contaminated surface water and groundwater with PCBs and VOCs. The facility was listed as a Superfund site in 1989 for this reason (U.S. EPA, 2004). Constituents monitored in the landfill leachate and monitoring wells include PCBs and VOCs. Monitoring for PFOA and other PFAS compounds was not undertaken until an investigation in March and April of 2016 (Weston Solutions, Inc., 2016), during which PFOA was measured at concentrations ranging from 5,300 parts per trillion (ppt) in the leachate vault to non-detect (< 2 ppt) in the two bedrock wells sampled (B-2-3 and B-6-3). Groundwater sampling and analysis of shallow monitoring wells (i.e., not screened in bedrock) was performed on behalf of the U.S. Environmental Protection Agency (EPA) in 2017 as part of a larger investigation, and included completion of soil borings and monitoring wells and collection of soil and groundwater samples in the southwestern portion of the landfill area (Weston Solutions, Inc., 2017). Sample splits were collected by Saint-Gobain representatives during this monitoring well sampling event. The results of that investigation are discussed in this report.

2.2 Physical Setting

The regional and site geology and hydrogeology are described in the following sections. Additional data related to the physical setting were collected during this investigation and an interpretation of the geology and hydrogeology within the investigation area is presented in Section 4.1 and Section 4.2, respectively.

2.2.1 Regional Geology and Hydrogeology

The investigation area is located in the Northeastern Appalachians groundwater region which is characterized by rolling topography that primarily reflects the weathered bedrock surface with glacial and fluvial landforms mantling the bedrock (Randall et al., 1988). The following regional geology and

hydrogeology summary is condensed from that resource. Additional discussion of information related to regional geology and hydrogeology is provided in the Draft CSM Report (Barr, 2017c).

The bedrock in the area consists of folded and faulted metamorphosed sedimentary rocks with low primary porosity due to the rock matrix. Water is conducted in the bedrock through secondary porosity caused by, for example, fracturing, faulting, or dissolution. In some rock types, such as those in the North Bennington area, the secondary porosity is enhanced and enlarged by dissolution. The frequency and permeability of secondary porosity typically decreases with depth. Most bedrock wells in the region are between 200 and 400 feet deep.

Glacial erosion and deposition produced changes in drainage and topography and deposited a nearly continuous layer of unconsolidated till over the bedrock. Stratified drift units, chiefly sand and gravel, are associated with the larger valleys such as those of the Walloomsac River.

Recharge to the bedrock is controlled by the permeability and thickness of the overlying glacial deposits and overburden. Runoff in upland areas is focused to seasonal streams that typically lose discharge in areas in which they flow over stratified drift at the margins of the larger valleys. Bedrock wells that are pumped continuously may receive pumping-induced recharge from adjacent stratified drift aquifers. Discharge is primarily from the bedrock to wells and to the stratified drift in the large valleys. Inter-basin flow systems with significant discharge have not been discovered in the bedrock.

In addition to discharge to the stratified drift filling the larger valleys from minor upland streams, recharge to these units is from direct infiltration of precipitation, and discharge from bedrock. Recharge from the larger rivers may take place in the case of localized pumping from the stratified drift or higher-than-normal river stage. Discharge from the stratified drift is via pumping wells, evapotranspiration, and to the larger rivers when their stage is at or below typical levels.

2.2.2 Groundwater Use in the Investigation Area

Groundwater use in the investigation area is primarily from private wells. No high-capacity industrial or commercial wells permitted by the State of Vermont were identified from available information on water use and well construction. The municipal water sources for the Town of Bennington are Morgan Spring and Bolles Brook, which are spring and stream source types, respectively (Town of Bennington, 2016). The municipal water sources for the Village of North Bennington are supply wells near the surface-water intake on Basin Brook, which is located in Shaftsbury (VTDEC, 2016). PFOA has not been detected in these municipal water supplies (Barr, 2017c).

2.3 Characteristics of PFOA, PFOS, and Associated Compounds

PFOA is one of a class of fluorinated hydrocarbon compounds associated with fluoropolymers – most commonly recognized as Teflon®. Like PFOA, PFOS is a “C8” PFAS, possessing eight carbon atoms, each attached to two fluorine atoms. PFOA was manufactured by 3M and DuPont in the United States until its production was phased out between 2002 and 2006. PFOS was manufactured by 3M using an electrochemical fluorination process until 2002. Since that time, shorter-chain PFAS have been used.

PFOA and PFOS were used in the manufacture of many commercial materials for industrial and residential use, including stain-resistant carpeting/furniture/upholstery, non-stick cookware, food package coatings, aqueous film-forming foams, moisture-resistant breathable fabrics, concrete and rock sealants, electrical capacitors, dyes, paints and coatings, ski wax, batteries, photographic films, printing inks, herbicide and pesticide formulations, car wash surfactants, and as a vapor suppressor in metal plating processes (Kissa, 2001; 3M, 1999; Knepper and Langue, 2012; Olsen, 2015). They are a ubiquitous presence in most households and consequently are found in municipal waste streams and in most landfills (Busch et al., 2009; MPCA, 2009). In some manufacturing processes, PFOA is used and may be released to the environment through air emissions. They are also found in biosolids, in part from municipal sewer sludge used as amendments to soil in agricultural and landscaping applications, including gravel pit reclamation (Sepulvado et al., 2011; NEBRA, 2014).

PFOA and PFOS are highly soluble in water and are resistant to degradation. They readily bond to organic material surfaces and adsorb poorly to surfaces with low organic material content. In saturated, unconsolidated deposits and fractured rock media, they are mobile and migrate as an unattenuated solute in flowing groundwater. In general, they are considered "conservative" solutes because the compounds' conservation of mass is confined to the dissolved phase, and the compound is not substantially converted to other forms. Very small sources of PFOA and PFOS can result in detections in groundwater in the ppt and parts per billion (ppb) range.

Perfluoroalkyl carboxylates, the class of PFAS containing PFOA, are stable in water, sediment, and soil under natural conditions (Kissa, 2001). As a result, longer-chained perfluoroalkyl carboxylates are not known to degrade into PFOA in these media in the environment. PFOA is itself a degradation product of perfluoro alcohols. The perfluoro alcohols that degrade to PFOA are volatile and may be emitted into the atmosphere as part of various chemical manufacturing processes. Consequently, PFOA is found worldwide at low levels (Strynar et al., 2012; Rankin et al., 2016).

Several studies have quantified the structure-activity relationships between the length of linear PFAS and transport potential through soil columns (Higgins and Luthy, 2006; Zhang et al., 2013; Gellrich et al., 2012; Hirata-Koizumi et al., 2015). Perfluorotetradecanoic acid (PFTeDA) and other longer-chained fluorocarbons also have a much higher partition coefficient with respect to organic carbon, resulting in very low mobility in the soil column which prevents them from migrating to groundwater (which likely explains why they are not detected in deeper soils or groundwater where they were detected in shallow soils). Thus, compared to PFOA, PFTeDA and other long-chain linear PFAS found in organic soils are less likely to affect groundwater because of their inability to percolate through soil columns.

2.4 PFOA Fate and Transport Conceptual Modeling Approach and Results

For purposes of the Draft CSM Report (Barr, 2017c), numerical models were developed to simulate PFOA transport and migration from points of release at the former Chemfab facilities to the groundwater-flow system and private wells. Given the limited amount of hydrogeologic data and some uncertainties in sources of PFOA, the modeling approach constituted a conservative, regional evaluation. The simulations

allowed for a quantitative and physically-based assessment of the mechanisms that could potentially result in detectable PFOA concentrations in soils and groundwater.

For purposes of the Draft CSM Report (Barr, 2017c), the complete PFOA transport pathway was simulated to assess the possibility of transport from source to potential receptors. As indicated above, these transport paths are through air, unsaturated soil and other unconsolidated materials, and saturated aquifer materials. No single model in standard practice exists that can account for transport through these different media. Therefore, transport was simulated by a series of distinct models, with the output of one or more models serving as input for the model simulating the next segment of the transport pathway. Each of these individual models is a widely accepted tool for the transport pathway for which it was applied. Assumptions and limitations in the modeling approach are described in the Draft CSM Report (Barr, 2017c). For purposes of the Draft CSM Report (Barr, 2017c), the only sources of PFOA that were evaluated were air emissions from the two former Chemfab facilities—other local sources and regional or worldwide deposition sources were not part of the modeling evaluation.

The primary findings of the conceptual modeling were that the distribution of PFOA measured in private wells could not be explained solely by air emissions from the former Chemfab facilities, and that such emissions would not have potentially produced PFAS concentrations above 20 ppt in private wells in Corrective Action Area II (CAAI).

3.0 Work Plan Implementation Activities and Results

The Work Plan (Barr, 2017a) was implemented through soil and bedrock boring advancement, temporary and permanent monitoring well installation in select borings, collection, and laboratory analysis of soil and groundwater samples, a bedrock study, groundwater elevation monitoring, and hydraulic conductivity testing. The work was performed to assist in meeting the investigation objectives described in Section 1.1.

Investigative activities commenced following August 12, 2017 approval of the Work Plan by VTDEC (VTDEC, 2017a). The field activities were performed between August 14, 2017 and January 31, 2018, and the field methods were generally consistent with the Work Plan with the exception of minor deviations described in Appendix A and discussed in Section 3.6.

A summary of the investigation locations, soil samples collected, and well construction details are provided in Table 3.1, Table 3.2, and Table 3.3, respectively. Field investigation documentation including soil boring logs, field notes, well construction logs, and photographs from the investigation are provided in Appendix A. Analytical methods are summarized in Table 3.4. Laboratory reports are included in Appendix B, and a summary of QA/QC results is provided in Appendix C.

3.1 Unconsolidated Soil Characterization

Locations of background, shallow and deep unconsolidated investigation borings are shown on Figure 3.1. When considering the shallow and deep soil borings together, borings were placed in an approximate grid pattern across the investigation area. The grid-sampling concept was proposed to allow for flexibility in interpreting results as a function of distance in any direction from a feature of interest (e.g., the former Chemfab facilities and Bennington Landfill). Access restrictions resulted in minor deviations to this approach.

Soil samples were collected from each soil boring for geologic characterization, field screening, and laboratory analysis. Geologic characterization included classifying material using the visual and manual test methods described in ASTM D-2488, Standard Practice for Description and Identification of Soils (Visual/Manual). Field screening included classification of soils in accordance with the methods in the Field Sampling Plan (FSP) included as part of the Work Plan (Barr, 2017a), and visually examining the soil samples; recording observations of moisture, odor, and discoloration; and determining the presence of organic vapors with a photoionization detector (PID) equipped with a 10.6-electron-volts lamp. Laboratory analysis of soil samples was completed at the locations and for the parameters identified in Table 3.2, and was completed in accordance with the Quality Assurance Project Plan (QAPP; C.T. Male, 2016a) and QAPP Addendum included as part of the Work Plan (Barr, 2017a; Barr, 2017d).

In general, soil samples were collected from investigation borings at selected intervals: 0-0.5 feet below ground surface (bgs), 0.5-1 feet bgs, 1-1.5 feet bgs, 3-4 feet bgs, 8-9 feet bgs, and at 10-foot intervals to the terminus of the boring at the water table or bedrock, whichever was shallower. If saturated soil was encountered at a depth of less than 8 feet, a sample was collected from just above the observed groundwater level. If refusal was encountered at a depth of less than 8 feet, a sample was collected at the

terminus of the boring, unless refusal was encountered at depths less than 4 feet bgs, in which case additional attempts were made within the vicinity of the originally proposed location. Additional soil samples were also collected if highly organic soils were encountered or if field screening identified signs of obvious environmental impacts.

Laboratory analytical services were provided for soil samples, grain-size distribution samples, and water samples by Eurofins Lancaster Laboratories Environmental, LLC (ELLE) of Lancaster, Pennsylvania in accordance with the QAPP (C.T. Male, 2016a) and the QAPP Addendum attached to the Work Plan (Barr, 2017a; Barr, 2017d). Laboratory reports are included in Appendix B. The analytical data QA/QC reviews are included in Appendix C. As requested by VTDEC in comments on the revised Work Plan (VTDEC, 2017a), additional soil samples were analyzed for grain-size distribution using ASTM D422 (Standard Test Method for Particle-Size Analysis of Soils) to allow interpretation of soil type and estimation of hydraulic conductivity. The sample intervals for grain-size distribution analysis were selected to represent the range of lithologies observed in the soil borings.

3.1.1 Background Soil Borings

Background soil borings were completed to evaluate local background PFAS soil concentrations. The boring locations were selected in consultation with VTDEC and were placed near sampling locations previously completed by VTDEC during a statewide study of the background concentrations of polycyclic aromatic hydrocarbons (PAHs), arsenic, and lead in surface soils (VTDEC, 2017b). The borings were advanced in Shaftsbury State Park, Woodford State Park, Molly Stark State Park, and at the Lake Raponda boat launch area (Figure 3.1).

The four shallow background soil borings (BG1 to BG4) were advanced using direct-push methods (i.e., Geoprobe®) within the unconsolidated materials and continuously logged to depths ranging from 3.5 to 10 feet bgs. The Work Plan specified collection of an analytical soil sample at a depth of 8 feet bgs or at the interpreted groundwater surface (whichever was shallower). Background soil borings BG1 and BG2 were advanced to 10 feet bgs to obtain a sufficient volume of soil for analysis; boring BG3 encountered refusal at 3.5 feet bgs and multiple additional attempts in the vicinity were unsuccessful; and BG4 encountered groundwater at 8 feet bgs.

No odor, sheen, or discolored soil was observed during field screening of the collected samples. Headspace readings were measured below background at all locations.

A total of 18 soil samples were collected and analyzed for PFAS, TOC, soil moisture, and soil pH (Table 3.5). Soil analytical results from samples collected at background locations are shown on Figure 3.2, Figure 3.3, and Figure 3.4 for PFOA, PFOS, and TOC, respectively. PFOA was detected above the MDL in ten samples at concentrations ranging from 0.37 to 7.2 ng/g, and PFOS was detected above the MDL in four samples at concentrations ranging from 0.42 to 1.1 ng/g. The highest concentrations of PFOA and PFOS (7.2 and 1.1 ng/g, respectively) were detected in a sample collected from surface soils in boring BG2 located near a hiking/skiing trail in Woodford State Park, approximately 7 miles east of the investigation area (Figure 3.2 and Figure 3.3). As shown on Figure 3.5, PFOA and perfluorobutanoic acid (PFBA) were the most frequently detected of the PFAS compounds in samples from background locations.

The concentrations of PFOA and TOC from the four background soil borings were compared to background soil concentrations collected from across North America. The purpose of this evaluation was to assess whether the background samples collected during this investigation were representative of known background conditions in North America. The results of this evaluation are provided in Appendix D.2 and an interpretation of the background soil analytical evaluation is discussed in Section 4.4.1.

3.1.2 Shallow Soil Borings

As shown on Figure 3.1, a total of 46 shallow soil borings (S1 to S14, S11-2, S16, S18, S20 to S22, and S24 to S49) were drilled within the unconsolidated materials using hand auger or direct-push methods (i.e., Geoprobe®) along an approximate grid across the site (when taking into account deep borings and access issues). In general, shallow borings were planned to be advanced to depths of 8 feet bgs or until reaching saturated conditions, whichever was shallower; however, because the Work Plan specified collection of an analytical soil sample at a depth of 8 feet bgs from each shallow boring, final boring depths of up to 10 feet bgs were common in order to obtain a sufficient volume of soil for analysis. Monitoring wells were completed in select shallow borings as described in Section 3.3.1 and shown on Figure 3.1.

No odor, sheen, or discolored soil was observed during field screening at these borings with the exception of boring S02, where both a sheen and odor were noted at a depth of 1.5 feet bgs. Headspace readings were measured above background in soils from seven borings (S02, S11, S11-2, S14, S21, S27, and S38) and PID readings in these borings ranged from 5.3 to 35 parts per million (ppm). Elevated headspace readings at borings S11 and S14 appear to have been sourced by equipment interference (i.e., elevated ambient air readings inside empty headspace bags) and are likely not representative of actual conditions.

A total of 214 soil samples were collected from shallow borings and analyzed for PFAS, TOC, soil moisture, and soil pH (Table 3.6). Due to elevated organic vapor concentrations, petroleum odors, and potential staining in samples from S02 (0.5-1 feet bgs) and S02 (1-1.5 feet bgs), location S02 was resampled and analyzed for VOCs, semivolatile organic compounds (SVOCs), PCBs, and pesticides. Results of these additional analyses show no elevated concentrations to suggest an environmental impact (Table 3.7). Soil analytical results for shallow soil borings are shown on Figure 3.6, Figure 3.7, and Figure 3.8 for site-wide PFOA, PFOS, and TOC, respectively, and on Figure 3.9, Figure 3.10, and Figure 3.11 for Bennington Landfill-specific PFOA, PFOS, and TOC, respectively.

PFOA was detected above the MDL in 120 shallow soil boring samples at concentrations ranging from 0.17 to 130 ng/g, and PFOS was detected above the MDL in 73 samples at concentrations ranging from 0.32 to 17 ng/g. The highest PFOA and PFOS concentrations were detected in samples collected from shallow soils in borings S20 and S29. The samples from these borings were collected from the organic soil and peat in a wetland area east of Bennington Landfill. The TOC concentrations in these samples were also the highest measured during this investigation, with measured concentrations up to 512,000 milligram per kilogram (mg/kg) (Figure 3.8).

3.1.3 Deep Soil Borings

Deep soil borings were located along transects between Bennington Landfill and the former Chemfab facilities and in each cardinal direction (i.e., north, south, east, and west) from the former Water Street facility (Figure 3.1). The locations were intended to target areas that have been undisturbed since the late 1960s; were away from potential PFAS sources that may have been associated with the former Chemfab facilities; and for which access could be obtained. Access restrictions resulted in the majority of soil borings being placed within public rights-of-way along roadways, where disturbance of the shallow soil layers may have occurred since the late 1960s; however, areas of apparent disturbance or fill were avoided, as practicable, as were locations of potential deep excavations or thick fill sequences (e.g. Vermont State Highway 279).

A total of 25 deep soil borings (D01 to D03 and D05 to D26) were drilled within the unconsolidated materials to bedrock or the water table (whichever was shallower) using rotasonic methods and logged continuously to depths ranging from 10 to 200 feet bgs. Four of the scoped deep borings (D06, D08, D14, and D19) were completed to shallow depths (between 10 and 20 feet bgs) after groundwater and/or bedrock was encountered. Monitoring wells were completed in select deep borings as described in Section 3.3.2 and shown on Figure 3.1.

No odor, sheen, or discolored soil was observed during field screening for these borings. Headspace readings were measured above background (i.e., 5.0 ppm) in soils from seven borings (D03, D05, D06, D07, D10, D11, D20, and D23) and PID readings in these borings ranged from 5.3 to 23 ppm. Elevated headspace readings at borings D03 and D05 appear to have been sourced by equipment interference (i.e., elevated ambient air readings inside empty headspace bags) and are likely not representative of actual conditions.

A total of 336 soil samples were collected from deep soil borings and analyzed for PFAS, TOC, soil moisture, and soil pH (Table 3.8). Of these 336 soil samples, 190 samples from deep borings were also analyzed for target-analyte-list (TAL) metals. Due to elevated organic vapor concentration and potential staining in sample D20 (78 to 79 feet), this sample was also analyzed for VOCs, SVOCs, PCBs, and pesticides (Table 3.7). Soil analytical results from samples collected from deep borings are shown on Figure 3.6, Figure 3.7, and Figure 3.8 for site-wide PFOA, PFOS, and TOC, respectively and on Figure 3.9, Figure 3.10, and Figure 3.11 for Bennington Landfill-specific PFOA, PFOS, and TOC, respectively.

PFOA was detected above the MDL in 80 samples at concentrations ranging from 0.16 to 7.2 ng/g and PFOS was detected above the MDL in 34 samples at concentrations ranging from 0.28 to 16 ng/g. The highest PFOA concentrations were detected in samples collected from shallow soils in boring D05 located approximately 0.7 miles north of the former Water Street facility. The highest PFOS concentrations were detected in samples collected from shallow soils in borings D08 and D10 located at Bennington Landfill.

3.1.4 Comparison of Results by Depth and Area

To meet the site-specific investigation objectives of evaluating relationships and trends in PFAS distribution (specifically in relation to PFOA and PFOS) and identifying some of the other potential sources

of PFAS within the investigation area, analytical results collected as part of this Work Plan were evaluated by depth over different soil depth intervals (e.g., surface, shallow, and deep) and by area for the landfill and non-landfill areas, as described in Section 3.1.4.1 below.

A synthesis of soil and groundwater data collected adjacent to the former Chemfab Water Street facility and away from the former Water Street facility area (i.e., Water Street and non-Water Street) is provided in Section 4.4.2. A discussion is not provided in this section, Work Plan Implementation Activities and Results, as the Water Street-specific data was collected as part of a separate investigative effort (C.T. Male, 2016a).

3.1.4.1 Comparison of Shallow to Deep Results

The comparison of soil results across depth intervals is informed by the PFAS detection frequency plots shown on Figure 3.12, Figure 3.13, and Figure 3.14. Figure 3.12 shows that PFOA is the most frequently detected PFAS compound in both shallow (0-5 feet) and deep (>5 feet) soils across the entire investigation area, with just over half of shallow samples having detectable concentrations of PFOA. The figure also shows a clear difference in the frequency of detection between shallow and deep soils for all of the PFAS compounds analyzed for the investigation. Figure 3.13 shows that at Bennington Landfill, there is a pronounced difference in detection frequency of PFAS parameters within the shallow soil layer, with detections in the upper 1.5 feet of the soil layer being roughly twice as frequent as detections in the depth interval 1.5-5 feet bgs. Outside of the landfill (Figure 3.14), there is no notable difference in PFOA detections between surface soil (upper 1.5 feet) and shallow soil from 1.5 -5 feet bgs for PFOA, but there is a marked decrease in the frequency of detection between surface and deeper soils for other PFAS parameters outside the landfill.

3.1.4.2 Comparison of Landfill to Non-Landfill Results

Shallow and deep unconsolidated borings completed in Bennington Landfill and outside the landfill were sampled and analyzed as described above. Analytical data discussed above and summarized on Table 3.6 and Table 3.8 for shallow and deep soil borings, respectively, are also presented in Table 3.9 and Table 3.10 for non-landfill and landfill locations, respectively.

A total of 318 soil samples were collected from outside the landfill and analyzed for PFAS, TOC, and soil pH (Table 3.9) as part of this investigation. Of these 318 soil samples, 126 samples from deep soil borings were also analyzed for TAL metals. Soil analytical results are shown on Figure 3.6, Figure 3.7, and Figure 3.8 for PFOA, PFOS and TOC, respectively. PFOA was detected above the MDL in 151 samples at concentrations ranging from 0.18 to 12 ng/g. PFOS was detected above the MDL in 71 samples at concentrations ranging from 0.28 to 4.4 ng/g. The highest PFOA concentrations were detected in samples collected from shallow soils in boring S33 located along Harwood Hill Road (Route 7), approximately 1.7 miles east of the former Water Street facility (Figure 3.6) and ranged from 10 to 12 ng/g. The highest PFOS concentration (4.4 ng/g) was detected in a sample collected from surface soils in boring S46, located in the southern portion of the investigation area (Figure 3.7).

A total of 232 soil samples were collected from the landfill and analyzed for PFAS, TOC, and soil pH (Table 3.10). Of these 232 soil samples, 175 samples from deep soil borings and bedrock monitoring well

borings were also analyzed for TAL metals. Soil analytical results from samples collected at the landfill are shown on Figure 3.9, Figure 3.10, and Figure 3.11 for PFOA, PFOS and TOC, respectively. PFOA was detected above the MDL in 54 samples at concentrations ranging from 0.16 to 130 ng/g and PFOS was detected above the MDL in 39 samples at concentrations ranging from 0.28 to 17 ng/g. The highest PFOA and PFOS concentrations were detected in samples collected from shallow soils in borings S20 and S29. The samples from these borings were collected from the organic soil and peat in a wetland area east of the landfill. The TOC concentrations in these samples were also the highest measured during this investigation.

While PFOA and PFOS were the most frequently detected of the PFAS compounds in samples from both landfill and non-landfill locations (Figure 3.15), PFOA was detected approximately twice as frequently in non-landfill samples as it was in landfill samples (47% and 24% of samples, respectively), reflective of elevated soil PFOA concentrations in CAAI discussed in Section 4.3.1.2. Similarly, PFOS and other non-PFOA PFAS parameters were detected at a similar frequency in both non-landfill and landfill locations, which is consistent with an elevated PFOA source outside of the landfill. In addition, within the landfill, the difference in the frequency of detections between PFOA and PFOS is less pronounced at each depth interval than it is outside the landfill. Within the landfill (Figure 3.13), PFOA and PFOS were detected at 58% and 47%, respectively, of surface soil samples; 37% and 21%, respectively, of shallow soil samples; and 7% and 3%, respectively, of deep soil samples. Outside of the landfill (Figure 3.14), PFOA and PFOS were detected at 64% and 44%, respectively, of surface soil samples; 63% and 6%, respectively, of shallow soil samples; and 18% and 1%, respectively, of deep soil samples.

3.2 Bedrock Characterization

To assess bedrock hydrogeologic conditions beneath and downgradient of Bennington Landfill, a bedrock desktop review and outcrop study was conducted to guide the placement of bedrock borings (shown on Figure 3.1). After the boreholes were drilled, downhole logging was completed to further assess bedrock conditions using geophysical logging and hydrogeologic flow logging. Discrete interval sampling was completed to sample groundwater intervals of interest based on the results of the downhole logging.

3.2.1 Bedrock Desktop Review and Outcrop Study

A desktop review of publicly available information regarding the bedrock geology, an outcrop mapping analysis, and a photo lineament analysis were performed for the area around Bennington Landfill. The discussion below provides a summary of the study methods and results that are discussed in detail in Appendix E.1. The objectives of this work were to:

- guide placement of bedrock borings and monitoring wells near the landfill along potential preferential groundwater flow paths;
- verify recent Vermont Geological Survey (VGS) geologic mapping of the area;
- collect geologic, hydrogeologic, and rock mass discontinuity data at the outcrop-level scale; and
- collect lineament trend data at a larger scale.

Candidate outcrop locations were identified through the desktop review of the recent geologic mapping by VGS (e.g., Kim, 2017), the Remedial Investigation Report documents for Bennington Landfill (McLaren/Hart, 1997), and other regional geologic reports. Candidate outcrop locations were further screened using Google Earth's Street View to identify outcrop locations that were publicly accessible.

Outcrops of the Cambrian Cheshire Formation quartzite were identified and quartz veins filling joints were noted at the outcrops. Ogden (1969) described quartz veins in the Cheshire Formation quartzite associated with the kaolin deposits in the Cheshire Formation near Monkton, Vermont. Ogden (1969) also noted that several kaolin deposits have been identified along the western base of the Green Mountains from Bennington to East Monkton. All of the deposits are in a zone of faulting. Kaolin in the Cheshire Formation quartzite, arising from hydrothermal alteration, appears to be consistent with available information regarding the stratigraphic position and chemical composition of these deposits. Previous work at Bennington Landfill indicated weathering as the source of alteration of bedrock beneath the landfill. For example, McLaren/Hart (1997) described a weathered zone on the bedrock ranging from 0 to greater than 150 feet thick and classified it as a saprolite (clay-rich rock formed by chemical weathering of igneous or metamorphic rock; AGI, 1976).

The outcrop mapping analysis consisted of photographing the outcrops studied, compiling lithologic descriptions, taking representative lithologic samples, and mapping discontinuities in the rock masses. Discontinuity types included bedding-plane fractures, joints, faults, and mineralized veins. Discontinuity characteristics documented include the type, orientation, persistence, aperture, infilling material (if present), shear strength of infilling material, surface roughness, shape, presence of water, and spacing. The outcrop study identified one bedding plane discontinuity set and two primary joint sets. Bedding-plane strikes are predominantly northeasterly, with dips both toward the southeast and northwest at angles ranging from 3 to 32 degrees. The field measurements are consistent with a broad, open anticline mapped by VGS (Kim, 2017) with its axis located west of the landfill. Joint sets J1 and J2 have azimuths of approximately 21 and 306 degrees azimuth. Joint and bedding-plane discontinuity spacing ranges from 0.04 feet to 5 feet. Joint and bedding-plane persistence (lateral and vertical continuity) on the outcrops examined ranges up to 50 feet. Bedding-plane apertures average 0.3 inches and the apertures of joint sets J1 and J2 average 0.7 and 1.2 inches, respectively.

The photo lineament analysis identified possible joint sets at the regional level and compared their orientations with those measured in the outcrop study with the results of a similar study conducted for the investigation of the landfill (McLaren/Hart, 1997). An area around Bennington Landfill with an approximate radius of 12,500 feet was studied as part of the desktop review and outcrop study. The photo lineament analysis identified two predominant lineament directions: 44 and 311 degrees azimuth. These lineaments correspond with the J1 and J2 joint sets identified in the outcrop study. Spacing between these lineaments ranges from 50 to 2,200 feet and 100 to 3,700 feet, respectively.

Representative orientations of the J1 and J2 joint sets, based on combined information from the outcrop and lineament analyses, are 33 and 310 degrees azimuth, respectively. These orientations are shown graphically in Appendix E.1 and are projected from the limits of the capped portion of Bennington Landfill on Figure 3.16 to show alignment of the joint sets relative to investigation locations in the landfill area.

3.2.2 Bedrock Borings

Four bedrock borings (SG3-MW17-BR1 to SG3-MW17-BR4) were drilled at the locations encompassing the Bennington Landfill shown on Figure 3.16. The primary joint sets, identified in the bedrock study described in the previous section and in Appendix E.1, and the alignment of these joint sets served as a guide for bedrock boring siting (Figure 3.16). Final locations were contingent on access and were shifted from planned locations within the joint alignments slightly. The borings were advanced through the unconsolidated materials, altered bedrock, and into the competent bedrock to depths ranging from 263 to 459 feet bgs. (A discussion of altered and competent bedrock is provided in Section 4.1.2.2 and Section 4.1.2.3, respectively.) The bedrock borings were completed as monitoring wells as described in Section 3.3.3.

The borings were drilled through the unconsolidated materials using rotasonic methods and were logged continuously until competent (unaltered) bedrock was encountered (at depths ranging from 106 to 397 feet bgs). Upon encountering competent bedrock, borings were then drilled approximately ten feet into the bedrock and a 6-inch-diameter steel casing was installed and grouted into place. Within the grouted casing, the borings were further drilled into bedrock using a combination of methods including rotasonic (all borings), air rotary with downhole hammer (borings SG3-MW17-BR2 and SG-MW17-BR3), and diamond bit coring (borings SG3-MW17-BR4 and approximately the upper 21 feet of SG3-MW17-BR1). Total depths of borings were targeted to be consistent with nearby drinking water wells (generally 150-400 feet bgs), to the extent that depths of bedrock alteration allowed such consistency. Logging of bedrock materials included descriptions of interpreted lithology, mineralogy, and potential fracturing based on visual observation of the bedrock core or drill cuttings.

The encountered bedrock lithologies were consistent with the Cheshire and Dunham Formations. The Cheshire Formation was encountered in bedrock borings SG3-MW17-BR1, SG3-MW17-BR2 and SG3-MW17-BR3, and the Dunham Formation was encountered in bedrock boring SG3-MW17-BR4.

Hydrothermally altered rock (described in Section 4.1.2.2) was encountered at bedrock borings SG3-MW17-BR1 and SG3-MW17-BR4, located north and east of the landfill, respectively. The hydrothermally altered rock was observed from 130 to 304 feet bgs in bedrock boring SG3-MW17-BR1 and from 192 to 392 feet bgs in bedrock boring SG3-MW17-BR4. Hydrothermally altered bedrock was also encountered in SG3-MW17-BR1 below 304 feet bgs. As noted on the boring log for this well included in Appendix A, the hydrothermally altered bedrock was not recovered but was inferred from "soft" zones for drilling and brown-colored return water. The low recoveries noted on the log and the brown-colored return water support that the altered zones were likely washed out during drilling. Hydrothermally altered bedrock was observed in SG3-MW17-BR4 to a lesser extent than SG3-MW17-BR1. As noted on the boring log for SG3-MW17-BR4, discrete zones with vugs and pitting moderately healed with quartz and some fractures/joints filled with kaolin were observed.

3.2.3 Borehole Characterization

Borehole characterization activities were completed in the four new bedrock wells (SG3-MW17-BR1, SG3-MW17-BR2, SG3-MW17-BR3, and SG3-MW17-BR4), and in existing Bennington Landfill bedrock

monitoring wells B-2-3, B-4-3, and B-6-3. Well construction records, boring and coring logs, and a construction summary for the existing landfill bedrock wells are included in Appendix E.2. This characterization consisted of downhole geophysical logging and flow logging to supplement geologic log information and to identify significant intervals of flow for discrete interval groundwater sampling.

The methods and locations of borehole characterization activities, including geophysical logging methods, flow logging, and discrete interval sampling, are summarized in Table 3.11. Reasons for partial data collection at select wells are discussed below. Results of the borehole characterization activities are discussed below and presented in detail in an investigation report included in Appendix E.3.

As discussed in Section 3.3.3 below, borehole instability during drilling of wells SG3-MW17-BR1 and SG3-MW17-BR4 required installation of 4-inch-diameter polyvinyl chloride (PVC) risers and screens to complete the boreholes. Due to the presence of the PVC, only a partial suite of geophysical logs could be completed in the wells as listed in Table 3.11. The screened interval in well SG3-MW17-BR1 is approximately 299 to 369 feet bgs. The screened interval in well SG3-MW17-BR4 is approximately 409 to 459 feet bgs.

Borehole instability was also encountered in SG3-MW17-BR3 after the first discrete interval sample was collected. The borehole was stabilized by installing 4-inch-diameter PVC riser and screen, which prevented completion of the additional discrete interval sampling planned at this well. The screened interval in well SG3-MW17-BR3 is approximately 112 to 272 feet bgs. Only a partial set of geophysical logs, as shown in Table 3.11, was collected from the existing Bennington Landfill bedrock monitoring wells (B-2-3, B-4-3, and B-6-3) due to their completion with 2-inch-diameter PVC riser and 15-foot-long well screens, limiting the downhole tool diameters. Logging of wells B-2-3, B-4-3, and B-6-3 was performed from the ground surface to the depth at which there was no longer sufficient clearance for the logging tools to pass; deviations in PVC riser straightness prevented the tool from reaching the bottom of the well. Logging depths for these wells are noted in Table 3.11.

3.2.3.1 Geophysical Logging

The geophysical logging methods consisted of natural gamma, resistivity, spontaneous potential, caliper, and optical and acoustic televiwer borehole imaging; and the fluid logging methods consisted of temperature, resistivity and conductivity. Borehole geophysical logging results are presented in Appendix E.3. The primary findings are discussed below in relation to the intended use and utility of the logs to characterize site hydrogeology.

The geophysical logs for wells SG3-MW17-BR1, SG3-MW17-BR2, SG3-MW17-BR3, and SG3-MW17-BR4 were compared to the corresponding field boring logs to identify any geophysical signatures of the lithology encountered. The comparisons indicate that the relationship between field logging descriptions and borehole geophysical logging results is generally consistent between wells. Changes in lithology noted in the field are generally consistent with changes in the gamma, spontaneous potential, and resistivity logs. For example, the gamma response of unaltered quartzite (based on field identification) is typically lower than the gamma response in zones of altered bedrock or in zones of phyllite (a rock

enriched in clay minerals) (Figure 3.17 and Figure 3.18Figure 3.20). This correlation is consistent with hydrothermal alteration of the bedrock, which results in the formation of naturally radioactive kaolin.

The geophysical logs for the existing bedrock monitoring wells B-2-3, B-4-3, and B-6-3, which were limited in number due to the completion of the wells with 2-inch-diameter PVC and relatively short screens, showed little consistency between log response and lithology. The natural gamma log is the only log performed in these wells that could have interpretive value, as the resistivity signal would not penetrate the steel well casing and would be affected by the PVC well screen. Gamma responses for B-2-3 and B-6-3 do not exhibit much similarity with the gamma responses for either SG3-MW17-BR2 or SG3-MW17-BR3. Zones interpreted as sands and clays above the bedrock in B-2-3 and B-6-3 do not have corresponding responses at similar depths in SG3-MW17-BR2 or SG3-MW17-BR3. Gamma responses in B-4-3 are of similar overall trend to that of SG3-MW17-BR2 at depths of approximately 25 feet and below, but do not correlate with responses for SG3-MW17-BR3.

The borehole imaging logs for wells SG3-MW17-BR2 and SG3-MW17-BR3 were reviewed to identify planar features (i.e., bedding, foliation, joints/fractures) intersecting the open boreholes. The borehole imaging logs are spatially oriented so the logs can be used to determine orientations of the fracture planes. A total of 232 planar features were identified intersecting the open borehole in SG3-MW17-BR2 and 204 planar features were identified intersecting the open borehole in SG3-MW17-BR3 (Appendix E.4). Approximately 45% of the planar features intersecting well SG3-MW17-BR2 are consistent with the orientation of joint set J1 that was identified in the outcrop mapping and lineament analysis described in Appendix E.1 and summarized in Section 3.2.1. Approximately 1% of the planar features intersecting SG3-MW17-BR2 are consistent with the orientation of joint set J2 that was identified in the outcrop mapping and lineament analysis. Approximately 34% of the planar features intersecting well SG3-MW17-BR3 are consistent with the orientation of joint set J1. Approximately 3% of the planar features intersecting SG3-MW17-BR3 are consistent with the orientation of joint set J2 that was identified in the outcrop mapping and lineament analysis.

Similarities and differences between the joint set orientations mapped in outcrops and the orientations determined from the borehole imaging logs are presented in Appendix E.5. The reader should note when reviewing the information in Appendix E that structural orientation data from the borehole geophysical logging is presented in upper hemisphere stereonet (Appendix E.3), while the structural orientation data in the comparison of the outcrop mapping results to the borehole geophysical logging results is presented in lower hemisphere stereonet (Appendix E.5).

Due to the well screens and risers in SG3-MW17-BR1, SG3-MW17-BR4, B-2-3, B-4-3, and B-6-3, no imaging logs useful for evaluating the presence and orientation of fractures intersecting the boreholes could be collected at these locations.

3.2.3.2 Flow Logging

Borehole flow logging was performed using an electromagnetic flow meter trolling up and down in the borehole under ambient (non-pumping) and dynamic (pumping) conditions. Results from flow logging are presented in Appendix E.3 and discussed below in relation to the intended use and utility of the logs

to characterize site hydrogeology. As discussed in Appendix E.3, the flow logging results for wells SG3-MW17-BR2 and SG3-MW17-BR3 did not identify upward or downward flow in the wells under ambient conditions.

The borehole geophysical logs and flow logging results for wells SG3-MW17-BR2 and SG2-MW17-BR3 were used to identify potentially more hydraulically active zones within the open borehole portions of the wells. Flow, fluid temperature, and fluid conductivity logging data indicate that there was not significant vertical flow within the open borehole of either well under ambient conditions, as discussed in Appendix E.3. This suggests that flow through the borehole is predominantly lateral.

An analysis of the flow, fluid temperature, and fluid conductivity logging data collected under both ambient and pumping conditions for well SG3-MW17-BR2 indicates the presence of five intervals considered to be more hydraulically active than other zones within the well (Figure 3.19Figure 3.20). These intervals are identified on Figure 3.19Figure 3.20 and in Appendix E.3. The more hydraulically active intervals are in the following depth intervals: 259 to 264 feet bgs, 250 to 254.5 feet bgs, 204 to 217 feet bgs, 167 to 172 feet bgs, and 148 to 151 feet bgs. These more hydraulically active intervals appear to include groups of bedding planes and joints in the J1 joint set. As discussed in Section 3.2.3.1, there are very few joints in the J2 joint set that intersect SG3-MW17-BR2, and these joints do not appear to be hydraulically active. As shown in Appendix E.3, essentially no vertical flow (i.e., flow rates were less than 0.1 gallons per minute or gpm) under ambient flow logging conditions was measured in SG3-MW17-BR2. Under dynamic flow logging conditions, measured flows in the more hydraulically active intervals ranged from less than 0.1 gpm to 1.7 gpm

A total of six more hydraulically active intervals were identified in well SG3-MW17-BR3 based on the flow, fluid temperature, and fluid conductivity logging data collected under both ambient and pumping conditions (Figure 3.20). These intervals are identified on Figure 3.20 and in Appendix E.3. The more hydraulically active intervals are in the following depth intervals: 246.0 to 248.0 feet bgs, 223.5 to 226.0 feet bgs, 205.0 to 210.0 feet bgs, 188.2 to 192.7 feet bgs, 167.8 to 174.0 feet bgs, and 147.4 to 158.5 feet bgs. These more hydraulically active intervals appear to include groups of bedding planes and joints in the J1 joint set. As discussed in Section 3.2.3.1, there are very few joints in the J2 joint set that intersect SG3-MW17-BR3, and these joints do not appear to be hydraulically active. As shown in Appendix E.3, essentially no vertical flow (i.e., flow rates were less than 0.2 gpm) under ambient flow logging conditions was measured in SG3-MW17-BR3. Under dynamic flow logging conditions, measured flows in the more hydraulically active intervals ranged from 2.0 gpm to 6.0 gpm.

3.2.3.3 Discrete Interval Sampling

The borehole geophysical logs and flow logs for wells SG3-MW17-BR2 and SG3-MW17-BR3 were used to select discrete intervals for groundwater sampling using straddle packers. As shown in Table 3.12, four zones in well SG3-MW17-BR2 were selected for discrete interval sampling, and six zones were selected for sampling in well SG3-MW17-BR3 (Table 3.12).

The testing procedure called for selected intervals to be isolated using inflatable packers set above and below the interval. If the selected target interval extended up to the water surface in the well, one packer

set at the bottom of the interval was used to isolate the interval for testing. If the selected target interval extended to the bottom of the well, one packer set at the top of the interval was used to isolate the interval for testing. During installation of the packers and pumping in the selected interval, In-Situ Level TROLL 300 pressure transducers were used to monitor heads below the bottom packer (when used), in the test interval, and above the top packer (when used). Depth to water was also measured manually during the discrete interval sampling to provide the real-time data that could not be obtained from the pressure transducers due to the inability to use data cables out of concern of PFAS cross contamination. Representative depths to water for packed intervals are summarized in Table 3.12.

Heads monitored above and below the test interval, as applicable, were reviewed upon retrieval of the transducers to determine if the packers had sufficiently isolated the test interval from the rest of the borehole. If the water pressures above and below the test interval did not change appreciably (0.1 foot or less) during pumping within the test interval, the packers were determined to adequately isolate the test interval. This range is below the accuracy of the transducers used in the packer tests. For some intervals, the water pressure above or below the test interval changed by more than 0.1 foot (Appendix E.3) and was attributed to hydraulic connections around the borehole through fractures rather than inadequate packer inflation. Groundwater sampling within the discrete intervals followed the standard well sampling protocol outlined in the FSP, i.e., a minimum of three well volumes was purged from the isolated interval prior to stabilization and groundwater sampling.

The first interval tested in well SG3-MW17-BR3 was from the water surface in the well to 161 feet bgs. After this discrete interval sample was collected and as the packer assembly was being removed, some lithologic material in the open borehole portion of the well collapsed onto the packer assembly, causing it to become stuck. The packer assembly was ultimately freed but no additional discrete interval sample collection was attempted in the well out of concern that additional borehole collapse would cause the equipment to become irretrievably stuck. A 4-inch-diameter PVC screen and riser was installed to keep the borehole open. The screened interval in well SG3-MW17-BR3 is from a depth of approximately 112 feet to approximately 272 feet.

Due to the presence of the well screens and risers, no discrete interval sampling was completed in wells SG3-MW17-BR1 and SG3-MW17-BR4.

Analytical results from the discrete interval groundwater sampling are summarized in Table 3.14. The PFAS results for the discrete interval samples are generally the same as the results for the standard well samples collected following borehole development and logging. For SG3-MW17-BR2, the PFOA concentrations for the discrete interval samples range from 140 to 170 nanograms per liter (ng/L) and the concentrations for the standard samples are 160 and 180 ng/L. The discrete interval sample from 131 to 161 feet bgs in SG3-MW17-BR3 had a PFOA concentration of 72 ng/L, while the standard samples had concentrations between 62 and 67 ng/L. The general agreement between discrete interval samples and standard well samples is attributed to either vertical mixing of concentrations in the bedrock aquifer or limitations of the discrete interval sampling approach due to fracturing in the surrounding bedrock resulting in a hydraulic connection around the packers during sampling.

3.3 Groundwater Characterization

Groundwater samples were collected from permanent and temporary unconsolidated monitoring wells installed as part of the investigation, previously existing Bennington Landfill bedrock and unconsolidated monitoring wells, and newly installed Bennington Landfill area bedrock monitoring wells. A leachate sample also was collected from the Bennington Landfill leachate vault. A summary of the groundwater sampling locations, including well construction details, is provided in Table 3.3. Locations of shallow (temporary) monitoring wells, deep (temporary and permanent) and bedrock (permanent) monitoring wells are shown on Figure 3.1. Additional supporting information associated with the monitoring well installation is included in Appendix A. Permanent monitoring wells constructed during this investigation were installed by a Vermont-licensed well contractor and were constructed in accordance with VTDEC well code. Prior to sampling, monitoring wells were developed using a combination of surging and pumping to remove fines from the well screen.

Groundwater samples were collected following methods described in the FSP. Groundwater samples were analyzed for PFAS, TAL metals/cations (excluding cyanide and mercury), anions, ammonia, orthophosphate, and total Kjeldahl nitrogen. Groundwater samples collected within the Bennington Landfill area were also analyzed for cyanide, mercury, VOCs, and 1,4-dioxane. The analytical methods for each of the analyte groups for groundwater were consistent with those listed in the Work Plan and Table 3.4. The analytical method for PFAS was a modified version of EPA Method 537, to allow for the analysis of other matrices.

Following installation and sampling of wells completed as part of this investigation, hydraulic conductivity testing was completed, and the groundwater elevation was measured. Hydraulic conductivity testing consisted of slug testing, evaluation of grain-size distribution samples, and specific capacity testing. Groundwater elevations were collected using manual measurements and data-recording pressure transducers.

Groundwater analytical results for groundwater samples collected at Bennington Landfill and outside of Bennington Landfill are summarized in Table 3.13 and Table 3.14, respectively. Non-landfill groundwater analytical data are shown on Figure 3.21 and Figure 3.22 for PFOA and PFOS, respectively, and Bennington Landfill groundwater data are shown on Figure 3.23 and Figure 3.24 for PFOA and PFOS, respectively. Frequency of PFAS detections in groundwater by depth are provided on Figure 3.25, and frequency of PFAS detections in groundwater by area are provided on Figure 3.26.

3.3.1 Shallow Unconsolidated Monitoring Well Sampling

One groundwater sample was collected from a temporary well installed at a background location (BG4). Groundwater was not encountered at any of the other three background locations over the depth of the shallow soil boring. Only PFOA and perfluoroheptanoic acid (PFHpA) were detected in the temporary monitoring well installed in boring BG4, at concentrations of 6 and 0.9 J ng/L, respectively (Table 3.13).

Groundwater analytical results for the shallow unconsolidated samples are summarized in Table 3.13 and Table 3.14 for non-landfill and landfill locations, respectively. Temporary wells were installed in 14 shallow

soil borings (S01, S03, S04, S08, S10, S16, S21, S24, S28, S34, S37, S41, S42, and S49) and groundwater samples were collected from each well screened at or across the water table. PFOA was detected in all samples from shallow boring temporary wells at concentrations ranging from 5 to 850 ng/L. PFOS was detected in eight samples at concentrations ranging from 2 J to 23 ng/L. The highest groundwater concentration of PFOA was detected in temporary well S34 located on the eastern side of CAII, along the Walloomsac River and north of Vermont Route 279 (VT 279) (Figure 3.1). The highest groundwater concentration of PFOS was detected in temporary monitoring well S03 located beyond the northern boundary of CAII, near Shaftsbury (Figure 3.1). Detection frequency plots show that PFOA, PFHpA, and perfluorohexanoic acid (PFHxA) were the most frequently detected PFAS compounds in groundwater samples collected from shallow unconsolidated wells (Figure 3.25).

3.3.2 Deep Unconsolidated Monitoring Well Sampling

Groundwater samples were collected from the deep unconsolidated groundwater at temporary wells installed in two borings (D03, D13) and from eight deep soil borings after they were converted into permanent monitoring wells (SG3-MW17-01 to SG3-MW17-07, SG3-MW17-WWTP). (Monitoring well SG3-MW17-WWTP was completed in fractured bedrock directly underlying fine-grained unconsolidated materials. For purposes of this report, it is considered a deep unconsolidated monitoring well for the evaluation of unconsolidated aquifer water levels and water quality.) The eight permanent monitoring wells completed in deep soil borings were screened at or across the water table. Note that for purposes of this report, the soil borings in which permanent monitoring wells were installed typically have a different name than their associated monitoring well. For example, monitoring well SG3-MW17-01 was constructed in soil boring D06. The relationship between soil boring and monitoring well names is summarized in Table 3.1.

Groundwater analytical results for the deep unconsolidated monitoring well samples are summarized in Table 3.13 and Table 3.14 and shown on Figure 3.21, Figure 3.22, Figure 3.23 and Figure 3.24. PFOA was detected in samples from all of the deep unconsolidated groundwater wells at concentrations ranging from 8 J to 240 ng/L. PFOS was detected in seven of the eight deep unconsolidated groundwater wells at concentrations ranging from 2 J to 10 ng/L.

The highest concentrations of PFOA and PFOS in the deep unconsolidated groundwater (240 and 10 ng/L, respectively) were at temporary well D03, located 1.2 miles north of the former Water Street facility. Detection frequency plots show that PFOA, PFHxA, PFHpA, PFNA, PFOS, and perfluoropentanoic acid (PFPeA) were the most frequently detected PFAS compounds in samples from deep unconsolidated wells (Figure 3.25).

3.3.3 Bedrock Monitoring Well Construction and Sampling

Bedrock monitoring well designs were based on the results of the geological and geophysical logging. Bedrock well SG3-MW17-BR2 was completed with the uncased portion left as an open interval. Due to borehole collapse, bedrock wells SG3-MW17-BR1, SG3-MW17-BR3, and SG3-MW17-BR4 could not be completed as open hole wells and were completed as monitoring wells with a 4-inch-diameter PVC screen and riser. This diameter is larger than the 2-inch-diameter specified in the Work Plan in order to

accommodate the downhole geophysical logging tools. The hole was allowed to collapse around the PVC screen in these wells and no filter pack or grout was installed in the borehole annulus. The well construction records for the new monitoring wells are included in Appendix A.

The bedrock wells were completed, developed, and sampled in the following manner:

- Well completion consisted of a protective casing extending approximately three feet above ground surface with locking caps.
- Wells were developed by over pumping and surging. Wells drilled using the down hole hammer (SG3-MW17-BR2 and SG3-MW17-BR3) were air lifted prior to over pumping. Development was completed to produce sediment-free water (turbidity less than 50 Nephelometric Turbidity Unit or NTU).
- Groundwater samples were collected from the developed wells prior to geophysical logging and after borehole logging was complete.
- In addition to measurements completed during drilling, depth to water was measured following well development and prior to sample collection. Additional depth to water measurements made as part of groundwater level monitoring are described in Section 3.3.6.

In addition to the sampling following well development and borehole logging described above, discrete interval groundwater sampling was completed at bedrock well SG3-MW17-BR2 and attempted at SG3-MW17-BR3. The discrete intervals were chosen based on a review of the geophysical and flow logs; see Section 3.2.3.3 for additional discussion.

Groundwater analytical results for the bedrock samples are summarized in Table 3.14 and shown on Figure 3.23 and Figure 3.24. PFOA was detected in samples from all four of the bedrock groundwater wells at concentrations ranging from 0.4 J to 180 ng/L. PFOS was detected in two of the bedrock groundwater wells at concentrations ranging from 0.9 J to 3 J ng/L. The highest concentrations of PFOA and PFOS in the bedrock groundwater (180 and 3 J ng/L, respectively) were at bedrock well SG3-MW17-BR2, located directly downgradient (southwest) of the toe of the landfill (Figure 3.23).

3.3.4 Bennington Landfill Groundwater Split Sampling

Groundwater samples from a subset of existing Bennington Landfill monitoring wells were obtained by Saint-Gobain representatives as sample splits during a groundwater sampling event performed by VTDEC/EPA representatives on May 24 and 25, 2017. This monitoring round is described in the *Final Technical Report for the Bennington Municipal Sanitary Landfill Site* (Weston Solutions, 2017). Purging and stabilization of each monitoring well was performed by the VTDEC/EPA representative, and laboratory sample containers were filled by Saint-Gobain representatives and shipped to the appropriate laboratory for analysis (ELLE or Test America). Groundwater samples were analyzed for the parameters listed in Table 3.4 and were also analyzed for PFAS precursors using the Total Oxidizable Precursor (TOP) Assay methodology (Appendix F).

Samples were collected from non-bedrock monitoring wells at the landfill and from the leachate vault (VLT-01). Sample identification and sampling rationale are presented below, and sample locations are shown on Figure 3.16.

- VLT-01: leachate vault
- B-2-1: shallow well adjacent to leachate vault
- B-2-2: intermediate depth well adjacent to leachate vault
- B-12: previously sampled in 2016, with elevated results
- MW-07: not previously sampled well west of landfill
- MW-100S: shallow monitoring well completed by VTDEC/EPA in 2017

Attempts to collect a sample from monitoring well B-6-1, located beneath the landfill cap, were unsuccessful, due to an insufficient volume of water for sample collection.

Results for the split sampling are summarized in Table 3.14 and shown on Figure 3.23 and Figure 3.24 and include:

- PFOA was detected in all of the samples at concentrations ranging from 16 J ng/L at MW-07 to 4,700 ng/L at the leachate vault (VLT-01)
- PFOS was detected in five of the six samples (all except MW-07) at concentrations ranging from 4 J ng/L at MW-100S to 35 J ng/L at the leachate vault
- VOCs were not detected at concentrations above the laboratory reporting limits
- Chloroethane, ethyl benzene, and xylene were detected at concentrations near the reporting limits in the landfill leachate.

Additional discussion and analysis of the split sampling results is provided in Section 4.3.2 and Section 4.4.3.

3.3.5 Hydraulic Conductivity

Hydraulic conductivity testing was performed to provide a data set to improve the understanding of hydrogeologic conditions in the investigation area and to compare with values of hydraulic conductivity that were used in the conceptual modeling, as described in the Draft CSM Report (Barr, 2017c). For purposes of this investigation, hydraulic conductivity testing methods included slug testing of the new monitoring wells completed as part of the investigation, analysis of grain-size distribution data for soil samples, and analysis of specific capacity information collected as part of dynamic flow logging. Hydraulic conductivity testing and data analysis procedures for the three methods employed for purposes of this investigation are described in Appendix G. A summary of hydraulic conductivity results is provided in Table 3.15.

Slug testing was conducted in the new monitoring wells SG3-MW17-01, SG3-MW17-02, SG3-MW17-03, SG3-MW17-04, SG3-MW17-06, SG3-MW17-07, and SG3-MW17-WWTP. The process involves rapidly raising or lowering the water level in a well by inserting or removing a solid “slug”, monitoring the water-level recovery to equilibrium conditions, and fitting a theoretical model to the field data. Hydraulic

conductivity (K) values estimated from slug tests performed in wells ranged from 0.01 to 230 feet/day. The highest hydraulic conductivity values (86 – 230 feet/day) were calculated from slug tests performed in the three wells located near the former leachate infiltration area at Bennington Landfill (SG3-MW17-01, SG3-MW17-02, and SG3-MW17-06).

Hydraulic conductivity estimates obtained from analysis of grain-size distribution data for samples collected from investigation soil borings are also summarized in Table 3.15. Most of the K values estimated from the grain-size distribution data are low (< 0.1 feet/day), which reflects the high percentage of fine-grained materials in the unconsolidated deposits.

Hydraulic conductivity values for bedrock near Bennington Landfill were estimated from specific capacity information following the method of Bradbury and Rothschild (1985). Specific capacity data were collected during dynamic flow logging at bedrock monitoring wells SG3-MW17-BR1, SG3-MW17-BR2, SG3-MW17-BR3, and SG3-MW17-BR4, which is described in Section 3.2.3.2. Water levels and pumping rates were continuously recorded during the tests, which were conducted over a period of 2 to 4.5 hours. Hydraulic conductivity estimates from the specific capacity information ranged from 0.6 feet/day at SG3-MW17-BR2 to 54.2 feet/day at SG3-MW17-BR1 (Table 3.15).

3.3.6 Groundwater Elevations

Groundwater elevations were collected using manual water level meters and using automated data-logging pressure transducers.

Manual depth-to-water measurements were taken at twelve permanent monitoring wells installed as part of this investigation, three existing monitoring wells at Bennington Landfill, and seven private wells near Bennington Landfill. Table 3.16 and Figure 3.27 summarize the water elevations measured during two synoptic rounds of manual depth-to-water measurements collected on November 2, 2017 and January 9, 2018. Field sheets from the manual measurements are included in Appendix A. A potentiometric surface map for the surficial aquifer in the vicinity of the landfill is shown on Figure 3.28. Well construction records of the residential water wells obtained from VTDEC and a construction summary for the existing landfill bedrock wells are included in Appendix E.2.

The observed bedrock groundwater levels are generally consistent with a potentiometric surface map of the bedrock aquifer for the area (Kim and Dowey, 2017). The observed bedrock groundwater levels were used with the potentiometric surface map from Kim and Dowey (2017) to develop the groundwater contours on Figure 3.29. Groundwater elevations and inferred flow directions are discussed in Section 4.2.2 and Section 4.2.3.3.

In addition to the manual measurements, bedrock groundwater elevations were continuously monitored with data-logging pressure transducers (i.e., In-Situ Level TROLLs) installed in the three existing bedrock monitoring wells at Bennington Landfill and in seven residential water wells near the landfill. The wells in which transducers were installed are shown on Figure 3.30 and listed below. The Vermont Unique GIS Name is included in parenthesis, where available.

- B-2-3
- B-4-3
- B-6-3
- 32 Cortland Lane (BI128)
- 49 Michaels Drive (BI16851)
- 54 Cortland Lane (BI2093)
- 901 Rocky Lane (BI48930)
- 1102 Rocky Lane (BI30221)
- 1371 Harwood Hill Road (BI7)
- 1400 Becks Drive (BI15386)

The residential water wells were identified from candidate locations in consultation with VTDEC and contingent on gaining access from well owners. In addition to the wells listed above, access was also provided to the residential well at 1152 Harwood Hill Road (Vermont Unique GIS Name BI35563); however, this well did not have sufficient water column height to allow for transducer installation.

The transducers were installed in the residential water wells in a 1-inch-diameter PVC stilling pipe to a depth of approximately 20-30 feet below the water level at the time of stilling pipe installation. Due to pumping in several of the residential wells during the transducer installation, a true static water level could not be measured for those wells (well pumps were not pulled to complete transducer installation).

Transducer installation at the landfill monitoring wells (B-2-3, B-4-3, and B-6-3) was more straightforward, as there were no pumps and associated riser pipe and cables obstructing access to the water column.

Transducer installation occurred between October 20, 2017 and October 31, 2017. The transducers in B-4-3 and B-6-3 were removed on December 4, 2017, and all remaining transducers were removed on January 9, 2018.

PFAS was detected in rinsate blanks of the in-situ transducer cables that allow use of vented transducers and live display of transducer readings. Unvented (i.e., absolute pressure) transducers were used instead and were suspended in the wells with fishing line. To allow removal of the barometric pressure component of the absolute pressure readings (barometric compensation), a data-logging barometer (i.e., In-Situ BaroTROLL) was deployed at the landfill to measure barometric pressure.

The transducers and barometer were programmed to record pressure measurements at consistent one-hour intervals. After removing the barometric component of the absolute pressure measurements, the resulting water pressure measurements were converted to water elevations. This was done by correcting the pressure measurement to water column height and then converting the depths to water to elevations based on the surveyed elevation of the stilling pipe or casing.

Figure 3.31 shows a hydrograph of the groundwater-elevation data collected by transducers installed in the wells shown on Figure 3.30. Hydrographs of individual wells are included in Appendix G.1.

3.4 Desktop Review of Other Potential PFAS Sources

A preliminary evaluation of other potential PFAS sources in the investigation area was performed using available public records and groundwater analytical data collected prior to this investigation. The evaluation methods and findings are provided in Appendix H.1. This evaluation was not intended to identify all other potential PFAS sources in the investigation area and additional investigation may indicate other potential sources. An additional evaluation of potential PFAS sources to the Bennington Landfill is provided in Appendix H.2.

3.5 Implementation Logistics

Logistics associated with implementing the Work Plan are described below and include: sampling and analysis QA/QC; management of investigation derived waste (IDW); coordination of property access; and surveying investigation locations.

3.5.1 Sampling and Analysis QA/QC

3.5.1.1 Analytical Method Modification

US EPA Method 537 is listed as the reference method in the title of ELLE Standard Operating Procedure; however, US EPA Method 537 is written specifically for the analysis of drinking water samples. As a result, the laboratory has made modifications to the extraction procedure to accommodate the preparation of non-aqueous samples, consisting of ultrasonic extraction for an hour with a 4:1 acetonitrile/water mixture instead of solid-phase extraction. Other modifications include the following:

- For aqueous samples, 100 mL was subsampled for extraction instead of using the entire container contents (250 mL) and Sep-Pak C18 solid-phase extractions cartridges were used instead of polystyrenedivinybenzene.
- Additional labeled analogs of target PFAS compounds were used as pre-extraction internal standards.
- As labeled internal standards were added prior to extractions, additional labeled surrogate compounds were not used.
- A holding time of 28 days from collection to extraction was established for soil samples.
- The lab did not control the relative percent difference (RPD) between high and low areas for each internal standard in the initial calibrations.
- The laboratory did not calculate peak asymmetry factors.
- The laboratory did not rotate the lowest and highest standards from the initial calibration in the continuing calibration standards.
- The laboratory did not rotate the concentration used for the laboratory control sample (LCS).
- The calibration standards were only linear PFAS isomers; however, available branched and linear standards were analyzed for qualitative purposes.
- Alternate ion transitions were used for perfluorobutanesulfonic acid (PFBS).

Additional details regarding the modifications made to the method are detailed in the project QAPP (C.T. Male, 2016a), as well as in the body of each data validation report included in Appendix C.2.

3.5.1.2 Field Quality Control

As a check on the potential for cross contamination, quality-control samples were collected and analyzed for PFAS from source materials and equipment used in the investigation. Equipment and source materials with PFAS detections that posed a risk of cross-contamination or of otherwise affecting sample results were not used in the investigation. Source materials sampled included water used by the drilling contractor for drilling and equipment decontamination and water used as final decontamination rinse water. Equipment sampled included: casing, rods, core barrels, macro core samplers, totes, tanks, and submersible pumps used by the drilling contractor; water-level pressure transducers and associated stilling tubes, suspending cables, rope, and wire; downhole equipment used by the geophysical logging contractor; and monitoring well construction materials. When possible, equipment/rinsate blanks were collected and analytical results reviewed prior to mobilization of the equipment to the site. To the extent practical, source equipment was segregated and not used for any other purpose by the drilling and geophysical logging contractors from the time that field quality control samples were collected to the time that the equipment was mobilized to the site for the investigation.

The water used by the drilling contractor was obtained from the potable water supply for the City of Troy, New York (City Water) and then treated to reduce levels of PFAS to below the associated MDLs or to concentrations not expected to cross-contaminate environmental samples. The City Water was delivered by tanker truck to the former Water Street facility for storage and treatment. Samples of City Water were collected from the truck's pump; the end of the truck's discharge hose; and from the storage tank into which the City Water was pumped. The City Water was then pumped through a point-of-entry treatment (POET) system and into another storage tank for the treated water. Samples for PFAS analysis were collected from the discharge of the POET system before the treated water went into the storage tank and, once the treated water tank was full, samples were collected from the treated water tank for laboratory analysis. Treated water sample results are presented in Appendix C.

Field quality-control samples included equipment blanks, field blanks, and duplicates. Quality-control samples were taken for each media type at a ratio of one (1) set of quality control samples per 20 media samples. Laboratory-prepared trip blanks were submitted with aqueous samples requiring analysis for Target Compound List VOCs or PFAS. The types of field quality-control samples collected and the sampling method and rationale were detailed in the FSP of the Work Plan (Barr, 2017a). Quality-control samples for the TOP Assay were limited to duplicate collection. Equipment blank and duplicate results are presented in Appendix C.

3.5.1.3 Data Validation

All laboratory analytical data were evaluated by Environmental Standards (ES) using standard operating procedures based on guidance documents and/or method documentation. In the case of PFAS analyses, ES developed a document based on the raw data they received from ELLE.

Data were evaluated at different levels of review as part of the validation process. A Level IIA data review confirms that analytical methods, analytes, and reporting levels are consistent with project objectives and applicable state and federal regulations. Additionally, this level of review verifies that supporting QA/QC

are of a level of quality necessary to support sample results. Level IV data validation consists of a full review and evaluation of all summaries and raw data associated with the data package, and ensures the highest level of defensibility.

In general, the level of validation/review effort for this investigation was as follows:

- soils undergo 90% Level IIA review, 10% Level IV validation;
- new investigative media (groundwater, surface water, sediment) undergo 100% Level IV validation; and
- as determined by the project team (i.e., additional samples may be selected for Level IV validation).

Assessments of QA/QC results, including a description of any potential impacts on the usability of project sample results, are provided in the individual data validation reports prepared by ES for each Sample Delivery Group. Validation reports and additional information on Level IIA and Level IV evaluation are included in Appendix C.

3.5.2 IDW Management and Disposal

IDW consisted of soil cuttings from boring activities and water from equipment decontamination and monitoring-well development and sampling. Handling and disposal of IDW was consistent with the approach described in the Work Plan and approved by VTDEC in the conditional approval and comment letter dated August 12, 2017 (VTDEC, 2017a). Handling and disposal of IDW differed by the media type and by the location of IDW generation (either at Bennington Landfill or elsewhere). Handling of IDW at the landfill differed from handling at other locations due to the presence of hazardous, non-PFAS contaminants (primarily PCBs), in the landfill area. A table summarizing how IDW was managed is provided in Appendix A.

Soil cuttings generated from the investigation activities were either thin-spread at each respective boring location or containerized for waste characterization and disposal. Soil cuttings from borings at Bennington Landfill were containerized in 55-gallon drums and are presently stored at the landfill, pending waste characterization and disposal. Results of the waste characterization will determine the nature of the IDW material, which will in turn dictate whether disposal of the cuttings is by thin-spreading in the vicinity of the soil boring or is in a facility permitted to receive the waste material. Outside of the landfill, soil cuttings from borings were thin-spread in the vicinity of each soil boring to the extent practicable. Excess soil cuttings from deep borings were containerized in 55-gallon drums, transported to the former Water Street facility, and transferred into a covered 20-cubic-yard roll-off container. Waste characterization sampling for soil from the roll-off container will be completed prior to future off-site disposal.

Water generated during monitoring well development, sampling, and equipment decontamination at Bennington Landfill was discharged to the ground surface. Outside of the landfill, water generated from well development and equipment decontamination was containerized in a frac tank that was staged at the former Water Street facility. Containerized water was treated prior to being discharged to the sanitary sewer in a manner consistent with the plan provided to and approved by VTDEC in response to the 1272

Order. Additional containerized water remains at the former Water Street facility and will be treated and discharged in a similar manner at a later date.

3.5.3 Property Access

Investigation activities completed on public and private property included investigative drilling, monitoring well installation, transducer installation and download, and slug testing. Access to municipal rights-of-way was obtained by contacting the respective municipality (Town of Bennington, Village of North Bennington, or the Town of Shaftsbury) in which the particular investigation activity was planned. Access to the William H. Morse State Airport was obtained by contacting the Vermont Agency of Transportation and access to Vermont State Parks property was obtained through a Special Use Permit from the VTANR.

Access to private properties was obtained by individual Access and Maintenance Agreements signed by each property owner. Prior to investigation activities on private property, at least 24-hour advance notice was provided by telephone or email, as preferred by each property owner or unless agreed to in advance by the property owner. Environmental activities generally were undertaken between 9:00 am and 6:00 pm, Monday through Friday, unless prior permission from the property owner had been obtained.

3.5.4 Surveying

Investigative borings and monitoring well locations were surveyed in the field using Real-Time Kinematic Global Positioning System (GPS) methods. The survey information for each investigation location is summarized in Table 3.1. In addition, the elevation of the top of riser and/or casing and the ground elevation for each well were measured to the nearest 0.01 foot using typical survey methods. Elevations were surveyed relative to the North American Vertical Datum of 1988 (NAVD88). These elevation data are listed in Table 3.3.

3.6 Deviations from Work Plan

Investigation and sampling activities were conducted in general accordance with the Work Plan. Minor deviations and modifications from the Work Plan are summarized in Appendix A. Deviations included minor adjustments to proposed boring locations and changes to sample collection methodology:

- Permanent monitoring well SG3-MW17-WWTP was completed in fractured bedrock, rather than the unconsolidated clay and clayey till overlying bedrock.
- Transducers were deployed in the existing Bennington Landfill bedrock wells B-4-3 and B-6-3 and collected data through December 4, 2017 when they were removed for geophysical logging. Upon re-suspending the transducers in these wells, the transducers were inadvertently installed above the well water level.
- Bedrock wells installed during this investigation could not be completed with rotasonic drilling methods. Diamond core and air rotary drilling methods were also utilized to complete the bedrock wells. Wells SG3-MW17-BR2 and SG3-MW17-BR3 were completed with air rotary with downhole hammer. Due to the drilling methods, complete rock core could not be produced and fracturing could not be observed at these locations.

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- A PVC riser was installed in bedrock wells SG3-MW17-BR1 and SG3-MW17-BR4 due to borehole collapse limiting geophysical logging and eliminating discrete interval sampling at these locations. A PVC riser was installed in bedrock well SG3-MW17-BR3 after geophysical logging and collection of the first discrete interval sample due to concerns regarding potential borehole collapse. The diameter of the PVC riser was larger than the 2-inch-diameter specified in the Work Plan in order to accommodate the downhole logging tools.

The following investigation activities were completed beyond the initial scope of work defined in the Work Plan, several of which were performed at the request of VTDEC (VTDEC, 2017a):

- Existing Bennington Landfill bedrock monitoring well B-4-3 was developed and sampled.
- Existing Bennington Landfill bedrock monitoring wells B-2-3, B-4-3, and B-6-3 were surveyed.
- Soil samples from deep borings were analyzed for TAL metals to provide a better understanding of the adsorption and attenuation of PFAS compounds in the unsaturated zone.
- Grain-size distribution analysis was completed to further assist in evaluating values of hydraulic conductivity and to compare against the assumed values that were used in the conceptual modeling described in the Draft CSM Report (Barr, 2017c). Soil samples were selected from investigation locations across the site and analyzed for grain-size distribution.
- Hydraulic conductivity estimates were obtained for all new monitoring wells for comparison against the values used in the conceptual modeling described in the Draft CSM Report (Barr, 2017c). Slug testing of the permanent monitoring wells in the unconsolidated materials and shallow fractured quartzite (SG3-MW17-WWTP) was completed. Hydraulic conductivity values for the deep bedrock wells installed at and near Bennington Landfill were estimated from specific capacities measured during dynamic flow logging.
- A preliminary desktop evaluation of other potential PFAS sources in the investigation area and Bennington Landfill was performed.
- Geophysical logging was completed at existing bedrock monitoring wells at Bennington Landfill (B-2-3, B-4-3, and B-6-3).

None of the additions or deviations were deemed to have had a negative impact on the investigation results.

4.0 Analysis

This section synthesizes the investigation results presented in Section 3.0 to address the primary objectives of the study. References to results of previous studies are made only to the extent necessary for analysis of the investigation results. The discussion does not reflect or address all available information related to the CSM.

4.1 Site Geology

The publicly available background information on site geology, which is described in the Draft CSM Report (Barr, 2017c), indicates that the geology of the investigation area generally consists of sandy glacial till deposits of varying thickness overlying bedrock that has been folded and faulted in a compressional structural setting. Bedrock across the site is almost entirely composed of various metamorphosed sedimentary rock formations. A series of thrust faults have been previously mapped in a northeast to southwest orientation across the investigation area, with older Cambrian rocks being thrust from the east over younger Ordovician rocks (Kim, 2017). The thrust rocks have formed topographic highs. General bedrock orientation and associated structure was inferred from Kim (2017) and based on mapping described in Appendix E.1.

The uppermost bedrock units mapped in the study area includes Cambrian and Ordovician bedrock formations. Cambrian formations in the study area include (from oldest to youngest) the Cheshire Formation, the Dunham Formation, the Monkton Formation, the Winooski Formation, and the Clarendon Springs Formation. The Cheshire Formation is generally composed of quartzite and sandstone. The Dunham Formation is generally composed of dolomitic quartzite, dolostone, sulfidic and phyllitic quartzite, and phyllite. The Monkton Formation is generally composed of sandstone, dolomitic sandstone, phyllitic quartzite, and phyllite. The Winooski Formation is generally composed of limestone and dolostone, and the Clarendon Springs Formation generally consists of dolostone and calcitic dolostone (Kim, 2017). Portions of the Cheshire and Dunham Formations have undergone hydrothermal alteration. Additional discussion of altered and competent bedrock units observed during the installation of deep borings and bedrock wells completed during the investigation is presented in Section 4.1.2.2 and Section 4.1.2.3.

The Ordovician formations in the study area include (from oldest to youngest) the Shelburne Formation, the Bascom Formation, and the Walloomsac Formation. The Shelburne Formation is generally composed of limestone and dolomitic limestone. The Bascom Formation consists generally of limestone with dolomitic and/or shaley layers. The Walloomsac Formation consists of phyllite and may contain thin limestone beds, breccias and chert. At some locations, the Walloomsac Formation is further divided into members which include an unnamed gray to blue chert member and the Whipstock Breccia Member that is described as being argillaceous and containing horizons of phyllite, phyllitic siltstone, and chert (Kim, 2017).

The unconsolidated materials and lithologies encountered in borings completed as part of this investigation is consistent with previous interpretations of site geology. Five cross-sections showing the

subsurface materials encountered during the investigation, their orientation, and major structural features have been constructed along transects shown on Figure 4.1. The cross sections are shown on Figure 4.2, Figure 4.3, Figure 4.4, Figure 4.5 and Figure 4.6. A contour map showing the interpreted depth to the bedrock surface is shown on Figure 4.7. The map shows that the greatest thickness of unconsolidated materials and altered bedrock occur in the vicinity of the Bennington Landfill. Photos of the geologic units encountered during drilling are provided in a photo log in Appendix A and referenced throughout this section.

4.1.1 Unconsolidated Materials

The unconsolidated materials encountered as part of this investigation largely consist of sandy glacial till (photo #10, Appendix A) with some outwash and alluvial deposits. Thickness of till units ranged significantly from several feet to over one hundred feet, with till thinner on hilltops and thicker in valleys. Till was typically classified as a sandy lean clay with gravel and some cobbles, and was typically yellowish brown to deep reddish brown. Though till was typically massive, layers and laminations that are indicative of glaciolacustrine deposition were observed in some borings. Potential lodgment till was observed in the eastern portion of the site near Bennington Landfill, where increased density and hardness made advancement difficult during drilling.

Cobbles and boulders were present within the till, with one quartzite boulder encountered at 65 feet bgs in SG3-MW17-07 measuring over five feet thick. Wherever possible, deep borings were advanced five to ten feet into rock to confirm the presence of bedrock, rather than a large boulder. Gravel and cobbles found in till were typically locally sourced (e.g., Cheshire Formation quartzite, Dunham Formation dolomite and marble, as well as other metamorphosed sedimentary rocks) and their angularity ranged from angular to rounded.

Some sandy outwash units were observed in thin intervals (typically 0.25 to 5 feet thick), randomly distributed within till. Outwash units were typically classified as silty sand or silty sand with gravel. Outwash units were saturated when encountered below the water table, whereas till units were classified as "moist", via *ASTM 2488-09a Visual-Manual Soil Classification* standards.

4.1.2 Bedrock

The bedrock lithology encountered throughout the study area during the investigation is generally consistent with publicly-available bedrock information for southern and middle portions of Vermont. Both hydrothermally altered bedrock and competent bedrock were encountered during the investigation primarily in the borings and wells installed around Bennington Landfill. This altered bedrock is likely the same material described as saprolite during previous investigations conducted at Bennington Landfill (e.g., McLaren/Hart, 1997). Further discussions of the altered and competent bedrock are provided in Sections 4.1.2.2 and Section 4.1.2.3 below. The bedrock boring and monitoring well locations are shown on Figure 3.1.

4.1.2.1 Structural Geology

As previously discussed in Section 4.1, several northeast-to-southwest trending thrust faults have been mapped in the study area (Kim, 2017). These faults are shown on the geologic cross sections on Figure 4.2 and Figure 4.4. Several joint/fracture sets and bedding/foliation planes have been identified in the study area in and around the Bennington Landfill from publicly-available information and field observations. The field measurements of bedding planes are consistent with a broad, open anticline mapped by VGS (Kim, 2017) with its axis located west of the landfill. Additional discussion of the structural geology in the study area is presented in Appendix E.1.

4.1.2.2 Altered Bedrock

Hydrothermally altered bedrock was encountered in the eastern half of the site at bedrock monitoring wells SG3-MW17-BR1 and SG3-MW17-BR4 at or near Bennington Landfill and at deep soil boring D07 along Chapel Road. Hydrothermally altered bedrock was also encountered at one location (deep soil boring D24) in the southwestern area of the site near William H. Morse State Airport (Figure 3.1). The hydrothermally altered bedrock typically consisted of parent lithologies of the Cheshire Formation or the Dunham Formation (Ogden, 1969). The hydrothermally altered rock was usually observed as a residual soil of clay or silt-sized material with minor sandy intervals, formed when the primary cementing silicates were leached from the parent rock and replaced with hydrous aluminosilicate clays (kaolin) (Ogden, 1969), along with minor iron and manganese oxide minerals (ocher and umber, respectively) (Burt, 1930). Previous investigations in the area have referred to these deposits as saprolite (McLaren/Hart, 1997).

Ocher, as encountered and interpreted in this study area, consists of sandy lean clay or gravelly lean clay with some quartz precipitate nodules (photo #13, Appendix A) and is typically associated with hydrothermal alteration of the Cheshire Formation, Dunham Formation, or Monkton Formation (Ogden, 1969). The material exhibits a distinct dark reddish-brown iron oxidation color. A five- to ten-foot-thick layer of ocher (photos #11 and 13, Appendix A) was typically present overlying kaolin in investigation borings completed near Bennington Landfill. Ocher was also encountered at deep soil boring D24 near William H. Morse State Airport (Figure 4.1 and Figure 4.2). This boring was terminated after penetrating 17 feet of ocher, and it is unknown if kaolin was present below it. The Clarendon Springs Formation has been mapped (Kim, 2017) as the uppermost bedrock just to the north of D24 and is inferred to be the parent material of the ocher observed in this boring. The presence of ocher above kaolin deposits is apparently common in southern Vermont (Burt, 1930). From review of publicly available information on the VTDEC Well Completion Report Searchable Database, hydrothermally altered bedrock appears to extend to the southwest of the Bennington Landfill at least 1,000 feet as illustrated in well number 23240 (well report 381) in the VTDEC Well Completion Report Searchable Database.

Kaolin as encountered and interpreted in this study (photos #12 and 15, Appendix A) was observed as a clayey or silty, friable, residual soil resulting from hydrothermal alteration of the Cheshire or Dunham Formations. Kaolin was observed at SG3-MW17-BR1, SG3-MW17-BR2, and SG3-MW17-BR4 and deep soil boring D07 (Figure 4.4, Figure 4.5, and Figure 4.6). Coloration of kaolin typically ranged from a white to yellowish powdery material to earthy reddish-brown. Intervals of argillaceous phyllite and quartz sericite phyllite-schist were observed within intervals of kaolin, along with milky white quartz vein fragments.

The parent rock of hydrothermally altered material such as kaolin and ocher was typically inferred by the presence of interbedded subfacies and from previous geologic investigations in the area that mapped the bedrock geology (e.g., Kim, 2017). Altered rock derived from the Dunham and Monkton Formations typically contained interbeds of carbonate rock such as limestone, dolomite, and marble. Altered rock derived from the Cheshire Formation was typically more homogenous and massive, which is indicative of Cheshire Formation quartzite parent rock.

The borehole geophysics data from monitoring wells SG3-MW17-BR1 and SG3-MW17-BR4 were consistent with the field observations of altered bedrock observed during the sampling of the well bores. The changes in lithology as noted in the field are generally consistent with changes in gamma, spontaneous potential, and resistivity response and are generally consistent between the noted well bores.

4.1.2.3 Competent Bedrock

For purposes of this report, competent bedrock refers to metamorphosed sedimentary rock that has not been hydrothermally altered. Competent bedrock was encountered across the site at depths ranging from surficial exposure of outcrops to nearly 400 feet bgs where significant hydrothermally altered bedrock was present. Competent bedrock generally consisted of very hard, metamorphosed sedimentary rocks.

Cambrian bedrock encountered during drilling consisted of a very hard, white quartzite (photo #18, Appendix A) of the Cheshire Formation or as various interbedded metamorphic rocks of the overlying Dunham Formation, consisting primarily of dolomitic quartzite. Cambrian formations mapped by others in the area include the Clarendon Springs, Winooski, and Monkton Formations (listed youngest to oldest). All of these formations lie stratigraphically above the Dunham Formation. Dolomite from the Clarendon Springs Formation was encountered in boring D15 during installation of monitoring well SG3-MW17-07. The Monkton Formation was encountered in boring D21 during the installation of monitoring well SG3-MW17-05 and was logged as gray dolomitic quartzite. The Winooski Formation, which has been mapped as limestones and dolostones with varying degrees of metamorphism (Kim, 2017), was not observed during this investigation.

The borehole geophysics data from monitoring wells SG3-MW17-BR1, SG3-MW17-BR2, SG3-MW17-BR3, and SG3-MW17-BR4 were consistent with the field observations of the competent Cambrian bedrock observed during the sampling of the well bores. The changes in lithology as noted in the field are generally consistent with changes in gamma, spontaneous potential, and resistivity response and are generally consistent between the noted well bores.

Ordovician bedrock was encountered as a suite of foliated metamorphic rocks such as slate and phyllite that was typically highly weathered at the contact with unconsolidated material. The Ordovician bedrock units at the site are members of the Walloomsac, Bascom, and Shelburne Formations (listed youngest to oldest), all of which are composed of chemical sedimentary rocks or fine-grained detrital sedimentary rocks that have undergone varying degrees of metamorphism (Kim, 2017). Black slate of the Walloomsac Formation was observed in deep borings D01 and D17. Marble and slate of the Bascom Formation was

observed in boring D14 (photo #16, Appendix A) and boring D22 (photo #19, Appendix A), respectively. Ordovician bedrock was only observed west of the Silk Road Fault (Figure 4.4).

4.2 Hydrogeologic Conditions

This section describes the hydrogeologic conditions encountered in the investigation including hydraulic conductivity, groundwater levels, discrete flow pathways, hydrothermally altered bedrock, and groundwater flow directions relating to the hydrogeologic setting of the Bennington Landfill.

4.2.1 Hydraulic Conductivity Values

4.2.1.1 Comparison to Published Values

The estimated hydraulic conductivity (K) values presented in Section 3.3.5 and Appendix G vary over six orders of magnitude and are consistent with published values associated with the types of geologic materials tested. Hydraulic conductivity can vary over several orders of magnitude even within a given material type (Freeze and Cherry, 1979), though hydraulic conductivity of unconsolidated sediments generally increases with increasing grain size. For the following general groups of materials, hydraulic conductivity values estimated from slug tests and grain-size distributions ranged from: clay, 0.00053 – 0.51 feet/day; silt, 0.0013 – 7.6 feet/day; sand, 0.0021 – 820 feet/day; and gravel, 0.1 – 230 feet/day. The overlapping ranges are to be expected given that the screened intervals/grain-size distribution samples tested are not homogenous, and the presence of an appreciable fine-grained fraction can significantly influence the hydraulic conductivity of a predominantly coarse-grained material (e.g., K = 0.1 feet/day for clayey gravel).

4.2.1.2 Comparison of Field and Model Values

Hydraulic conductivity values estimated from field data are consistent with values used in the groundwater flow model for the conceptual modeling described in the Draft CSM Report (Barr, 2017c). The conceptual modeling used horizontal hydraulic conductivity values for unconsolidated deposits that range from 0.49 feet/day for fine-grained sediments to 197 feet/day for sand and gravel. The values estimated from slug tests and grain-size distributions range from 0.00053 feet/day for clay to 820 feet/day for sand, and are consistent with the values for unconsolidated materials in the model, which all fall within the range of measured values. Hydraulic conductivity estimates obtained for quartzite bedrock range from 0.6 feet/day to 54 feet/day (Table 3.15). The model value of 1.0 feet/day for quartzite falls within this measured range.

4.2.2 Groundwater Level Monitoring

4.2.2.1 Unconsolidated Aquifer

Water levels measured at five wells completed in the unconsolidated aquifer at Bennington Landfill are shown on Figure 3.27 and Figure 3.28. Wells SG3-MW17-01, SG3-MW17-02, and SG3-MW17-06 are located near existing shallow monitoring wells B-2-1 and B-2-2. Groundwater elevations measured at B-2-1 in 30 sampling events from December 1992 through July 1997 ranged from 907.87 to 912.87 feet above mean sea level (MSL) and groundwater elevations measured at B-2-2 in 22 sampling events from April 1993 through July 1997 ranged from 907.88 to 912.89 feet above MSL (McLaren/Hart, 1997). Though the

specific datum used (Sea Level Datum of 1929 (NGVD29) or NAVD88) for these historical measurements is unknown, the November 2, 2017 groundwater elevations at SG3-MW17-01, SG3-MW17-02, and SG3-MW17-06, which range from 908.07 to 908.14 feet above NAVD88, appear to be consistent with the historical measurements at B-2-1 and B-2-2. No comparisons to existing data are possible for SG3-MW17-03 and SG3-MW17-04, the former because no wells were installed in this area as part of previous investigations at Bennington Landfill and the latter because unconsolidated monitoring wells near SG3-MW17-04 completed over a similar depth interval do not exist.

4.2.2.2 Bedrock Aquifer

Water-level variations due to pumping are evident in the data from domestic wells at 32 Cortland Lane, 54 Cortland Lane, and 1400 Becks Drive shown on Figure 3.31. Apparent water-level variations due to pumping are also observed on the individual hydrographs for 49 Michaels Drive, 1102 Rocky Lane, and 1371 Harwood Hill Road (Appendix G.1). Though pumping rates at these wells are not known, the smaller drawdowns observed for the latter group of wells may suggest that the bedrock aquifer is more permeable at these locations.

Hydrographs for all of the transducer-equipped wells indicate a decreasing water-level trend over the measurement period (Figure 3.31 and Appendix G.1). Water levels measured on January 9, 2018 were generally between 2 and 3 feet lower than water levels measured in late October 2017. It is common for groundwater levels to decrease during the winter months when there is reduced aquifer recharge from infiltrating precipitation due to a combination of frozen precipitation and frozen ground near the land surface.

Inspection of the individual well hydrographs (Appendix G.1) for wells with little or no pumping signal indicates that changes in barometric pressure correlate strongly with the smaller-scale observed water-level fluctuations not associated with the overall decreasing trend discussed in the previous paragraph. This barometric response, in which increases in atmospheric pressure cause water elevations to decrease and vice-versa, indicates that the bedrock aquifer has a relatively high barometric efficiency. In other words, changes in atmospheric pressure are not readily transmitted to the water in the aquifer adjacent to the open intervals of these wells resulting in potentiometric level changes in the well. Changes in water levels in a well as a result of atmospheric pressure changes are an indication that the aquifer is confined (Freeze and Cherry, 1979). Such conditions may also arise from other factors such as the location of a well screen beneath frozen soil or a perched zone (Butler et al., 2011). Where these barometric pressure responses are observed near Bennington Landfill, the response is interpreted as being the result of the perching of the unconsolidated aquifer above the bedrock aquifer.

Precipitation data from the William H. Morse State Airport in Bennington are also shown on the hydrographs. While increases in water level are observed at similar times to many precipitation events, it is likely that these water level changes are entirely due to the concurrent decreases in barometric pressure as discussed above.

The January 9, 2018 synoptic round of water level measurements at the bedrock wells near Bennington Landfill is shown with inferred groundwater flow directions on Figure 3.29. The spatial variability in the

water levels is typical of a heterogeneous fractured rock flow environment, and this variability complicates a detailed assessment of groundwater flow directions. Additional discussion of bedrock groundwater flow directions near Bennington Landfill is provided in Section 4.2.3.3.

4.2.3 Hydrogeologic Setting of Bennington Landfill

The hydrogeologic data collected during this investigation reveal a complex hydrogeologic setting at Bennington Landfill. A discussion of this hydrogeologic setting is provided below, and includes the presence of discrete flow pathways, the hydrogeologic significance of bedrock alteration, and groundwater levels and flow directions.

4.2.3.1 Discrete Flow Pathways

Geologic and hydrogeologic information gathered as part of the investigation were combined with existing information to evaluate the existence of discrete groundwater flow paths in the bedrock. The locations of wells referenced in this section are shown on Figure 3.1.

Results from the bedrock study (Appendix E.1) and bedrock drilling and borehole characterization reveal near-vertical joint and near-horizontal bedding-plane/foliation discontinuities which are interconnected, have average apertures ranging from 0.3 to 1.2 inches, and have spacings that are small relative to the extent of the model domain (142 square miles).

Bedding/foliation and fracture/joint strike and dip data obtained from the borehole geophysics completed in SG3-MW17-BR2 and SG3-MW17-BR3 were compared to data collected as part of the bedrock study (summarized in Section 3.2.1 and included in Appendix E.1). The data provided by the borehole geophysics identified shallow dipping bedding/foliation and a steeply dipping fracture/joint set that primarily correlates with the shallow dipping bedding/foliation (B1) and joint set 1 (J1), respectively, identified during the bedrock study (Appendix E.4). Joint set 2 (J2), identified during the bedrock study, was less prevalent than J1, and dynamic flow logging did not indicate J2 was a primary flow pathway.

Analysis of the flow logging data for SG3-MW17-BR2 and SG3-MW17-BR3 shows that groundwater predominantly flows in zones that generally correlate to two sets of structures: (1) the shallow dipping bedding/foliation B1, which trends west-northwest to east-southeast and dips to the south-southwest and (2) a steeply dipping fracture/joint set J1 which trends roughly northeast-to-southwest and dips primarily to the northwest and southeast. At the scale of the landfill area, bedrock groundwater flow is primarily through discrete pathways such as these. The lack of a vertical hydraulic gradient across the bedrock boreholes, which is illustrated by the lack of vertical flow in the boreholes (Appendix E.3) and the generally consistent packed interval heads in SG3-MW17-BR2 (Table 3.12), indicates that the discrete flow pathways are hydraulically connected. Flow through the borehole would be predominantly lateral. This hydraulic connection supports the representation of such features using an equivalent porous medium approach at the scale of the regional model used for the conceptual modeling described in the Draft CSM Report (Barr, 2017c).

The most hydraulically active intervals identified by the geophysical and flow logging at bedrock wells SG3-MW17-BR2 and SG3-MW17-BR3 are summarized below:

- SG3-MW17-BR2: 148.0 to 151.0 feet bgs; 167.0 to 172.0 feet bgs; 204.0 to 217.0 feet bgs; 250.0 to 254.5 feet bgs; and 259.0 to 264.0 feet bgs.
- SG3-MW17-BR3: 147.4 to 158.5 feet bgs; 167.8 to 174.0 feet bgs; 188.2 to 192.7 feet bgs; 205.0 to 210.0 feet bgs; 223.5 to 226.0 feet bgs; and 246.0 to 248.0 feet bgs.

4.2.3.2 Hydrogeologic Significance of Bedrock Alteration

Altered bedrock zones, where present, appear to have enhanced hydraulic conductivity based on data gathered during well installation, development, and flow logging. Wells SG3-MW17-BR1 and SG3-MW17-BR4 encountered altered bedrock ranging in thickness from approximately 170 to 200 feet, as shown on the well logs provided in Appendix A. The hydrothermally altered rock consisted usually of clay or silt-sized material with minor sand or sandy intervals, formed when the primary silicate cement was leached from the parent rock and replaced by hydrous aluminosilicates (clays) and oxides.

Borehole instability and relatively high yields of boreholes SG3-MW17-BR1 and SG3-MW17-BR4 are evidence of enhanced hydraulic conductivity in zones of bedrock alteration. Both well boreholes experienced collapse during the drilling process, which necessitated the installation of PVC well screen and riser pipe to keep the bedrock from collapsing and filling the well bore. Altered bedrock is present in the screened intervals in both SG3-MW17-BR1 and SG3-MW17-BR4 (Appendix A). In addition, both of these wells were able to sustain a pumping rate of 20 to 30 gpm with minimal (less than 5 feet) drawdown during well development activities. For comparison, at SG3-MW17-BR2, where no significant altered bedrock was observed, the well could sustain a pumping rate of only approximately 10 to 12 gpm during development and had a corresponding drawdown of over 40 feet.

To evaluate the horizontal extent of altered zones, lithology descriptions from private well logs available on the VTDEC Well Completion Report Searchable Database were reviewed for descriptions of altered bedrock (e.g., "ochre"). This exercise identified alteration in the vicinity of the landfill and extending to the southwest of the landfill. The unique well number of evaluated Bennington wells that identified the presence of altered bedrock include 23240, 405, 16851, and 553.

4.2.3.3 Groundwater Levels and Flow Directions

Groundwater levels in the Bennington Landfill area measured during this investigation and as part of previous investigations (e.g., McLaren/Hart, 1997) reflect a complex groundwater flow system with significant heterogeneity at the local scale of the landfill. Groundwater flow through the discrete flow pathways in the fractured and altered bedrock in the landfill area results in heterogeneity in heads and complicates an interpretation of flow directions between locations. Groundwater levels measured during the investigation show that groundwater in the unconsolidated aquifer is perched above the bedrock aquifer in portions of the landfill area. The interpretation of a perched unconsolidated aquifer was previously presented by McLaren/Hart (1997), and is supported by heads in nested unconsolidated and bedrock wells installed at the landfill that differ by up to nearly 200 feet (e.g., SG3-MW17-03 and SG3-MW17-BR1; Figure 3.27). The interpreted groundwater flow direction in the shallow unconsolidated aquifer to the southeast (McLaren/Hart, 1997) is supported by data collected during this investigation (Figure 3.28).

Bedrock groundwater flow directions inferred from the investigation data and regional mapping are predominantly to the west and southwest in the landfill area (Figure 3.29). The regional potentiometric surface (Kim and Dowe, 2017) shows bedrock flow directions to the west and southwest (Figure 3.27) that generally follow ground surface topography and are consistent with simulated flow directions described in the Draft CSM Report (Barr, 2017c). Water levels measured in existing bedrock wells at the landfill as part of this investigation are also consistent with water levels measured during the period 1993 through 1996 (McLaren/Hart, 1997). Bedrock groundwater flow directions at the local scale of the landfill are heterogeneous due to the fractured- and altered-rock setting and the apparent mounding of the potentiometric surface of the bedrock aquifer beneath the landfill resulting in radial flow outward from the landfill (Figure 3.29), as discussed further in Section 4.4.3.2.

Local deviations from the regional flow field are typical of groundwater flow conditions in fractured rock settings and in areas of local mounding of the potentiometric surface. At the regional scale, the frequency and interconnectedness of fractures becomes high and a regional continuity to the potentiometric surface becomes apparent as shown by Kim and Dowe (2017). In other words, the orientation of fractures has a greater control on groundwater flow direction at the local (landfill) scale than at the regional scale. It was for this reason that bedrock monitoring wells in the landfill area were located using outcrop fracture and joint orientation data with the goal of sampling groundwater in fractures that had a higher likelihood of traversing beneath the landfill. The local mounding of the bedrock aquifer's potentiometric surface results in the potential for localized groundwater flow towards the south, southeast, north, northwest, and west. The response of groundwater levels in the bedrock aquifer to barometric pressure changes suggests that the aquifer is confined at the depths and locations monitored. The bedrock aquifer may be unconfined where the bedrock is saturated at shallower depths and the overlying unconsolidated material is thin.

4.3 Relationships and Trends in PFAS Concentrations

4.3.1 Soil

4.3.1.1 Relationship of Soil PFAS to TOC, Metals, and pH

As stated in Section 1.1, a primary objective of soil sampling was to determine what, if any, relationships exist between PFAS concentrations and soil geochemistry. Specifically, the relationship between PFAS concentrations and soil TOC, metals concentrations, and pH was evaluated as a possible explanation of the observed distribution of PFAS concentrations in soil. The data analysis addresses a central aspect of the Draft CSM (Barr, 2017c), which is that TOC is the primary control on PFAS sorption to subsurface materials. The relationships between PFAS concentrations and soil TOC, metals concentrations, and pH were assessed statistically. The soil statistical evaluation includes soil data collected in 2016 at and around the former Water Street facility and data collected during this investigation. The data set includes 147 shallow soil samples (including 20 at the former Water Street facility) from the *Shallow Soil Sampling Investigation* (C.T. Male, 2016b), 54 samples collected at the former Water Street facility during the *Water Street Initial Site Characterization Investigation* (C.T. Male, 2016a), and 568 samples collected during this investigation. Details of the statistical methods and results are included in Appendix D.1a, and a summary is provided below.

The statistical analysis of soil data focused on evaluating the differences in concentrations of PFAS, TOC, and metals among groupings that might potentially affect the attenuation of PFAS compounds in soils and on the correlations between these parameters. Correlations were assessed between concentrations of the various chemical parameters (e.g., metals, TOC, and PFOA).

The statistical analyses assessing geochemical relationships in the investigation soil data show that:

- **PFOA and TOC in soil are significantly positively correlated.** The relationship between PFOA and TOC concentrations in soil is shown on Figure 4.8. The significant correlation between PFOA and TOC suggests a better retention of PFAS by organic material. Significantly higher concentrations of both TOC and PFOA were observed at shallower depths.
- **PFOA in soil is not significantly correlated with metals expected to control PFOA sorption.** No significant correlation (positive or negative) was found between concentrations of PFOA and calcium and iron. A significant correlation existed only for silver (positive), but this result was likely affected by a high proportion of non-detect values. Correlation values and scatter plots for each metal parameter are provided in Appendix D.1a.
- **PFOA in soil and soil pH are significantly negatively correlated.** This relationship may be due to a lesser ability for acidic soils to adsorb or otherwise retain PFAS.

4.3.1.2 Spatial Trends in Soil PFAS

Spatial trends in soil PFAS were assessed through two complementary methods: statistical analysis and plotting of PFAS concentrations with distance from the former Chemfab facilities. Similar to the statistical analysis described in the previous section, spatial trends in soil PFAS concentrations were identified through statistical analyses. Details of the statistical methods and results are included in Appendix D.1a, and a summary is provided below. The statistical analysis included soil data collected for this and previous investigations, including the 2016 shallow soil sampling investigation discussed in Section 2.1.1 (C.T. Male, 2016b), and 2016 former Water Street facility investigation performed by C.T. Male (discussed further in Section 4.4.2.2). For purposes of this report, multivariate analysis of the soil data across the entire investigation area was not performed because the data indicate that PFAS concentrations in soils across the investigation area are most closely tied to background sources. In addition to the statistical analysis, the PFAS concentration at distance plot (Chart 2 of C.T. Male's *Draft Shallow Soil Sampling Report* (2016b)) was updated to include the available data and is discussed below.

Statistical Analysis

The spatial trends evaluated in the statistical analysis focused on differences in the distribution of PFOA concentrations in samples collected at different depth intervals in the soil profile, and at increasing distances from known PFAS sources, including the former Water Street facility and Bennington Landfill. For purposes of this report, groupings included depth intervals (surface, 0 to 1.5 feet bgs; shallow, 1.5 to 5.0 feet bgs; and deep samples, greater than 5 feet bgs) and geographic areas (the four background sampling locations; the former Water Street facility; Bennington Landfill; and the non-landfill, non-Water Street investigation area, referred to in this section as "other" locations).

Detailed results of the statistical analyses are included in Appendix D.1a. General findings of the statistical analyses assessing spatial trends in the soil data include the following:

- **PFAS and other parameters vary significantly with soil depth, with TOC and PFAS parameters decreasing with depth.** Organic parameters (PFAS and TOC) and several major cations (e.g., calcium and magnesium) were notably higher in concentration in the surface soils than in the deep soils, whereas iron and trace metals increased in concentration with depth. The variability of PFOA with depth is shown on Figure 4.9, with statistical groupings shown as letters (a, b, c, etc.) to indicate which classifications were or were not statistically significantly different (i.e., classifications that were not significantly different have the same letter(s) in their groupings). As shown on the figure, concentrations of PFOA are not statistically significantly different between the surface and shallow soils, although the median value in surface soil (0.81 ng/g) is greater than the value for shallow soils (0.55 ng/g). Both surface and shallow soils are statistically significantly different from the deep soils, which have a median PFOA concentration of 0.21 ng/g (Figure 4.9). The decreasing PFOA concentrations with depth are consistent with PFAS sources at the ground surface. Higher concentrations of TOC are also measured in surface soils, which is consistent with a soil weathering profile where the highest TOC is found in the uppermost soil horizons.
- **PFOA concentrations in surface soils from all areas except Water Street were not significantly higher than background.** When soil PFOA concentrations are normalized to TOC concentrations, there is no statistically significant difference between the background surface soil samples and either the landfill ($p=0.443$) or “other” ($p=0.539$) samples (Figure 4.9). PFOA concentrations normalized to TOC at Water Street are statistically significantly different from the samples in the landfill, background, and “other” locations.
- **PFOA concentrations in surface soils in CAII and background locations are not significantly correlated to distance from the former Water Street facility.** No statistically significant trend was found between distance from the former Water Street facility and PFOA concentrations normalized to TOC in surface soils (up to 1.5 feet bgs) in CAII and at background locations (Figure 4.10). The trend line through the data in this region, which is neither increasing nor decreasing with distance from the Water Street facility, closely approximates the regional background value discussed in Section 4.4.1. The variability of soil data in CAII and at background locations is reflected by the relatively low R^2 values for the best-fit model.
- **PFOA concentrations are significantly correlated to distance from the former Water Street facility and Bennington Landfill when including data for all investigation areas, but the distance trend predicts the data poorly.** When including data for all investigation areas, a statistically significant correlation was found between PFOA concentrations normalized to TOC (PFOA/TOC) in surface soils (up to 1.5 feet bgs) and distance from two locations of interest in the investigation area: the former Water Street facility and Bennington Landfill. The trends are decreasing PFOA/TOC (logarithmic) with distance (logarithmic) from the former Water Street facility (Figure 4.10), and increasing PFOA/TOC (logarithmic) with distance (logarithmic) from Bennington Landfill (Figure 4.11). While the correlation is significant, the agreement between the

measured PFOA/TOC and the best-fit model is poor, as reflected by relatively low R^2 values. This means that the statistically significant distance trend is only a weak predictor of soil PFOA concentrations, even when the PFOA concentration is normalized to account for the effect of another statistically significant predictor of PFOA, TOC.

The relationship between PFOA and TOC suggests that evaluating these parameters together can assist in the interpretation of PFAS distributions across the investigation area. However, the results of these analyses show that even when PFOA concentrations are normalized to account for variability in TOC, some surface activities, local conditions, or localized sources of PFAS may lead to highly localized variations in soil chemistry that have the potential to confound interpretation of larger-scale patterns. Additional potential localized PFAS sources are discussed further in Appendix H.1.

PFAS Concentration-Distance Plots

Figure 4.12 shows average soil concentrations of total PFAS, total sulfonated PFAS, and PFOA at increasing distance from the former Chemfab facilities. The figure was prepared to update Chart 2 of C.T. Male's *Draft Shallow Soil Sampling Report* (2016b) with available data from this investigation and the 2016 investigation performed at the former Water Street facility. Samples with no detected concentration were assigned values of half of the associated MDL in the total and average calculations, which due to the many non-detects for sulfonated PFAS, means the average total sulfonated PFAS concentrations on Figure 4.12 are likely overestimated.

The influence of the former Water Street facility on soil concentrations is illustrated by Figure 4.12. The average of total PFAS and average of PFOA follow a similar trend of relatively elevated concentrations at the former facility decreasing within the 2,000-5,000 foot interval to levels consistent with the background sampling locations (located 25,000+ feet from the facility). This decrease in soil PFOA concentrations west of the CAI-CAII boundary (located approximately 6,000 feet from the former Water Street facility) is supported by the statistical analysis described above, which shows a weak decreasing trend for all data from all areas and no trend for data outside of CAI. Between 5,000-10,000 feet and 10,000-15,000 feet distance from the facility, the concentrations increase due to the elevated PFAS concentrations associated with Bennington Landfill. The relatively high total PFAS at the former Water Street facility is due to the presence of longer-chain (and less mobile) PFAS compounds in the surface soils.

The average of total sulfonated PFAS concentrations at distance from the former Water Street facility do not identify soil impacts from PFOS that would suggest residual air emissions. The ratio of total-to-sulfonated PFAS is lowest in proximity to the former Water Street facility due to the sulfonated PFAS concentrations being consistently low across all plotted distances from the facility.

In contrast to the decreasing trend in PFOA soil concentrations with distance from the former Water Street facility in CAI suggested by Figure 4.12 and the statistical analysis described above, no distance-based trend was observed from the former Northside Drive facility. The lack of impact from the former Northside Drive facility on soil concentrations is illustrated on Figure 4.12, which shows no consistently decreasing concentration of average PFOA with distance from the former Northside Drive facility to suggest residual soil impacts from air emissions. The average of total PFAS and total sulfonated PFAS also

suggest no residual soil impacts, as the concentration of total PFAS does not show a decreasing trend and the concentration of total sulfonated PFAS is generally constant across all distances from the facility.

4.3.1.3 Presence of Sulfonated PFAS Compounds

Given that PFOA is understood to be the only PFAS compound in the air emissions from the former Chemfab facilities, the presence of sulfonated PFAS compounds in soil samples is indicative of a different PFAS source. This is further supported by the distribution of sulfonated compounds that does not change with distance from the former facilities (Section 4.3.1.2), the infrequent detection of sulfonated compounds near the former Water Street facility (Section 4.4.2.2), and the presence of sulfonated compounds in four of the background soil samples, at concentrations ranging from 0.42 to 1.1 ng/g (Table 3.5), and at a number of the non-background soil sampling locations (Table 3.6).

4.3.2 Groundwater

The analyses of groundwater data described below include both an analysis of groundwater data collected as part of this investigation and the 2016 investigation of the former Water Street Chemfab facility (45 data points) and an analysis of other available area groundwater data (over 700 data points). The purpose of the analyses was to evaluate the transport of PFAS at the ground surface and deeper soils through the unsaturated zone to the water table and to evaluate the role of Bennington Landfill and the former Water Street Chemfab facility as PFAS sources.

Section 4.3.2.1 provides the statistical analysis of groundwater data completed from a limited investigation data set (the leachate vault, temporary wells and permanent investigation monitoring wells), and Section 4.3.2.2 provides the analysis of the broader dataset (including private well sampling data). Findings from the more robust, comprehensive data set are considered to be more representative of site conditions.

4.3.2.1 Analysis of Investigation Groundwater Data

A statistical evaluation of the groundwater quality data collected across the investigation area was conducted to identify similarities and differences in the distributions of PFAS concentration in Water Street, landfill and other areas. Classification of water type from major ion chemistry was also performed to identify similarities and differences in groundwater geochemistry suggesting a different or shared source. Details of the statistical evaluation of investigation data are in Appendix D.1b.

The statistical analyses of the 45 investigation groundwater data do not identify a statistically significant difference in the distribution of concentrations of PFOA or PFOS in groundwater sampled at Bennington Landfill and at locations outside of the landfill or in PFOS in groundwater sampled at the former Water Street Chemfab facility and at locations away from Water Street. However, the analysis did identify a statistically significant difference in the concentrations of PFOA between groundwater sampled at the former Water Street Chemfab facility and at locations away from Water Street. The water type, as identified from the relative abundance of major ions in the groundwater sample, does not indicate substantially different water sources for the unconsolidated aquifer, suggesting that the water samples were collected from a chemically similar source of water, such as the same aquifer.

4.3.2.2 Multivariate Analysis of Groundwater PFAS Data

For purposes of this report, PFAS groundwater data collected to date were statistically evaluated with the private well sampling data collected by the VTDEC and POET influent monitoring data collected on behalf of Saint-Gobain. Statistical tools were used to determine if there were patterns outside of spatial considerations that could be used to identify groups or “families” of samples. By using multivariate analysis (MVA) statistical approaches on the available dataset, previously unrecognized relationships between samples may identify potentially separate sources of PFAS, or may potentially identify the likelihood of a common source. The MVA analysis presented herein is an update of the analysis presented in the Draft CSM Report (Barr, 2017c) and reflects the inclusion of data collected as part of this investigation.

MVA is a well-established set of statistical methods for evaluating data involving more than one variable. MVA procedures were applied to the data from the investigation area to discern data patterns and relationships between data points, based solely on the composition of the PFAS compounds. For purposes of this report, the statistical add-on tool to Microsoft Excel, XLStat (www.xlstat.com), was used in the MVA procedures.

Agglomerative Hierarchical Cluster (AHC) analysis was used in the MVA evaluation. AHC involves evaluating the data (i.e., multiple PFAS compounds from many locations) without any consideration to mechanisms, physical processes, trends, or spatial location. AHC analysis seeks to identify which data locations are similar to other locations and groups or “clusters” like-data based on statistical thresholds. AHC is a statistical iterative classification method which produces a dendrogram (clustering tree), representing a hierarchy of objective similarities and dissimilarities. The agglomerative methodology used was Ward’s method using Euclidian distance proximity. Truncation of the AHC was conducted manually and favored fewer groups with larger populations over many groups with smaller (or single) populations.

PFAS data included in the MVA analysis included those PFAS compounds from EPA Method 537 that were found throughout the results and included: PFBS, PFHpA, perfluorohexanesulfonic acid (PFHxS), perfluorononanoic acid (PFNA), PFOS, and PFOA. For purposes of this report, where no value is reported in the private well sampling dataset provided by VTDEC, a value of “no detection” was assumed for all parameters not reported for a sample. The presence of non-detect concentrations required assignment of values to allow analysis. Detection limits may be different for different compounds and for different samples. For purposes of this analysis, a non-detect result was treated as a value of 0 ppt and J-qualified values (i.e., values between the instrument detection limit and the practical reporting limit) were not altered.

PFAS data were selected from the available groundwater and POET influent data and transformed in the following manner prior to analysis:

- The geographic location of the data within the investigation area was not a factor in data selection or statistical analysis.
- Multiple observations made at a single location were treated as separate data points due to the amount of time between sample collections (approximately quarterly).

- Of the 1169 total groundwater samples, 711 data points met the criteria of at least one PFAS compound detected and all six PFAS compounds analyzed for.
- The data set was normalized to convert to fractional concentrations of the sum of PFAS measured and the resultant data was processed with XLStat for AHC. Data processed in this manner allowed for clustering with an emphasis on the total sample profile, diminishing the role played by absolute PFOA concentration and influences on absolute concentrations such as dilution and mixing.

Overall, three groups were identified from the statistical clusters. The groups are largely divided by contributions of PFOA versus sulfonated compounds, and are as follows:

- Group A – This group contained 600 samples. Members of this group are characterized by the dominance of detections of PFOA and PFHpA, and no sample with total sulfonated compounds greater than 3% of the total PFAS detected. All other PFAS compounds were less than 5% of the total PFAS compounds detected.
- Group B – This group contained 99 samples. Members of this group largely have a sulfonated presence (88 of 99 samples). All PFAS compounds have a presence in this group, but not across all samples. The PFOA contribution ranges from 59 to 92% of the total PFAS detected.
- Group C – This group contained 12 samples. The group is characterized by a relatively small contribution from PFOA, ranging from 0 to 45%. The samples can be considered heavily sulfonated, with contributions from sulfonated compounds ranging from 44 to 79% in the samples where detected.

The three groups identified by the AHC analysis are shown on Figure 4.13. The groups are described as (A) groundwater with PFOA or PFOA/PFHpA and low or no sulfonated PFAS compounds; (B) groundwater with widespread sulfonated compounds, and (C) groundwater with high proportions of sulfonated compounds.

The overall patterns of groupings and characteristics of the groupings for this analysis are very similar to the analysis results presented in the Draft CSM Report (Barr, 2017c). The additional groundwater data collected as part of this investigation reinforce the AHC findings. Wells in the general vicinity of the former Chemfab facilities have PFAS detections that are predominantly PFOA or PFOA/PFHpA and typically do not have sulfonated PFAS compounds (Group A). This profile is consistent with a source of PFAS similar to PFOA-based air dispersion and is consistent with the results of the CSM modeling of air emissions and deposition from the former Chemfab facilities. Wells in CAAII, northeast of the former Chemfab facilities and in the vicinity of Bennington Landfill, are statistically grouped together because they generally contain detectable concentrations of sulfonated compounds such as PFOS, PFHxS, and PFBS and are not correlated with PFOA/PFHpA (Group B). This profile suggests a mixture of PFAS compounds from a variety of commercial, domestic, and industrial sources other than the former Chemfab facilities. The sporadic presence of wells with high proportions of sulfonated PFAS compounds throughout the investigation area (Group C) indicates a greater statistical commonality with the wells in the Bennington Landfill area than wells in the vicinity of the former Chemfab facilities, again indicating that there are a variety of localized, mixed PFAS sources within the investigation area.

4.3.2.3 Evaluation of Groundwater PFAS Ratios

Support for distinct sources of PFAS to groundwater in the investigation area is also provided by an evaluation of the ratios in concentrations of PFHpA and PFOA in groundwater. Such an evaluation is possible due to the fact that these compounds are two of the most commonly detected PFAS parameters in data from this investigation and from private well and POET monitoring performed by VTDEC and on behalf of Saint-Gobain. Figure 4.14 shows the spatial distribution of the ratio of PFHpA/PFOA across the investigation area, calculated from the same groundwater PFAS dataset as the multivariate analysis described in Section 4.3.2.2, including data from the CSM site investigation, the 2016 Water Street site investigation, and POET and private well sampling data. It is clear from Figure 4.14 that the PFHpA/PFOA ratio, which ranges from approximately 0.02 to 1.73, is not spatially uniform. The lowest PFHpA/PFOA ratio values (0.02-0.05) are clustered around the former Water Street facility and other areas of CAAI near the Walloomsac River and, to a lesser extent, downgradient of Bennington Landfill. The relatively low PFHpA/PFOA ratio values in these areas are likely reflective of relatively high PFOA concentrations resulting from air emissions from the former Water Street facility (in the Water Street area) and from groundwater impacts from Bennington Landfill (in the Bennington Landfill area). In the outer regions of CAAI and across CAAII (including along Harwood Hill Road, downgradient of Bennington Landfill) the PFHpA/PFOA ratio values are similarly clustered but at higher ratio values (0.05-0.23). The difference in PFHpA/PFOA ratio value groupings between the areas near Bennington Landfill and the former Water Street facility is consistent with different sources of PFHpA and PFOA to groundwater in those areas.

4.4 Evaluation of PFAS Sources

As stated in Section 1.1, a primary objective of the investigation was to identify some of the other potential sources of PFAS within the investigation area. The specific PFAS sources evaluated as part of this investigation include the Chemfab facilities, background sources, sources related to Bennington Landfill, and PFAS precursors. Other potential PFAS sources in the investigation area were identified as part of a preliminary desktop evaluation. These potential sources are discussed in the following sections.

4.4.1 Background PFOA Soil Concentration

The concentration of PFOA in soil has two components: the local source contribution and the background contribution. To better understand the magnitude of the local source contribution, the background concentrations were subtracted from measured soil concentrations for purposes of this report. A source of background soil concentrations of PFOA is the atmospheric transport and deposition of PFOA and its precursors (Rankin et al., 2016). Background concentrations are also influenced by local water infiltration rates and by the retention of PFOA within the soil column. A soil's TOC content strongly influences PFOA retention (Zareitalabad et al., 2013). A positive correlation between TOC and PFOA concentrations exists for soils in the investigation area (Appendix D.1a). This suggests that for the same background PFOA deposition rate, the background soil PFOA concentration will vary with soil TOC content. Background soil PFOA concentrations can be calculated by multiplying the background PFOA/TOC ratio by the soil's TOC content. A background PFOA/TOC ratio was developed for the investigation area soils.

As described in Appendix D.2, background (off-site) soil samples were collected at four locations north and east of Bennington, at distances ranging from approximately 8 to 22 miles from the former Water Street facility and in locations not expected to be impacted by emissions from the former Chemfab facilities (Figure 3.1). Two approaches were used to develop a background soil PFOA concentration. The first approach was based upon the average PFOA/TOC ratio for soil samples collected from the upper 18 inches of the soil column. This approach yielded an average PFOA/TOC value of about 0.0586 ng PFOA/mg TOC. The second approach was based upon the measured PFOA and TOC values for soil samples collected from only the upper 6 inches of the soil column. From the slope of line generated by a least-squares-fit of the PFOA and TOC data, the background deposition of PFOA corresponds to a PFOA to TOC ratio of about 0.109 ng PFOA/mg TOC. Both values of PFOA/TOC are within the range of values observed in the Rankin et al. (2016) database for thirty-three North American undisturbed and surficial soil samples. For purposes of this report, evaluations regarding the effect of a background deposition on the distribution of PFOA in soil, and simulated soil concentration from the CSM, the lower of the two estimated values (0.0586 ng PFOA/mg TOC) was used.

Soil concentrations of PFOA above the range of background concentrations were calculated for each soil sample, based on the measured TOC concentration in the sample. The TOC-normalized background PFOA concentration for each sample was calculated from the measured TOC concentration and TOC-normalized background PFOA concentration of 0.0586 ng PFOA/mg TOC. This sample-specific background PFOA concentration was subtracted from the measured PFOA concentration to provide the above-background PFOA concentrations shown on Figure 4.15 and Figure 4.16. The results on these figures show no clear spatial trend or clustering of concentrations exceeding background values that would suggest potential other current sources in addition to background. Evaluation of other current sources was beyond the scope of this investigation and the potential for other current sources has not been fully evaluated.

Simulations of the fate and transport of PFOA in the vicinity of the former Chemfab facilities, described in the Draft CSM Report (Barr, 2017c), used a TOC concentration of 25,000 mg/kg soil (2.5 percent by weight) to characterize the retention of PFOA in the surficial soil. This investigation shows a positive correlation between TOC and PFOA concentrations in soil samples collected from the Bennington area (Figure 4.8; Appendix D.1a). Thus, for purposes of this investigation (comparing simulated soil concentrations to those measured in the field), a background PFOA concentration associated with the average background PFOA/TOC ratio of 0.0586 ng PFOA/mg TOC and a TOC of 25,000 mg/kg was calculated to be 1.46 ng PFOA/g soil (i.e., 1.46 ppb).

4.4.2 Former Chemfab Facilities

The former Chemfab Northside Drive facility operated from 1969 until mid-1978 when operations were transferred to the Water Street facility. The former Chemfab Water Street facility operated from mid-1978 until February 2002, when the plant closed. Modeling results presented in the Draft CSM Report (Barr, 2017c) suggest that air emissions of PFOA from the former Chemfab facilities contributed PFOA, in varying amounts, to groundwater in certain, but not all portions of the investigation area. Investigation data, where available, were evaluated to assess the modeling results and the Chemfab facilities as a source of impacts to the investigation area.

4.4.2.1 Northside Drive

Site-specific data was not collected for the former Chemfab facility at Northside Drive, because access was not available at the time of Work Plan development and approval. General site data indicate that PFOA deposited by historical air emissions at the Northside Drive facility is no longer present at detectable levels in soil in this area, likely due in part to the fact that this facility ceased emissions approximately forty years ago in 1978.

The lack of impact from the former Northside Drive facility on soil concentrations is illustrated by Figure 4.12, which shows the PFAS concentrations (as average of total PFAS, average of total sulfonated PFAS, and average of PFOA) at increasing distance from the facility. The average of total PFAS and average of PFOA follow a similar trend that shows no consistently decreasing concentration with distance from the former Northside Drive facility to suggest residual soil impacts. The average of total sulfonated PFAS are relatively low and consistent across all distances from the facility suggesting potential impacts from regional background sources.

4.4.2.2 Water Street

Site-specific data for the former Chemfab facility at Water Street was collected in 2016 as described in Section 2.1.1 and the Water Street Work Plan (C.T. Male, 2016a). Validated soil and groundwater quality data collected as part of this investigation and figures interpreting the data are included in Appendix I, Tables I.1 and I.2 and Figures I.1 through I.9.

Soil Data

During the 2016 investigation, a total of 54 soil samples were collected and analyzed for PFAS and TOC. PFOA was detected above the MDL in 46 samples at concentrations ranging from 0.49 to 30 ng/g, whereas PFOS was detected above the MDL in four samples at concentrations ranging from 0.76 to 1.1 ng/g. The maximum TOC concentration was 97,300 mg/kg. Water Street PFOA, PFOS and TOC soil data are provided in Appendix I, Figures I.1, I.2 and I.3, respectively, and are shown with site data from the CSM SI and the 2016 shallow soil investigation discussed in Section 2.1.1 (C.T. Male, 2016b).

Elevated PFOA concentrations are present in the immediate vicinity of the former Water Street facility; however, Figure I.1 does not show concentrations uniformly radiating from the facility, suggesting the additional presence of localized sources and reflecting the period of time since emissions ceased (Appendix I, Figure I.1). The frequency of PFOA detections in soil sampled at Water Street (86%), and the consistent PFOA detections across sampled depths, support that PFOA was present in the air dispersion from the facility (Appendix I, Figure I.6 and Figure I.7, respectively).

Low to non-detect concentrations of PFOS in the vicinity of the former Water Street facility indicate the facility is not a source of PFOS, consistent with the understanding that PFOS was not present in the dispersions used at Water Street (Appendix I, Figure I.2). The frequency of PFOS detections in soil sampled at Water Street (< 10%) and the limited detections across sampled depths, further support that PFOS was not present in the air dispersion from the facility (Appendix I, Figure I.6 and Figure I.7, respectively). The influence of the former Water Street facility on soil concentrations is illustrated by Figure 4.12, which

shows the PFAS concentrations (as average of total PFAS, average of total sulfonated PFAS, and average of PFOA) at increasing distance from the facility. The average of total PFAS and average of PFOA follow a similar trend of relatively elevated concentrations at the former facility decreasing to levels consistent with the background sampling locations (which are 25,000+ feet from the facility) at approximately the 2,000-5,000 foot interval from the facility. The concentrations increase between 5,000-10,000 feet and 10,000-15,000 feet distance from the facility, which is consistent with approaching the elevated concentrations associated with Bennington Landfill. The relatively high total PFAS at the former Water Street facility is due to the apparent presence of longer-chain (and less mobile) PFAS compounds in the surface soils. The average of total sulfonated PFAS shows no soil impacts from PFOS that would suggest residual air emissions, as the concentration is relatively low across all plotted distances from the facility.

Groundwater Data

The absence of sulfonated PFAS compounds in air emissions from the former Chemfab facilities is evidenced by the fact that these compounds are not uniformly detected in groundwater in the vicinity of the former facilities and do not consistently increase in groundwater with increasing proximity to the former facility (Appendix I, Figure I.4 and Figure I.5).

Two rounds of groundwater samples were collected in 2016 from 11 monitoring wells installed near the former Water Street facility. PFOA was detected at all sampled locations during both sampling events at concentrations ranging from 34 to 4,800 ng/L. PFOS was detected above the MDL at six locations during the December 2016 monitoring round, with all but two of the six detections at qualified concentrations at (2 ng/L) or near (3 ng/L) the detection limit and the remaining two detections at concentrations of 14 and 17 ng/L. PFOA concentrations in groundwater are uniformly present and elevated in the vicinity of former Water Street facility (Appendix I, Figure I.4), whereas PFOS concentrations in groundwater are low to non-detect and not uniformly present (Appendix I, Figure I.5).

The detection frequency plot for groundwater by area indicates that PFOA is detected at the highest concentration (4,800 ng/L) and is detected most consistently at non-landfill, landfill, and Water Street locations (Appendix I, Figure I.8). The frequency of PFAS detection in groundwater by depth identified PFOA as most consistently identified in shallow unconsolidated, deep unconsolidated and bedrock wells, likely due in part to PFOA being present at higher concentrations and being more mobile than longer chain PFAS compounds.

4.4.3 Bennington Landfill Impacts

The role of Bennington Landfill as a potential source of PFAS to soil and groundwater in the unconsolidated and bedrock aquifers is illustrated by the analytical results from this investigation, from private well sampling data collected by VTDEC, from POET influent sampling data collected on behalf of Saint-Gobain and by bedrock groundwater flow directions in the landfill area.

The investigation data were evaluated to assess the presence of current and residual impacts in soil and groundwater. Widespread residual impacts were not expected in surficial soil and shallow groundwater, given the time between the investigation and completion of corrective actions at the landfill. Capping of

the unlined landfill was completed in 1999, which effectively cut off the source of infiltration within the footprint of the cap. However, ongoing sources of PFAS impacts to groundwater in the vicinity and downgradient of the landfill may include: infiltration of leachate from the landfill that was not removed by the leachate collection system; impacts sorbed to soils beneath the landfill, the soil stockpile area and/or former leachate infiltration gallery; downward leakage from the shallow aquifer layer; and water migrating from the unsaturated zone beneath the bedrock aquifer and the shallow aquifer into the bedrock aquifer.

A drain structure, referred to as the Upgradient Groundwater Isolation Trench, was installed to intercept shallow groundwater and divert that water around the landfill (TRC, 1998). Since the leachate collection system was decommissioned in 2008, no leachate infiltration has occurred in the leachate infiltration gallery and little or no infiltration has occurred through the footprint of the landfill in recent years. Accordingly, any impacts from the landfill were expected to be most likely present in deeper media like bedrock groundwater.

4.4.3.1 Soil

Residual impacts to soil from historical wastewater treatment plant (WWTP) sludge storage at the landfill were not apparent in the data from two areas designated as storage areas in historical landfill records. The soil analytical results shown on Figure 3.9 and Figure 3.10 illustrate that current concentrations of PFOA and PFOS in soil in the former sludge storage areas are not consistently or uniformly elevated. However, with the exception of shallow soil boring S08, concentrations of PFOS in soil exceed 1.5 ng/g only at the landfill (Figure 3.7), suggesting the presence of localized sources. While the investigation data show no residual impacts in soil at sludge storage areas, the current concentrations do not preclude historical impacts by the landfill. Because PFOA has migrated downward through the soil column with infiltration since landfill leachate infiltration and sludge storage ceased, residual impacts in the soil from the leachate were not expected in the investigation data.

4.4.3.2 Groundwater

The previous interpretation of a groundwater flow direction in the shallow unconsolidated aquifer to the southeast (McLaren/Hart, 1997) is supported by site data, as shown on Figure 3.28. Concentrations of PFOA in the shallow unconsolidated aquifer decrease with distance downgradient of the leachate vault (VLT-01) and in the vicinity of the former leachate infiltration area, which is consistent with leachate being a source of PFOA and other PFAS parameters in the shallow groundwater (Figure 3.2.3). Evidence of leakage through the base of the overburden saturated zone, developed on a leaky basal layer (as opposed to an impermeable layer), includes historical detections of VOCs in bedrock wells at the landfill (McLaren/Hart, 1997).

The interpreted groundwater flow direction in the bedrock aquifer in the region surrounding Bennington Landfill is to the west and southwest (Kim and Dowe, 2017), with the available site data suggesting radial flow outward from the landfill (Figure 3.29). Monitoring wells SG3-MW17-BR2 and SG3-MW17-BR3, located adjacent to the landfill cap and southwest of the landfill, respectively, are considered downgradient, as the dominant flow direction in the southwestern portion of the landfill is to the west and

southwest. Wells SG3-MW17-BR1 and SG3-MW17-BR4 are sidegradient of the landfill, due to radial groundwater flow outward from the landfill (Figure 3.29).

The evaluation of Bennington Landfill impacts on groundwater is illustrated by the inferred bedrock aquifer flow directions and the PFOA and PFOS analytical results from this investigation and from private well sampling shown on Figure 4.17 and Figure 4.18, respectively. To further assess the role of Bennington Landfill as a potential source of PFAS to groundwater in the bedrock aquifer, an additional evaluation of the analytical results was completed by interpolating measured PFAS concentrations in the investigation and private well data sets (Appendix H.3). For the interpolated concentrations presented in Appendix H.3, data from the most recent sampling event were used at each location, and where samples were non-detect for a given parameter, one-half the MDL was used. Due to the limited number of PFAS parameters analyzed for private wells and the high proportion of non-detects for some of the PFAS parameters, groundwater concentrations for the following parameters were evaluated: PFOA, PFOS, PFHpA, PFHxA, and PFHxS. The distribution of these parameters in groundwater is shown on the figures in Appendix H.3, and the results of this evaluation are summarized below.

Concentrations of PFOA and other PFAS parameters identify impacts from the landfill, with the highest concentrations in the downgradient direction. Table 4.1 summarizes the concentrations of PFAS parameters in the Bennington Landfill leachate vault sample (VLT-01) and landfill bedrock wells SG3-MW17-BR1, SG3-MW17-BR2, SG3-MW17-BR3, and SG3-MW17-BR4. PFOA is detected at the highest concentrations in the wells downgradient of the landfill (SG3-MW17-BR2 and SG3-MW17-BR3) and at lower concentrations in the sidegradient wells (SG3-MW17-BR1 and SG3-MW17-BR4). Concentrations of several other PFAS parameters identified in landfill leachate were detected at increased concentrations in downgradient monitoring wells, but are absent or are detected at low concentrations in sidegradient monitoring wells. These detections of PFAS at low concentrations in sidegradient wells may result from radial flow outward from the landfill. This could also explain low-level detections of PFOA in bedrock well SG3-MW17-BR1 (north of the landfill) and bedrock well B-4-3 (located northwest of the landfill), as well as the low-level detections of PFOA in private wells north and west of Bennington Landfill shown on Figure H.3.1.

The general magnitude of specific PFAS parameters in leachate is reflected in the magnitude of specific PFAS parameters in downgradient wells, indicating the landfill is a source of impacts to groundwater. As shown on Table 4.1 and Figure H.3.1, the concentrations of PFOA are significantly higher in the leachate vault and in downgradient wells. This is generally consistent with PFOA concentrations increasing while groundwater flows beneath the landfill area, suggesting that the landfill is the primary source of PFOA in bedrock groundwater in the areas southwest of the landfill (Figure 4.17 and Figure H.3.1). Leachate concentrations for PFHpA (Figure H.3.3) and PFHxA (Figure H.3.4) are detected at the second- and third-highest concentrations in the leachate vault (after PFOA) and are detected in groundwater wells at a greater magnitude than concentrations of PFOS (Figure H.3.2) and PFHxS (Figure H.3.5).

Variability in PFAS groundwater concentrations downgradient of the landfill is present. Corrective action at the landfill (including capping, upgradient groundwater diversion, and leachate collection) over

the past two decades may have reduced PFAS migration and infiltration of water from the landfill to the bedrock flow system. Subsequent pore flushing of the bedrock fractures since the implementation of corrective action at the landfill has resulted in the potential for a decrease in PFAS concentrations in bedrock groundwater adjacent to the landfill, relative to concentrations further downgradient.

PFOS concentrations do not identify a concentration gradient that would indicate a potential source of air emissions from the former Chemfab facilities. Concentrations of PFOS in groundwater downgradient of the landfill (Figure H.3.2) indicate the presence of multiple sources, including the landfill, and are not representative of air emissions (Figure 4.18).

Table 4.1 Bennington Landfill Groundwater Impacts to Bedrock

Area Location Depth Parameter	Leachate	Sidegradient		Downgradient					
	VLT-01	SG3-MW17- BR1	SG3-MW17- BR4	SG3-MW17- BR2				SG3-MW17- BR3	
				165- 175 ft	199- 219 ft	243- 265 ft	open hole 147- 263 ft	131.27- 161 ft	screened 112-272 ft
Per- and Polyfluoroalkyl Substances [ng/l]									
Perfluorooctanoic acid (PFOA)	4700	14	0.4 J	170	140	160	160	72	64
Perfluorohexanoic acid (PFHxA)	590 J			41	41	42	38		0.4 J
Perfluoroheptanoic acid (PFHpA)	280	0.8 J		38	37	45	39	2	2
Perfluorobutanoic acid (PFBA)	130 J			16	18	18	17		
Perfluoropentanoic acid (PFPeA)	92			28	30	31	30		
Perfluorohexane sulfonate (PFHxS)	48			3	2	3	2		
Perfluorooctanesulfonic acid (PFOS)	35 J			1 JN	0.9 J	2 J	2 J		
Perfluorobutane sulfonate (PFBS)	9 J			4	5	8	7		
6:2 Fluorotelomer sulfonate (6:2 FTS)	4 J								
Perfluorononanoic acid (PFNA)	3			3 J	2	5 J	4		
n-Ethyl perfluorooctanesulfonamidoacetic acid (N-EtFOSAA)	2 J								
Perfluorodecanoic acid (PFDA)	0.8 J								
8:2 Fluorotelomer sulfonate (8:2 FTS)									
Perfluorooctanesulfonamide (PFOSA / FOSA)									
n-Methyl perfluorooctanesulfonamidoacetic acid (MeFOSAA)									
Perfluoroundecanoic acid (PFUnA / PFUnDA)									
Perfluorododecanoic acid (PFDoA / PFDoDA)									
Perfluorotetradecanoic acid (PFTA / PFTeDA / PFTeA)									
Perfluorotridecanoic acid (PFTTrDA / PFTTriA)									

Blank cells indicate non-detect results.

4.4.4 PFAS Precursors

Results of the TOP Assay (Appendix F) for groundwater samples collected at Bennington Landfill suggest that there is a negligible mass of PFAS precursors in the shallow groundwater system. Based on the results of the TOP Assay, it appears that the amounts of chemically-oxidizable precursors present in the samples from the Bennington Landfill are negligible, with the exception of the sample from the leachate vault (VLT-01) (Table 3.14). The lack of significant PFAS precursor mass suggests that the background soil concentrations described in Section 4.4.1 result from the ongoing addition of PFAS from other background sources. Further, although the TOP Assay analysis has some limitations, the analyses indicate

that concentrations of PFAS seen in the future would not be due to the presence of precursors, provided the vault integrity at the landfill is maintained.

4.4.5 Other Potential PFAS Sources

The desktop evaluation of other potential PFAS sources in the investigation area described in Appendix H.1 preliminarily identifies local sources other than the former Chemfab facilities. Appendix H.1 describes other potential PFAS sources in CAAI and CAII that likely existed (and in some cases potentially still exist) in the investigation area. Appendix H.2 summarizes an evaluation of potential PFAS contributors to the Bennington Landfill based on the EPA's "Explanation of PRP Waste Contribution Estimates, Bennington Landfill Superfund Site". The results of this preliminary evaluation suggest that additional investigation of these potential sources by the appropriate regulatory agency may be warranted.

4.5 Evaluation of Expected Distribution of PFAS Concentrations

As stated in Section 1.0, a primary objective of the investigation was to evaluate the distribution of PFAS in soil and groundwater modeled in the CSM against the concentrations identified during the investigation. As described in the Draft CSM Report (Barr, 2017c), numerical models were developed to simulate potential transport mechanisms of PFOA released by historical air emissions at the former Chemfab facilities that may have resulted in the presence of PFOA in soil and groundwater in the vicinity of these facilities. The results of the simulations were compared to measured PFOA concentrations in soil and groundwater.

This section presents comparisons of PFOA concentrations measured in soil and groundwater during this investigation with simulated values from the models presented in the Draft CSM Report (Barr, 2017c). The comparison does not include the other PFAS parameters measured as part of this investigation, as PFOA was the only PFAS parameter simulated in the conceptual modeling. The fraction organic carbon (f_{oc}) used in the conceptual modeling is also compared to values measured during this investigation. The results of this analysis are used to update the conceptual model as presented in Barr (2017c).

4.5.1 Saturated Zone Groundwater

A scatter plot of simulated PFOA concentration in groundwater versus concentrations measured as part of this and previous investigations is presented on Figure 4.19. For purposes of this report, all non-detect values are plotted at their associated detection limit. Overall, there is a slight bias of simulated concentrations being lower than measured concentrations, particularly for higher measured concentrations.

4.5.2 Unsaturated Zone Soils

As shown on Figure 4.20A, simulated soil concentrations of PFOA are consistently lower than measured values across the investigation area. The model has no lower limit for simulated values and can simulate concentrations much lower than can be quantified by current laboratory methods. Figure 4.20A compares the measured concentrations of PFOA in soil to the model's predictions of soil concentrations for the date the sample was collected at locations and depths of the measured values.

For purposes of the conceptual modeling, sorption of PFOA on organic carbon was simulated using a linear sorption model with a distribution coefficient (K_d) based on literature values for the organic carbon partitioning coefficient (K_{oc}) and typical values of the f_{oc} in soils (Barr, 2017c). Values of f_{oc} assumed in the Draft CSM Report (Barr, 2017c), which decrease with depth below ground surface in soils, are compared with measured values from the current investigation on Figure 4.21A. Values shown on Figure 4.21 and summarized in Table 4.2 are also from C.T. Male (2016a, 2016b), and data for soil borings completed at Bennington College (Schroeder, T.; email communication, September 8, 2017).

Measured and modeled f_{oc} values are also compared in Table 4.2 for the depth ranges used in the unsaturated zone model (Appendix C of the Draft CSM Report (Barr, 2017c)). The model input values fall within the observed range for each depth interval. For summarizing data in Table 4.2, one-half the method detection limit was used in calculations of the geometric mean. Figure 4.21B shows the distribution of f_{oc} within each depth interval, demonstrating that the geometric mean is a better representation of the central tendency than the arithmetic mean. However, it is noted that for both the 10-36 cm interval and > 36cm interval, the f_{oc} values are neither normally, or log-normally distributed as shown with the Gaussian kernel density plots on Figure 4.21B. As stated in the Work Plan, in addition to samples from planned sampling intervals, samples were collected from organic-rich soils due to their likely high retention of aerially-deposited PFAS compared to low-organic soils; therefore, the results may be biased toward higher f_{oc} values. These results indicate that the f_{oc} values used to represent the three different depth intervals in the modeling are representative of the available data.

Table 4.2 Measured and Modeled f_{oc} Values for the Unsaturated Zone Model

Sample midpoint depth range	Geometric mean of detected values	Minimum detection	Maximum detection	Number of detections	Number of non-detections	Model input
0-10 cm	0.016	0.00041	0.17	130	0	0.025
10-36 cm	0.0084	0.00043	0.48	133	3	0.005
>36 cm	0.0022	0.00012	0.51	474	50	0.0005

f_{oc} expressed as mass fraction, decimal equivalent of percent (unitless)

4.5.3 Comparison to the Conceptual Model

The Draft CSM Report (Barr, 2017c) was reviewed to evaluate if any updates or revisions to the conceptual modeling are necessary in light of additional data collected as part of this investigation. In particular, the numerical models' prediction of groundwater and soil PFOA concentrations below measured values was considered. Background deposition from long-range transport of PFOA and potential contributions from other localized sources like those identified in Appendix H.1 are important contributors that were not included in the Draft CSM Report. The additional background sources and potential additional local sources are believed to be the primary explanation for the model predictions of soil concentrations below measured concentrations.

The model prediction that PFOA potentially emitted from the former Chemfab facilities would no longer be present in soil at the time of this investigation is supported by the finding that the investigation soil data do not show a strong spatial trend with distance from the former Water Street facility (Section 4.3.1.2). The measured PFOA concentrations in soil were higher than the numerical models predict and are consistent with a background source (Section 4.4.1). Figure 4.22 shows a simulated time series of soil concentration of PFOA east of the former Water Street facility resulting from modeled emissions from the former Chemfab facilities that illustrates the predicted attenuation of soil concentrations over the fifteen years since the emissions ceased at the former Water Street facility (2002). Soil concentrations are modeled to peak at around 17 ng/g in 1998 and decline rapidly after that, especially after emissions cease in 2002. In certain conditions (e.g., elevated TOC in soil, low infiltration) and certain locations, elevated PFOA concentrations in soil from the former Chemfab emissions may still be present, but overall soil concentrations present today are expected to be more representative of background or other regional sources. The simulated soil concentration for 2017 is 0.3 ng/g, which is only slightly above the detection limit for soil PFOA data collected as part of this study. This illustrates that PFOA potentially emitted from the former Chemfab facilities, in the form of elevated soil concentration, would not be expected 15 years after emissions ceased.

A background source of PFOA was not included in the conceptual modeling, given that the purpose of the modeling was to perform a screening level evaluation of the extent to which emissions from the former Chemfab facilities may have potentially contributed to PFOA concentrations in groundwater in certain areas. Figure 4.20B is similar to Figure 4.20A, but adds to the model's predicted PFOA concentrations in soil the background soil concentrations of PFOA as a function of the model's TOC values. Because different TOC values were used in the model over three depth intervals (0-10 cm, 10-36 cm, and >36 cm), adding the background contribution results in a clustering of the simulated concentrations with respect to the sampling depth. Figure 4.20B illustrates two considerations: (1) representing more of the measured spatial variability in TOC concentrations in the model would likely produce an even better match between simulated and observed PFOA soil concentrations and (2) including background contributions to soil concentrations is important in assessing the goodness of fit between model-simulated soil concentrations and measured concentrations.

Because transport and deposition is significant and ubiquitous across the modeling domain, simulated PFOA concentrations in both soil and in groundwater are lower than they would be if the background PFOA loading was applied. After accounting for background, model-simulated and measured soil and groundwater concentrations correlate well.

5.0 Summary

The following statements summarize the results from the investigation activities and data analyses described herein. These summary statements address the overarching investigation objectives of assessing hydrogeologic conditions in the investigation area, evaluating relationships and trends in PFAS concentrations, evaluating some of the other potential additional local PFAS sources, as well as measuring background conditions, and assessing measured PFAS concentrations against the modeled distribution.

- Hydrogeologic conditions were found to be very similar to those described in the CSM.
- Relationships and trends in PFAS concentrations indicate that current concentrations of PFOA in soil in CAAII are not indicative of historical impacts from the former Chemfab facilities. The measured soil concentrations are generally consistent with background concentrations and may also be indicative of localized sources of PFAS.
- Statistical analysis of available groundwater data from this investigation and monitoring of private wells and POET systems suggests different PFAS sources reflected by groundwater of different PFAS composition, specifically the relative proportion of sulfonated compounds.
- Bedrock groundwater analytical data indicate that the Bennington Landfill is the primary source of PFAS in groundwater southwest of the landfill.
- Atmospheric transport of PFAS from sources outside the investigation area appears to be a significant source of PFAS within the investigation area.
- A desktop evaluation of area industries identifies additional potential local sources of PFAS within the investigation area.
- After accounting for background sources, the data collected as part of this investigation are consistent with the CSM. The findings do not warrant further investigation for the purposes of assessing the effects of air emissions from the former Chemfab facilities.

It is not anticipated that any additional investigation will be required. However, as discussed in the report, other likely local sources contributing to the presence of PFAS exist within the investigation area and may warrant additional investigation by the appropriate regulatory agency.

5.1 Hydrogeologic Conditions

- The distribution and values of hydraulic conductivity were found to be similar to conditions described in the Draft CSM Report (Barr, 2017c).
- The bedrock aquifer near Bennington Landfill is confined and water levels are consistent with a published bedrock potentiometric surface in which groundwater flow within the bedrock is from the landfill to the south, southwest, and west. At the very local scale of the landfill itself, there is localized mounding of the potentiometric surface beneath the landfill, resulting in local flow directions that are to the north and northwest, as well as south, southwest, and west. While the predominant flow direction in the bedrock is southwest, the localized mounding beneath the landfill has resulted in more radial flow that merges with the regional flow field.
- Bedrock structures identified in the bedrock outcrop and lineament study of the Bennington Landfill area were found to be hydraulically active and altered bedrock zones were found to have

enhanced hydraulic conductivity. Groundwater flow in the local landfill area is through these discrete flow paths. Borehole logging data show the discrete flow paths to be hydraulically connected, supporting the representation of the bedrock as an equivalent porous medium at the regional scale of the conceptual modeling from the Draft CSM Report (Barr, 2017c).

5.2 Relationships and Trends in PFAS Concentrations

5.2.1 Soil

- A significant, positive correlation exists between soil PFOA and TOC. Higher PFAS concentrations were found in soils with higher TOC, indicating that the organic matter in the soils attenuates PFOA, PFOS and other PFAS compounds relative to the movement of water. A significant, negative correlation was found between PFOA and soil pH and silver. No other metals were found to significantly correlate with PFOA.
- PFAS concentrations in soil decrease with depth in the soil column, consistent with sources at the ground surface including atmospheric deposition and infiltration from the ground surface. TOC concentrations in soil decrease with depth in the soil column, consistent with a soil-weathering profile. PFAS fractions that are adsorbed to the organic carbon are highest in the shallow soil layers where organic material content is highest. These findings agree with the results and parameterization of the conceptual modeling.
- Normalizing PFOA concentrations by TOC revealed weak spatial trends in the PFAS; specifically, that the PFOA/TOC ratio decreases with increasing distance from the former Chemfab Water Street facility when accounting for all data, but does not follow a spatial trend with distance from the facility outside of CAAI.
- The presence and concentration of sulfonated PFAS compounds in soil was an indicator of impacts from other sources.
- With the exception of the soils at the former Water Street facility, PFOA concentrations were not statistically higher than concentrations at background sampling locations. This finding indicates that PFOA concentrations in soil are not indicative of past impacts from the former Chemfab facilities. In general, any PFOA in soils that potentially may have been due to historical deposition from the former Chemfab facilities appears to have now attenuated.

5.2.2 Groundwater

- Statistical analyses of groundwater data from this investigation and data from private well sampling and POET influent monitoring indicated distinct groupings of samples based on relative contributions of PFOA and sulfonated compounds. These groupings suggest different PFAS sources to groundwater.

5.3 PFAS Sources

5.3.1 Background

- The PFOA present in background soil samples appears not to represent residual from the historical emissions from the former Chemfab facility. The emissions stopped 15 years ago.

Flushing with infiltrating precipitation for 15 years should have removed all measurable residuals associated with the former Chemfab facility from the surficial soils.

- Analysis of background soil data suggests that the continuing atmospheric transport and deposition of PFOA and its precursors are responsible for measurable background soil concentrations of PFOA in the Bennington area. The average PFOA/TOC value for representative background soil samples collected for this investigation is within the range of PFOA/TOC values reported for surficial soil samples located across North America. The presence of a background source of PFOA is further supported by lack of a spatial trend in soil PFOA concentration outside of CAAI.

5.3.2 Bennington Landfill

- Analysis of PFAS compounds in relation to each other indicated distinct groupings of samples based on relative contributions of PFOA and sulfonated compounds. These groupings suggest different PFAS sources to groundwater at and downgradient of Bennington Landfill. The relative concentrations of PFAS compounds in groundwater is similar to the concentrations in the sample of leachate collected from the landfill leachate vault.
- The presence of sulfonated PFAS compounds in downgradient bedrock landfill wells suggests that the groundwater is being impacted by PFAS sources that are unlike Chemfab-related materials, which are not sulfonated.
- Groundwater flow directions in the bedrock in the vicinity of Bennington Landfill, along with the magnitude of PFAS concentrations relative to the leachate, indicate that private wells downgradient of the landfill have been affected by sources of PFAS in the landfill.

5.3.3 PFAS Precursors

- Results of the TOP Assay analysis for PFAS precursors show no significant precursor mass in landfill leachate and groundwater in the unconsolidated aquifer at Bennington Landfill.

5.3.4 Other Potential PFAS Sources

- Results of a preliminary desktop review of other potential PFAS sources in the investigation area indicate that a number of other potential sources exist. Additional investigation of these potential sources by the appropriate regulatory agency may be warranted.

5.4 Expected Distribution of PFAS Concentrations

- Model-simulated and measured concentrations correlate well when background sources are accounted for in the model.
- The organic carbon content of the soil measured for this study confirms that the f_{oc} values used in the unsaturated zone transport modeling are reasonable and representative of actual conditions. Model values fall in the range of measured values and follow the same general decrease with depth in the soil column.

6.0 References

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