UPDATED INTERIM GROUNDWATER CHARACTERIZATION REPORT

MEYERS LANDFILL

El Dorado County, California

Prepared for

UNITED STATES DEPARTMENT OF AGRICULTURE
FOREST SERVICE, REGION 5
LAKE TAHOE BASIN MANAGEMENT UNIT
South Lake Tahoe, California

Prepared by

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LIST OF ABBREVIATIONS AND ACRONYMS

1,2-DCB	1,2-Dichlorobenzene
1,4-DCB	1,4-Dichlorobenzene
2D	two dimensional
3D	three dimensional
µg/Kg	micrograms per kilograms
µg/L	micrograms per liter
µg/m^3	micrograms per cubic meter
AOC	Administrative Order of Consent
ARARs	applicable or relevant and appropriate requirements
ASTM	American Society for Testing and Materials
BAI	Broadbent & Associates, Inc.
bgs	below ground surface
CCR	California Code of Regulations
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cfm	cubic feet per meter
CFR	Code of Federal Regulations
CGS	California Geological Survey
CMP	corrugated metal pipe
cis-1,2-DCE	*cis*-1,2-Dichloroethene
cm/sec	centimeters per second
COC	contaminant of concern
CPT	Cone Penetration Test
DCA	dichloroethane
DHS	Department of Health Services
DIPE	diisopropyl ether
DO	dissolved oxygen
DWR	Department of Water Resources
E&E	Ecology and Environment, Inc.
EPA	United States Environmental Protection Agency
°F	degrees Fahrenheit
FEMA	Federal Emergency Management Agency
gpm	gallons per minute
GeoSyntec	GeoSyntec Consultants, Inc.
Geomatrix	Geomatrix Consultants, Inc.
HDPE	high-density polyethylene
JPA	Joint Powers Authority
LTBMU	Lake Tahoe Basin Management Unit
MCL	State of California Maximum Contaminant Level
MEK	Methyl ethyl ketone
MH	Manhole
mg/L	milligrams per liter
ml	milliliter
MSL	mean sea level
NCP	National Contingency Plan
OU	operable unit
ORP	oxidation-reduction potential
PCE	tetrachloroethene

LIST OF ABBREVIATIONS AND ACRONYMS (CONTINUED)

PDB	passive diffusion bag
Ppbv	parts per billion by volume
ppm	parts per million
PTEM	Phase Three Environmental Management
PVC	polyvinyl chloride
RCRA	Resource Conservation and Recovery Act
RI/FS	Remedial Investigation/Feasibility Study
RWQCB	Regional Water Quality Control Board
STPUD	South Tahoe Public Utility District
SWAT	Solid Waste Assessment Test
TBA	tertiary butyl alcohol
TCE	trichloroethene
TOC	total organic carbon
trans-1,2-DCE	*trans*-1,2-Dichloroethene
TtEMI	Tetra Tech EM, Inc.
TTLC	Total Threshold Limit Concentration
µg/L	Micrograms per Liter
USACE	United States Army Corps of Engineers
USFS	United States Department of Agriculture Forest Service
USGS	United States Geological Survey
VOCs	volatile organic compounds
WESTON	Weston Solutions, Inc.

1. INTRODUCTION

Weston Solutions, Inc. (WESTON®) was tasked by the U.S. Department of Agriculture, Forest Service (USFS) with continuing remedial investigations of the groundwater operable unit (Operable Unit 2 (OU-2)) at the Meyers Landfill (the "Site") (Figure 1). The primary objective of these additional studies is to close data gaps in the current understanding of the hydrogeologic model. These investigations supplement several phases of groundwater studies summarized in the *Remedial Investigation Report, Meyers Landfill; El Dorado County, California (Phase Three Environmental Management (PTEM), June 2000),* the *Supplemental Remedial Investigation and Feasibility Study, Meyers Landfill, El Dorado County, California (Weston Solutions, Inc., May 2007)*, and the *Interim Groundwater Characterization Report, Meyers Landfill, El Dorado County, California* (July 2012), and will ultimately provide data needed for completing a Supplemental Remedial Investigation and Feasibility Study (RI/FS) for OU-2.

The Meyers Landfill site is a former waste disposal site located northeast of the town of Meyers on Forest Service lands within the USFS Lake Tahoe Basin Management Unit (LTBMU), El Dorado County, California. The Site operated from about 1947 through 1971. Starting in 1955, operations continued under a series of Forest Service Special Use Permits that were issued to private parties and El Dorado County (County). During its operational history, the Site operated as a burn dump up until the early 1960s and from thereafter as a "trench and fill" waste disposal site. The peak period of waste disposal at the landfill occurred between 1965 and 1971. Waste deposited in the landfill included mixed municipal solid waste from South Lake Tahoe, unincorporated portions of El Dorado County, and from Douglas County (Nevada). The Site stopped receiving waste in 1971 and by 1973 the County had closed the dump and placed a 2- to 4-foot soil cap over the waste.

In 1974, inspections of the Site discovered that: i) landfill leachate was flowing from the buried culvert at the north end of the dump into nearby Saxon Creek, and ii) the soil covering the dump had eroded in certain areas, leaving waste exposed. This discovery prompted the Lahontan Regional Water Quality Control Board (RWQCB) to issue a Cleanup and Abatement Order (#75-5) in 1975, to the Forest Service and County. The Clean-up and Abatement Order required that the Forest Service and the County implement corrective measures to mitigate the leachate discharge and soil erosion. From 1975 to 1977, the Forest Service and the County implemented corrective measures that were successful in mitigating the environmental concerns that had been identified at the time.

In response to the requirements of the California State Water Code Section 13273, the Forest Service began site investigation efforts in 1991 for the purpose of preparing a Solid Waste Assessment Test (SWAT) Report for the Site. Groundwater investigations conducted as part of the SWAT found that groundwater beneath the waste was contaminated with volatile organic compounds (VOCs) including vinyl chloride and *cis*-1,2-dichloroethene (cDCE). In August 1996, vinyl chloride was detected in groundwater downgradient of the Site, and the Forest Service initiated a response under CERCLA to determine the impacts of the contamination, pursuant to its lead agency authority provided by Executive Order 12580. Since the discovery of groundwater contamination, the Site has become the focus of intensive investigations performed under the direction of the USFS, County, and the City of South Lake Tahoe. A brief summary of the contractors who have performed the work and the studies they performed is presented in Section 2.3.

In 2005, a technical workgroup composed of the USFS, County, the City of South Lake Tahoe (City), South Tahoe Public Utilities District (STPUD), and the Lahontan RWQCB identified a number of data gaps that had not been answered by, or were discovered during, the previous investigations that had been performed at the Site between 1975 and 2005. In response to the data gaps that were identified, the USFS tasked WESTON to conduct additional investigations and to prepare a Supplemental RI/FS. WESTON performed these remedial investigations in 2005 and 2006.

In the summer of 2006, the USFS separated the Site into two operable units (OUs), the landfill mass OU (OU-1), which includes the landfill itself, and the groundwater OU (OU-2), which consists of the groundwater underlying the landfill and the associated plume of VOCs moving off the landfill. In 2007, WESTON prepared the *Supplemental Remedial Investigation and Feasibility Study, Meyers Landfill, El Dorado County, California (May 2007)* which focused on the landfill OU-1. Based on the recommendations contained in the May 2007 RI/FS, the USFS entered a Record of Decision for OU-1 in November 2007, and contracted Engineering /

Remediation Resources Group, Inc. (ERRG) to prepare remedial design documents for the landfill cap.

In September 2007, the USFS contracted WESTON to conduct additional investigations with the intent of preparing a Supplemental Remedial Investigation report for OU-2. The primary objectives of these additional studies were to:

- Address existing data gaps regarding groundwater conditions beneath and downgradient of the Site.
- Continue routine monitoring of groundwater and surface water at the Site.
- Develop baseline/background groundwater data to assess metals and chemical indicators of the aquifer environment that may be used for evaluation of remedial alternatives for groundwater.

Due to the extended time required to complete the OU-2 investigation, the USFS requested WESTON to prepare an Interim Groundwater Characterization Report to document the results of the investigation to date. The Interim Groundwater Characterization Report (Weston, July 2012) presented results of the additional studies and provided further recommendations for closing existing data gaps in order to complete the Supplemental Remedial Investigation for OU-2. Recommendations in the Interim Characterization report included:

- Conduct additional groundwater monitoring to identify potential concentration trends and evaluate effect of landfill cap installed in 2011-2012.
- Install piezometers to investigate Perched Groundwater Zone (PGZ) along west side of landfill to evaluate effectiveness of new french drain system installed in 2010.
- Install an additional well (MW-30) in the Upper Groundwater Zone (UGZ) upgradient of the landfill to provide additional data to supplement well MW-16, which appeared to be screened across combined PGZ and UGZ. Install three additional wells (PU-1, PU-2 and MW-31A) in the UGZ west of the landfill to delineate the western lateral edge of the UGZ plume.
- Install three additional Middle Groundwater Zone (MGZ) monitoring wells (MW-28, MW-29, and MW-31B) to delineate the western lateral edge of the MGZ plume and confirm that flow in the MGZ is not affected by pumping of the Elks Club Well No. 2 production well to the west of the Site.

- Conduct additional stream gaging and sampling of Saxon Creek to evaluate potential surface water/groundwater interaction.
- Conduct sampling for physical and chemical groundwater quality parameters pertinent to evaluation of potential natural attenuation of the contaminant plume.

The USFS tasked WESTON to conduct additional investigative efforts to address these issues and provide groundwater modeling to refine the site conceptual model, further evaluate concentration trends, and simulate fate and transport of the contaminant plume under existing conditions. This document presents an update to the Interim Groundwater Characterization Report (WESTON, 2012), documenting additional subsurface investigation, additional groundwater and surface water monitoring, and results of groundwater modeling. This report presents additional conclusions and recommendations for further work to complete the Remedial Investigation and Feasibility Study process for assessment under CERCLA.

1.1 PURPOSE OF REPORT

The purpose of this Update to the Interim Groundwater Characterization Report is to document the methods and results of investigation activities completed by WESTON from July 2011 through May 2017. Results of these investigations have been used to further refine the site hydrogeologic model and address existing data gaps. These investigations will provide data necessary for completing a Supplemental Remedial Investigation for OU-2, and lead to completion of a Feasibility Study for mitigating groundwater impacts, if needed.

The specific project objectives included:

- Continue VOC delineation, especially vinyl chloride, in groundwater.
- Further refine the site hydrogeologic model.
- Evaluate seasonal groundwater variations as a result of precipitation, and snow melt.
- Evaluate the effectiveness of the french drain and source of vinyl chloride impacts to perched groundwater.
- Evaluate potential groundwater/waste interaction beneath the landfill.

- Develop baseline/background groundwater data to assess selected metals and chemical indicators of the aquifer environment that may be used for evaluation of remedial alternatives for groundwater.

- Evaluate the surface water/groundwater interaction.

- Evaluate general trends in VOC concentrations, and model potential fate and transport of vinyl chloride contaminant plumes.

To meet these objectives, the following activities were conducted during the recent investigations (July 2011 through May 2017):

- Conduct 24 quarters of monitoring well gaging and sampling for VOCs, from spring 2011 (July 2011) through spring 2017.

- Drill and install a total of seven (7) new monitoring wells to fill data gaps in the existing monitoring well network and one soil boring in which a well was not constructed due to not encountering groundwater.

- Drill and install a total of 13 piezometer wells on either side of the french drain west of the landfill to evaluate perched groundwater flow, potential impacts, and effect of french drain and two soil borings in which piezometers were not constructed due to not encountering groundwater.

- Gage the stream flow of Saxon Creek during 12 of the 24 monitoring events and sample surface water from Saxon Creek during 12 of the 24 quarterly monitoring events to evaluate potential groundwater/surface water interaction.

- Collect water samples for analysis of VOCs and gage flow from french drain outfall during the past ten of the 24 monitoring events and sample the culverted intermittent stream outfall during the past nine of the 24 monitoring events.

- Collect samples for analysis of bioattenuation parameters from five wells during six of the 24 quarterly monitoring events to provide background/baseline data for evaluating the potential for natural attenuation or in-situ groundwater treatment.

Results of each quarterly monitoring event have been summarized in quarterly reports submitted periodically during the investigation period. These results build on previous groundwater sampling results and are summarized in this report.

1.2 REPORT ORGANIZATION

The Interim Groundwater Characterization Report contains seven sections as described below:

- Section 1 describes the purpose and scope of the project.
- Section 2 summarizes the Site background and historical investigation activities.
- Section 3 describes the physical characteristics of the Site including surface features, surface water hydrology, geology and hydrogeology.
- Section 4 provides details on the investigations recently performed by WESTON from spring 2011 through spring 2017. The investigations performed include drilling and well installation, routine groundwater monitoring, and surface water monitoring.
- Section 5 presents results of the recent investigations. The new data is used to up-date the geologic and hydrogeologic models for the Site building on results from previous studies.
- Section 6 provides conclusions and recommendation for continuing the investigation activities leading to eventual completion of a supplemental RI/FS for OU-2.
- Section 7 provides a list of references.

2. SITE BACKGROUND

2.1 SITE DESCRIPTION

The Site is situated in the LTBMU (Latitude 38° 52' 28" North, Longitude 119° 59' 16" West) in El Dorado County, California. The Site is located approximately 1.9 miles northeast of the town of Meyers and 4.5 miles south of Lake Tahoe. The landfill surface elevation ranges from approximately 6,370 to 6,407 feet above mean sea level (MSL) in Lake Valley and was built in an unnamed intermittent stream valley (Figure 1).

Until the summer of 2010, the landfill waste mass occupied approximately 11 acres of a 17-acre plateau with a relatively flat surface that was sparsely covered by grasses and small trees. The landfill and plateau are situated in 25 to 30 acres of non-forested land. The northern and southern edges of the plateau slope steeply to the original surface of the valley floor. The eastern edge terminates at an unnamed ridge between the landfill and Saxon Creek. The landfill is bounded on the west by the paved USFS Road 12N08, also known as Garbage Dump Road, which provides gated access to the Site from Pioneer Trail. An active electrical substation operated by Liberty Utilities is north of the landfill, Saxon Creek is to the east, and the intermittent stream valley is to the south. The nearest residences are located approximately 1,500 feet north on Hekpa Drive and 1,100 feet west on Busch Way. The Trout Creek trunk line sewer, owned and maintained by STPUD, runs approximately parallel to the eastern and northern sides of the landfill mass (Figure 2).

In the summer of 2010, the County began work on the landfill to build an engineered cap as outlined in the Record of Decision. This activity has significantly altered the physical layout of the landfill as a result of waste consolidation, grading, construction of settling basins, installation of storm water controls, and abandonment of several former landfill wells. The cap was not completed by winter 2010, so work was halted until spring 2011 when work re-commenced. The impermeable cap liner was installed around July 2011, and construction of the cap was completed July 2012. A french drain was installed along the west side of the landfill in fall 2010 in conjunction with cap construction to capture and collect perched groundwater occurring along that area to prevent interaction of groundwater with landfill waste.

To facilitate earthmoving activities on the landfill, the older landfill wells (M-1 through M-5) were abandoned; however, the three newer wells (LF-1 through LF-3) remained. The two piezometers (P-1 and P-2) located along the west side of the landfill were also abandoned. In addition, a buried water supply well located near the northwest side of the landfill was uncovered and abandoned. It is likely this well is the one identified in a 1961 drawing titled "X – Section Lines, Lake Valley Garbage Site" provided by the USFS. No other information regarding the construction or use of the well has been found. Landfill facility drawings from 1961 illustrate that the well was located west of the limits of landfill waste near the site of several former structures associated with the County's landfill operations.

2.2 SITE HISTORY

The Site operated as a landfill from 1947 through 1971 with the County operating the landfill under a series of USFS Special Use Permits between the years of 1955 and 1972. Waste acceptance was reportedly terminated in 1971, prior to the expiration of the last Special Use Permit, and the landfill was covered with 2 to 4 feet of cover soil in 1973.

The peak period of waste disposal at the landfill was between 1965 and 1971 when an estimated 35 tons of waste per day was accepted (USFS, 1996). As part of the May 2007 RI/FS, WESTON estimated the total in-place volume of landfill waste to be approximately 305,000 cubic yards. The waste estimate was relatively consistent with the volume of 290,171 cubic yards estimated by Phase Three Environmental Management based on a waste disposal area estimate of 7.38 acres and an average waste thickness of 24.38 feet (PTEM, 2000).

The landfill operated as a burn dump until the early 1960s, after which time the waste was likely placed and directly buried with soil (PTEM, 2000). Waste deposited in the landfill included mixed municipal solid waste from residential and commercial sources in South Lake Tahoe, unincorporated portions of El Dorado County, and Douglas County (Nevada) (PTEM, 2000).

In 1972, the USFS issued a permit to the County to operate an asphalt batch plant adjacent to the landfill. A map dated January 1977 titled, Meyers Landfill Leachate and Erosion Control Project and Demonstration Planting Area, illustrates that the asphalt batch plant was constructed between the landfill and Saxon Creek, southeast of the landfill and plateau. The batch plant permit expired in 1992. The batch plant was operated by Herms Bros who had installed a water supply well in the area as indicated by a DWR well log. It is unknown if the well was properly abandoned or buried. The USFS reported that there was no other written documentation discussing the asphalt batch plant operations in their files.

In May 1975, the RWQCB collected samples of the leachate that was flowing from the landfill and water samples from the intermittent stream and Saxon Creek. In response to the detection of the leachate, the Lahontan RWQCB issued Cleanup and Abatement Order No. 75-5 to the USFS and the County.

In 1976, the USFS installed a french drain, and re-grading and re-vegetation of the landfill were completed as interim solutions while the USFS was developing a long-term solution for leachate control. The USFS also conducted a geophysical investigation of the landfill to determine the depth of soil cover and the volume of waste. The results of the investigation were reported in a 1977 report and concluded that the waste was over 40 feet thick and covered with a thin permeable soil cover (PTEM, 2000; USFS, 1996). The 1977 USFS report made recommendations to divert the intermittent stream in a culvert beneath the landfill and to install a cut-off wall across the fill to divert water from the landfill mass (LTBMU, 1977).

In 1979, the USFS installed five groundwater monitoring wells (M-1 through M5) within and through the landfill waste and well M-6 located just north of the landfill. There is little information available concerning the construction and sampling of these wells with the exception of frequent gaging of water levels. The first record of landfill well sampling was in 1991 and VOCs were detected in the groundwater for the first time at the Site.

The July 1996 USFS Solid Waste Assessment Test (SWAT) report documented the repair of the landfill cover and summarized field activities conducted to date (USFS, 1996). The report stated that the surface discharge of leachate present in 1975 was no longer evident and the sampling of surface water and soil from beneath the landfill indicated no leakage of hazardous constituents.

In response to the discovery of vinyl chloride contamination in groundwater beneath the landfill in 1991, the USFS initiated an RI under CERCLA to evaluate the environmental impacts caused by the landfill. As part of the RI, groundwater investigations were conducted between 1994 and

1996, including the installation of four wells outside of the landfill footprint to assess groundwater contamination migrating from the landfill. Vinyl chloride was detected in groundwater downgradient from the landfill for the first time in 1996.

PTEM completed an RI in June 2000 for the South Lake Tahoe Basin Waste Management Authority (Joint Powers Authority [JPA]). The JPA was formed by the City of South Lake Tahoe, El Dorado County, and Douglas County (Nevada) to accomplish goals and encourage construction of a materials recovery facility and other solid waste handling facilities in the Tahoe Basin. Ecology & Environment, Inc. (E&E) completed the feasibility study for the USFS in 2002. The RI identified approximately 33 VOCs in soil and/or groundwater that were migrating beyond the boundaries of the landfill. The list of VOCs identified in the groundwater included the following:

- Vinyl chloride
- *cis*-1,2-Dichloroethene (*cis*-1,2-DCE)
- *trans*-1,2-Dichloroethene (*trans*-1,2-DCE)
- Tetrachloroethene (PCE)
- Trichloroethene (TCE)
- 1,1-Dichloroethane
- 1,2-Dichloroethane
- 1,2-Dichlorobenzene (1,2-DCB)
- 1,2-Dichloropropane
- 1,2,4-Trimethylbenzene
- 1,3-Dichlorobenzene
- 1,3,5-Trimethylbenzene
- 1,4-Dichlorobenzene (1,4-DCB)
- 4-Isopropyltoluene
- Acetone
- Benzene
- Chlorobenzene
- Chloroethane
- Chloromethane
- Dichlorodifluoromethane
- Dichloromethane
- Ethylbenzene
- Isopropylbenzene
- Methyl ethyl ketone (MEK)
- Methyl tert-butyl ether
- *meta*- and *para*-Xylenes
- *ortho*-Xylene

- n-Butylbenzene
- n-Propylbenzene
- Naphthalene
- sec-Butylbenzene
- tert-Butylbenzene
- Toluene

Vinyl chloride concentrations at the Site are typically in the range of approximately 10 times higher than other VOCs. Vinyl chloride is considered to be more toxic then the other VOCs with a California Maximum Contaminant Level (MCL) of 0.5 micrograms per liter (µg/L) for drinking water. In January of 2002, the USFS issued a Feasibility Study and Proposed Plan for the Site, which identified and screened alternatives for remediation of the landfill and selected a preferred remedial alternative. The proposed preferred remedial alternative consisted of a cap over the waste-mass, consistent with EPA's guidance "Presumptive Remedy for CERCLA Municipal Landfill Sites", and a "pump and treat" system for addressing the groundwater contamination. As a result of comments provided by the County and the City to the proposed plan, the USFS entered an Administrative Order of Consent (AOC) with the County and a separate AOC with the City to conduct supplemental remedial investigations in 2003, 2004, and 2005. Under the AOCs, the County investigated the VOC groundwater contamination and the City investigated issues related to the landfill. The investigations under the AOCs terminated in 2005, and the USFS has since undertaken the program that includes these investigations. The Record of Decision to implement the remedy was signed by the USFS on November 15, 2007.

In 2009, ERRG completed the Meyers Landfill Final (100%) Remedial Design – Operable Unit 1, Multilayer Cap for the USFS. The remedial design detailed requirements for installation of a multi-layer landfill cap along with associated landfill gas control, stormwater protection and installation of a new french drain system to control perched water along the west and southwest margins of the landfill.

In 2010, the County began implementing the remedial action for OU-1. The first major task was to relocate and consolidate waste from the margins of the landfill to the main landfill mass, particularly along the east side where waste had been placed above the sewer line. Waste consolidation was completed in the 2010 resulting in approximately 107,000 cubic yards of material relocated over the main landfill mass. The consolidated landfill waste is covered with a multi-layer cap consisting of foundation, impermeable geosynthetic liner, drainage layer and vegetative cover. Construction of the remedial measures (engineered cap and associated elements) was completed in 2012, with the landfill cap liner installed in summer of 2011. A new french drain was installed on the western edge of the landfill in 2010 to capture shallow perched groundwater and divert it away from the landfill. Surface water runoff diversions and infiltration basins were also constructed.

2.3 PREVIOUS INVESTIGATIONS

The USFS, City, and County have conducted studies at the Site from 1975 to the present. Table 1 provides a summary of work done at the landfill. The following section summarizes the previous investigation work pertaining to the groundwater investigations. Additional details on these and other previous investigations, including investigations conducted on the landfill OU (OU-1) can be found in the *Remedial Investigation Report, Meyers Landfill; El Dorado County, California (Phase Three Environmental Management (PTEM), June 2000)*, the *Supplemental Remedial Investigation and Feasibility Study, Meyers Landfill, El Dorado County, California (Weston Solutions, Inc., May 2007)*, and the *Interim Groundwater Characterization Report, Meyers Landfill, El Dorado County, California (Weston Solutions, Inc., (July 2012)*. A summary of all groundwater VOC data collected from monitoring wells from initial sampling through the quarterly event in May 2017 is provided in Table 2.

2.3.1 Groundwater Investigations

Groundwater investigations began at the Site in 1979 with installation of landfill wells M-1 through M-6 (Figure 2). Initially, the wells were used for frequent groundwater elevation measurements that were collected approximately twice per week from August to December 1980, and monthly from October 1981 to October 1982. No laboratory data is available during this timeframe. The wells were constructed at depths ranging from 25 to 75 feet below ground surface (bgs), however the elevations of the screened intervals of the wells were not documented. Two to four feet of soil cover and 18 to 36 feet of waste were reported in the boring logs of the wells.

Groundwater samples were first collected from wells M-1 through M-6 in June 1991 and analyzed for VOCs. Vinyl chloride was detected in samples collected in 1991 and 1992 at concentrations ranging from 27 to 44 µg/L and 6 to 47 µg/L, respectively. *Trans*-1,2-DCE was detected at a concentration of 34 µg/L in well M-5 during the March 1992 sampling event (USFS, 1996).

In October 1994, E&E, under contract with the USFS, installed monitoring well M-7 located southwest (upgradient) of the landfill to provide background data for the Site (Figure 2). Well M-7 was drilled to 51 feet bgs. The well was screened from 30 to 35 feet bgs because low-permeability silty sand was encountered between 35 and 51 feet bgs. It is now clear that well M-7 was installed in the zone of perched water identified west of the landfill. Well M-7 was damaged by a fallen tree and abandoned by PTEM in June 1999.

Also in October 1994, E&E sampled landfill wells M-4 and M-5 and newly installed upgradient well M-7. Groundwater samples collected from M-7 contained no VOCs, while samples collected from wells M-4 and M-5 contained 32 µg/L and 3.9 µg/L of vinyl chloride, respectively (E&E, 1995a).

In May 1995, E&E conducted several investigations at the Site that included the following:

- Drilling 15 borings in and around the landfill to depths ranging from 15 to 70 feet bgs (HP-1 through HP-15).
- Collecting 14 groundwater grab samples from the borings to assess VOCs in and around the landfill.
- Investigating the thickness of the cover material and landfill waste.
- Sampling the wells within the footprint of the landfill.

Of the 15 borings, 10 were drilled on the landfill plateau, two were drilled north of the landfill, one was drilled northeast of the landfill, and two were drilled southeast of the landfill (Figure 3). E&E abandoned well M-6 in 1995 because of a damaged casing and lack of sufficient water in the well (E&E, 1995b).

The investigations determined that there was 6 to 40 feet of waste beneath a few feet of cover soil. The HP-3 boring log indicates that dark gray, clayey silt was encountered at 14 feet bgs, which E&E correlated with the low-permeability silty sand encountered between 35 and 51 feet bgs in well M-7. However, no perched water was reported in boring HP-3. Borings HP-1, HP-5, and HP-8 encountered occasional silt layers beneath the landfill and above the groundwater table. Boring HP-5, located in the southeastern portion of the landfill near well M-2, encountered thin zones of perched water at 20 feet and 35 feet bgs. The remainder of the borings typically encountered silty sands and sands beneath the landfill and throughout the rest of the study area.

VOCs were detected in five groundwater grab samples collected from beneath the landfill waste mass. Vinyl chloride was detected in four of the groundwater samples at concentrations ranging from 12 to 81 µg/L. Other VOCs detected in groundwater samples collected from the borings included: 1.5 µg/L *cis*-1,2-DCE, 4.5 µg/L chlorobenzene, 1.1 µg/L 1,2-DCB, and 2.1 µg/L 1,4-DCB. Groundwater samples collected from landfill wells M-4 and M-5 contained 50 µg/L and 36 µg/L of vinyl chloride, respectively (E&E 1995b).

In July 1996, E&E installed three groundwater monitoring wells M-8 through M-10 (Figure 2). Well M-8 and M-9 were drilled between the landfill and Saxon Creek, and M-10 was drilled downgradient of the landfill. The wells were installed at depths ranging from 24 to 30 feet bgs.

In August 1996, Broadbent & Associates, Inc (BAI) sampled groundwater wells M-7 through M-10 for VOCs and metals. Vinyl chloride was detected in well M-10 at a concentration of 5.5 µg/L. No other VOCs were detected in the wells. The wells were sampled by BAI again in November 1996, and no VOCs were detected in these samples (BAI, 1996a,b).

Regional groundwater studies were performed in November 1996 by the United States Geological Survey (USGS) and were documented in the Water-Resources Investigations Report 00-4001 (USGS, 2000). As part of the investigation, the USGS collected groundwater data from selected monitoring wells at Meyers Landfill in November 1996. Groundwater elevation data was collected from nine wells located both at the Site and outside of the study area. A groundwater quality sample was collected from well M-9 and was analyzed for nitrogen, phosphorous, iron, specific conductance, and hydrogen ion measurement (pH). Conductivity measurements (542 micro Siemens per centimeter) and dissolved iron concentrations

(1,800 µg/L) detected in the sample were compared to data from other wells in the study area and were found to be elevated. The report attributed the elevated conductivity and iron results to the close proximity of the landfill (USGS, 2000).

In accordance with the April 1997 AOC, the JPA consultant, PTEM, conducted investigations to delineate the groundwater vinyl chloride plume. Between May 1997 and October 1999, PTEM installed 29 monitoring wells and one extraction well during six field events. Wells M-11 through M-13 were installed in May 1997; wells T-1 through T-8 were installed in October 1997; wells T-9, T-10, M-8A, D-1, and OW-1 through OW-3 were installed in May 1998; wells T-11, T-12, and wells D-2 through D-6 were installed in October 1998; wells T-7 and T-14 were installed in June 1999; and extraction well X-1 was installed in October 1999 (PTEM, 2000) (Figure 2).

To construct wells M-8A, M-11 through M-13, OW-1 through OW-6, and T-1 through T-14, borings were drilled using a hollow-stem auger drill rig to depths ranging from 22 to 143 feet bgs. These wells penetrate the upper portion of the shallow groundwater aquifer and typically have a screened interval of 20 feet. The differences in well completion depths are generally a result of topographic differences; however, the majority of the wells have been constructed at depths shallower than 30 feet bgs. Soil encountered in the borings was typically logged as sandy silts, sand, and gravelly sand. Wells D-1 through D-3 were drilled in the lower shallow aquifer and extend to 85 feet bgs. Wells D-2 and D-3 were drilled using hollow stem augers but with a plug placed at the end of the drill string, so no split spoon samples were collected to log lithology. Wells D-4 through D-6 and X-1 were drilled using a mud rotary drill rig to depths ranging from 103 to 190 feet bgs. Wells D-4, D-5, and D-6 were clustered with wells T-6/D-1, T-11/D-2 and T-12/D-3, respectively. These wells appear to have been logged from cuttings and may not accurately describe the lithology or identify contacts. The well logs for D-4, D-5, and D-6 describe a substantial clay layer at depths ranging from 120 to 177 feet bgs, 100 to 120 feet bgs, and 117 to 140 feet bgs, respectively; the wells were screened in coarse grained sediments below the clay layer and are considered to be representative of the deep aquifer.

Between July 1997 and November 2000, PTEM conducted quarterly monitoring events, which focused on defining the lateral and vertical extent of the VOC plume in groundwater. The quarterly monitoring events added sampling from the new wells as they were installed. PTEM did not sample the monitoring wells located within the perimeter of the landfill, but did collect groundwater grab samples beneath the landfill mass from borings completed in June 1999.

In 1999, PTEM, pursuant to the AOC with the County, drilled 23 borings on the plateau and within the footprint of the landfill and collected groundwater samples from 16 of the borings to determine the following (PTEM, 2000) (Figure 3):

- The thickness and condition of the landfill cover.
- Location and thickness of the waste.
- Condition of the waste.
- Concentration of methane in the waste.
- The location of waste with respect to groundwater.

Groundwater grab samples were also collected from the borings to determine the concentration of VOCs beneath the waste. Vinyl chloride was detected in groundwater samples collected from all 16 borings with concentrations ranging from 0.091 µg/L to 60.9 µg/L. Of the 23 borings drilled on the landfill plateau, 16 borings encountered waste. Liquid was observed in the lower three feet of waste in boring B-20, located in the southern portion of the landfill, likely from perched groundwater shown to be present in the south/southwest corner of the site. The thickness of the cover soil identified in the borings ranged from 3 to 15 feet. Waste thickness generally ranged from 3.5 feet to 37 feet and was found to extend 50 feet below the landfill surface at one location. PTEM utilized the waste and cover thicknesses to determine the volume of landfill waste, which was calculated to be 290,171 cubic yards.

In October 1999, PTEM performed a 72-hour groundwater pump test in well X-1. The pump was installed in extraction well X-1 and wells T1, T-2, T-6, D-1, D-4, M-11, and M-12 were used as observation wells (Figure 2). Extraction well X-1 was installed in October 1999 with a 60-foot screened interval extending from 25 to 85 feet bgs. Well X-1 was pumped at variable rates while water levels were measured in the observation wells with pressure transducers and an electronic data logger. PTEM was able to maintain a pumping rate of 50 gallons per minute (gpm) for 36 hours but the cone of depression did not fully stabilize during this period of time. PTEM used what data they had obtained to calculate various aquifer parameters. For instance, PTEM estimated the transmissivity of the aquifer to range from 1.62 to 4.36 feet squared per minute (17,486 to 46,948 gallons per day per foot). The specific yield was estimated to be around 10% in the deeper portion of the aquifer (PTEM, 2000). Geomatrix reevaluated the PTEM pump test data and calculated an average hydraulic conductivity ranging from 40 to 90 feet per day using the aquifer thickness of 65 feet assumed by PTEM, and 20 to 60 feet per day using an assumed aquifer thickness of 100 feet (Geomatrix, 2004). In addition, Geomatrix estimated groundwater flow velocities ranging from less than 1 foot per day to approximately 3 feet per day.

A groundwater treatment system was installed and began operation in December 1999 to extract and treat VOC impacted groundwater. The treatment system experienced persistent problems during the period of operation from 1999 to 2005. Poor performance was determined to be caused by extensive bio-fouling and iron sedimentation associated with the treatment system, as well as silt build-up in the pump (BAI, 2004).

In 2003 and 2004, the USFS entered into an AOC with the County, who contracted with Geomatrix to conduct a supplemental investigation at Meyers Landfill that focused on further delineating the vinyl chloride plume in groundwater and establishing a perimeter of monitoring wells that had not been impacted by VOCs.

The Geomatrix investigation included the installation of two groundwater monitoring wells and the completion of 13 borings that were used to collect groundwater samples (Figures 2 and 4). Well OW-6 is located 1,900 feet northeast of the landfill near Trout Creek and was drilled to a depth of 26.4 feet bgs. Well OW-7 is located 1,900 feet north-northeast of the landfill and was drilled to a depth of 20.9 feet bgs. Wells OW-6 and OW-7 were installed in May 2004 and screened in the upper shallow aquifer downgradient of the VOC plume (Figure 2).

The 13 Geomatrix borings ranged in depth from 20 to 36.5 feet bgs. Borings GB-1 through GB-11 were drilled in November 2003 and borings GB-12 and GB-13 were drilled in May 2004 (Figure 4). Borings GB-1 through GB-6 were drilled north and northwest of the landfill; GB-7, GB-8, and GB-10 through GB-13 were drilled north and northeast of the landfill. Boring GB-9 was drilled adjacent to wells T-6, D-1, and D-4 for the purpose of collecting a soil sample that was used in a bioremediation evaluation. The boring logs generally describe the subsurface sediments as sand and silty sand; however, clay layers were mentioned in logs of borings completed along the slopes of the moraine along Pioneer Trail and near Saxon Creek.

Groundwater samples were collected from 30 wells in October/November 2003 and 11 wells in June 2004. The samples were submitted to a laboratory for VOC analysis. Groundwater samples were also collected from selected wells for analyses of natural attenuation parameters used to evaluate potential groundwater remedial technologies (Geomatrix, 2004).

Shallow groundwater grab samples were collected from Geomatrix borings GB-1 though GB-8, GB-10, and GB-11 in November 2003 and borings GB-12 and GB-13 in May 2004. VOCs were detected in 4 of the 12 samples, with vinyl chloride reported at concentrations up to 67 µg/L in boring GB-8 near Saxon Creek. Vinyl chloride was also detected in groundwater samples collected from boring GB-6, on the west side of Saxon Creek, and from borings GB-7 and GB-10, east of Saxon Creek along Power Line Road. Groundwater samples collected from new wells OW-6 and OW-7 did not contain any VOCs (Geomatrix, 2004).

Geomatrix compared water level measurements collected from monitoring wells at the Site and determined that strong downward (negative) vertical hydraulic gradients are present at the Site. Further discussion of the vertical hydraulic gradients is presented in Section 3.3.2.

From fall 2005 through summer 2006, WESTON conducted several investigations under contract with USFS. Although the focus of these studies was the landfill OU and completion of the RI/FS for OU-1, the work included installation of four groundwater monitoring wells (MW-14, MW-15A, MW-15B and MW-16), two piezometers (P-1 and P-2), groundwater grab sampling from five of six borings (SB-1 through SB-6), groundwater grab sampling from six of 13 cone penetrometer test borings (CPT-1 through CPT-13) and quarterly groundwater monitoring. Results of the groundwater grab sampling are shown on Figure 4. Results from the well sampling are provided in Table 2. These investigations further defined the vertical and lateral extent of the VOC groundwater impact and helped refine the site hydrogeologic model.

WESTON conducted further investigations under contract with USFS between fall 2007 and fall 2010. Investigation activities included installation of 24 monitoring wells, conducting 13 quarterly groundwater monitoring events, sampling for bioattenuation parameters from five wells during four monitoring events, gaging and sampling surface water from Saxon Creek during eight monitoring events, and conducting an updated regional well search to identify potential groundwater receptors. The construction details of these 24 monitoring wells are included on Table 3. The wells are identified as MW-17A through MW-27, D-7 and D-9, and LF-1 through LF-3. Multiple wells were installed at different depths at some locations. Proposed well D-8 was not installed due to refusal during drilling, suspected to be due to encountering bedrock. These wells were placed to fill data gaps in the existing monitoring well network, including evaluating the presence and/or extent of VOCs in multiple groundwater zones, replacing well MW-14, adding downgradient sentinel monitoring locations, and evaluating potential interaction between landfill waste and upper groundwater. Locations of the wells are shown on Figure 2. Results of groundwater and surface water monitoring are provided in Table 2. Results of bioattenuation parameter sampling are provided in Table 7.

WESTON conducted a regional search of groundwater supply wells in the vicinity of the Site to identify potential contamination receptors. The well search identified one active well within a one-mile radius of the Site, the Elks Club No. 2 well located approximately 0.75 mile west of the Site (Figure 1). This well is an active public supply well operated by STPUD.

2.3.2 Surface Water Monitoring

In May 1975, the RWQCB collected samples from the intermittent stream, from leachate flowing off the landfill and from Saxon Creek. These results clearly identified leachate emanating from the landfill as evidenced by elevated turbidity and high concentrations of suspended solids, nitrogen, iron, zinc, alkalinity, and chemical oxygen demand compared to upstream and Saxon Creek samples (RWQCB, 1975). VOC analysis was not conducted during this initial sampling.

Surface water sampling of Saxon Creek was conducted in 1976 at a location (43-4) established by the USFS downstream of the landfill near the Power Line Road Bridge. In 1979, sample location (43-5) was established by the USFS upstream of the intermittent stream diversion outlet near the Fountain Place Road Bridge. Locations 43-4 and 43-5 were sampled annually from 1980 to 1989 for the same parameters as the RWQCB sampling (USFS, 1996).

After the discovery in 1998 of vinyl chloride in surface water samples collected from Saxon Creek downgradient of the landfill, the RWQCB requested surface water sampling of Saxon Creek for VOC analyses. Water samples have been collected since 1998 from various locations on Saxon Creek, both upgradient and downgradient of the landfill. Between 1998 and 2003, vinyl chloride was detected in surface water samples collected downgradient of the landfill at concentrations ranging from 0.0526 to 0.28 µg/L. Quarterly surface water sampling of Saxon Creek was conducted by WESTON from September 2005 through October 2009. Neither vinyl chloride nor other VOCs were detected in water samples collected from Saxon Creek between 2003 and 2009 (Table 2).

3. PHYSICAL CHARACTERISTICS OF THE STUDY AREA

This section of the report is reproduced almost entirely from Section 3 of the Interim Groundwater Characterization Report (WESTON, July 2012).

3.1 SURFACE FEATURES

Until the summer of 2010, the landfill and the area immediately surrounding the landfill formed a plateau with a surface that was relatively flat. The entire plateau consisted of approximately 17 acres of which 11 acres were occupied by the landfill itself. The plateau slopes to the floor of the valley on the north and south edges. The landfill abuts an unnamed ridge along the east side and the eastern slope of a lateral moraine located on the west edge of the landfill (Figure 2). Access to the landfill is restricted to vehicles by a Forest Service gate on Garbage Dump Road at Pioneer Trail, and further by a gate at the entrance from Garbage Dump Road to the landfill.

Landfill cap construction activities begun in the summer 2010 have significantly altered the features at the Site. The current configuration of the landfill is a broad mound with 6 to 25 percent slopes along the lateral margins. Landfill waste was removed from east of the sewer line and consolidated within the main waste mass to the west. The consolidated landfill was covered with clean soil, an impermeable geosynthetic liner, and vegetative cover soil. Surface water diversions and storm water infiltration and settling basins have been installed around the landfill to control surface runoff. A new french drain has been installed west of the landfill. The multilayer cap was completed in 2012.

The main semi-permanent structures on the landfill are three remaining groundwater monitoring wells, two landfill gas vents and concrete V-ditches and surface water control features. The groundwater monitor wells are completed within concrete vault boxes approximately one foot above grade. The landfill gas vents consist of eight-inch diameter polyvinyl chloride (PVC) pipes extending approximately ten feet above grade. The vents are within fenced enclosures. Manholes associated with the STPUD Trout Creek trunk sewer that runs along the east side of the waste are present just outside the capped waste. The sewer line generally trends north-northeast, turns sharply to the west at the base of the northern slope of the landfill, and then curves again to the northeast along the west side of the electrical substation (Figure 2). The manholes are located at intervals and at the bends in the sewer pipe and typically extend a few feet above ground surface.

The main manmade surface feature north of the landfill is a fenced enclosure containing the electrical substation. Three aerial power lines run from the substation to the west, north, and northeast. Drainage ditches along the perimeter of the substation convey runoff to Saxon Creek.

Other surface features in the vicinity of the Site include a network of gated dirt roads paralleling the northern power lines and other dirt roads and trails used for recreational purposes and forest access (Figure 2).

3.2 SURFACE HYDROLOGY

Federal Emergency Management Agency (FEMA) flood zone maps show the area around Meyers Landfill and nearby streams as Zone D – *Areas of undetermined, but possible, flood hazards*. However, approximately 1,500 feet upstream from the landfill, the FEMA flood zone maps show Saxon Creek as Zone A – *Areas of 100-year flood; base flood elevations and flood hazard factors not determined* (FEMA, 2008). These maps do not identify the landfill, so it is expected that flooding would be limited to the ridge east of the landfill and not likely to extend to the landfill.

The USGS has conducted stream flow measurements in Saxon Creek as part of a surface water and groundwater study of the Upper Truckee River and Trout Creek watersheds. The investigation showed that groundwater is near the ground surface and that Saxon Creek maintained a base flow during the duration of the study (USGS, 2000).

Prior to the construction of the landfill, the Site consisted of a valley that had been incised by an intermittent stream that drained to Saxon Creek approximately 500 feet north of the landfill. Saxon Creek is a tributary of Trout Creek, which eventually flows to the Truckee Marsh and Lake Tahoe. The landfill was constructed in the intermittent stream valley, and the intermittent stream was routed through a 24-inch diameter corrugated metal pipe (CMP) culvert placed in the bottom of the original stream channel. Approximately 1,300 feet of the stream was buried beneath 30 to 50 feet of solid waste and fill dirt. The upper reaches of the intermittent stream above (south of) the landfill appears to be a gaining stream as evidenced by thick riparian vegetation and the presence of trace water even during summer months. This occurrence is a result of high perched groundwater levels present to the south west of the Site, especially during spring months after melting of the snow pack.

A 1976 video inspection performed on the CMP culvert situated in the stream channel beneath the landfill revealed that the pipe was blocked by debris, a separated joint, or a crushed section of pipe (USFS, 1996). In an effort to control erosion and to reduce leachate generation, the original CMP culvert was plugged and the intermittent stream was diverted near the south end of the landfill. The intermittent stream and surface water were directed to Saxon Creek through a series of surface drains and culverts. The diversion was completed during the 1975 and 1976 landfill cover renovation and erosion control work performed by the USFS. Other drainage and water diversion systems that were constructed during this timeframe included the following:

- A french drain was installed in the southwest corner of the landfill to collect and transport water from a spring south of the landfill and perched groundwater area south and west of the landfill to the drainage culvert located adjacent to the south side of the landfill.
- Collection galleries were located in the southern portion of the landfill to drain water from the landfill surface to the drainage culvert located adjacent to the south side of the landfill.
- Two surface drains were located in the central portion of the landfill along the access road allowing flow to the southeast to the intermittent stream diversion culvert.
- One surface drain was located in the northwest corner of the landfill to drain landfill surface water westward to a small intermittent stream drainage northwest of the landfill.
- One surface drain was located in the northern portion of the landfill to drain surface water from the north end of the landfill to the base of the north slope.

Until cap construction activities began in summer 2010, the surface drain system was only partially effective as the landfill had settled differentially since 1976 and water accumulated in depressions during the spring snowmelt and times of heavy precipitation. The cap construction work has significantly altered the topography and drainage from the landfill. Storm water controls were put in place to control sediment run-off from the construction area. Final cap construction was completed in 2012 and surface water control off the landfill is improved. A new french drain system has been installed along the west side of the landfill to collect and drain the perched water occurring in this area. Seasonal erosion damage to surface water drainage controls has occurred to some degree each year since the 2011-2012 cap construction. Maintenance of the drainage systems will need to be conducted annually to protect the landfill cap from damage.

3.3 REGIONAL GEOLOGY AND HYDROGEOLOGY

3.3.1 Regional Geology

The Site is located in Lake Valley, which was created by a combination of tectonic and glacial processes. Basin and range fault bounded blocks created Lake Tahoe between the granitic mountains of the Sierra Nevada crest to the west and the Carson Range to the east between 2 million and 3 million years ago. The Tahoe block (graben) is bordered to the north by the Holocene age North Tahoe-Incline Village fault zone, to the west by the West Tahoe-Dollar Point fault, and to the east by the East Tahoe fault zone. Individual faults of the East Tahoe fault zone are located within 3,000 feet north and south of the Site. An unnamed fault is mapped, noted with some uncertainty, along the west side of Pioneer Trail. Figure 5 illustrates the geology of the area.

In 2001, the US Army Corps of Engineers (USACE) initiated the Lake Tahoe Basin Framework Study Groundwater Evaluation, which was designed to enhance the understanding of the role groundwater plays in the processes reducing lake clarity. The study evaluated and modeled the groundwater in the South Lake Tahoe / Stateline aquifer (USACE, 2003). The following descriptions are, for the most part, from the USACE study.

Basin-fill deposits generally consist of unconsolidated glacial, lake, and stream sediments. These sedimentary deposits fill the lower reaches of the canyons that drain toward Lake Tahoe and underlie the relatively flat valley floors. The thicknesses of these deposits vary across the basin. In general, the basin-fill deposits are relatively thin toward the margins of the Basin and in areas where they cover shallow bedrock areas exposed within the Basin. The basin-fill deposits typically thicken away from these bedrock areas to fill the deepest portions of the Basin where the sediments range in thickness from approximately 600 feet to more than 1,000 feet.

Bedrock in the vicinity of the Site was encountered at a depth of 228 bgs (at approximate elevation of 6,052 feet) as documented in boring logs describing STPUD's Elks Club Well No. 2, located approximately 0.75 mile west of the Site. Boring logs completed for wells D-4 and D-5 indicate that bedrock was possibly encountered at depths between 150 and 187 feet bgs (approximate elevations ranging from 6,156 to 6,182 feet); however, it was not confirmed whether it was actually bedrock or boulders typical of the basal layer of glacial sediments that were encountered (PTEM, 2000). During WESTON's previous phase of investigation, granitic rock was encountered in boring D-8 at 126 feet bgs (approximate elevation 6,231 feet) which may also represent bedrock or a basal boulder layer. This boring was located south of the landfill near wells M-8/M-8A, adjacent to a topographic rise (Figure 2).

Glacial deposits form the majority of the aquifers in the Basin. Glaciations in the basin began around 1.5 million years ago when all but the highest peaks in the Sierra Nevada were inundated by ice. Evidence of four major glacial periods are found in the basin and are, from oldest to youngest, the Hobart, Donner, Tahoe and Tioga. Valley glaciers advanced north toward Lake Tahoe through the Upper Truckee River Valley. As these glaciers advanced and receded, lateral moraines formed along the edges of the glaciers and terminal moraines formed in front of the glaciers. These moraine deposits are typically jumbled deposits of clay- to boulder-size material, with moderate permeability. Sediment-laden melt-waters flowed from the receding glaciers north toward Lake Tahoe. These streams dropped their sediment loads in broad coalescing flood fans referred to as outwash plains. These glacial outwash deposits are composed of layered beds of well-sorted gravel, sand and silt-size material, with moderate to high permeability. Glacial streams deposited sediment directly into Lake Tahoe forming thick deltas of interlayered sand and fine-grained silt and clay. These delta sequences grade laterally and include the following sequences: lakeshore deposits, consisting of moderately well sorted sand and gravel deposits with relatively high permeability; marsh deposits, consisting of fine-grained sand, silt and clay; and lake deposits, consisting of silt and clay. Both the marsh and lake deposits have relatively low permeability. The glacial outwash and delta deposits form excellent groundwater reservoirs (STPUD, 2004).

The Hobart and Donner glaciations flowed out of Christmas Valley and covered the Meyers area. The ice advanced northward and was deflected to the east by Twin Peaks and deposited the

Airport Moraine located along Pioneer Trail. Pleistocene glacial outwash deposits are found east of the site between Saxon Creek and mountains of the Carson Range.

Several times during period of glaciation, ice dams blocked the Truckee River canyon north of Tahoe City, thus raising the level of Lake Tahoe up to 600 feet above present levels or 6,800 feet above MSL. The dam is believed to have been breached several times resulting in periodic lowering of the lake level. Each subsequent glacial period had raised the lake level to various degrees but to a lesser extent than the older Hobart and Donner glacial periods. The highest lake levels are estimated to be 7,000 feet MSL resulting from lava flows damming the lake’s outlet between 2.5 and 1.3 million years ago. The higher lake levels result in an expansive lake deposits in the Upper Truckee River Valley. Silt and clay layers encountered beneath the Site during subsurface investigations may be lake (lacustrine) deposits deposited during the periods of higher lake levels during these periods of glaciation and from the earlier lava dam. The fine grained, commonly laminated and rhythmically bedded or varved, sediments observed between the upper and lower aquifers are consistent with lacustrine deposits.

During interglacial periods sedimentary processes in the lake would have been dominated by fine-grained deposition. As glaciers were growing and shrinking, sediment loads in streams would increase, depositing silts in Lake Valley. Deltaic deposits occurred near the higher lake levels shoreline while suspension settling of finer sediments occurred away from the shore resulting in continuous fine-grained layers of silt and clay. These deposits would have been thickest over topographic lows and thinner over highs. The silt and clay layers also would have pinched out towards the lake shore where wave-based activity would have removed the fine sediments.

The geologic map of the Lake Tahoe Basin shows the landfill located on the eastern slope of a moraine with an axis trending along Pioneer Trail (Figure 5). The California Geological Survey (CGS) reports the moraine as till from the Pleistocene epoch that is deeply weathered, moderately to well compacted, unsorted bouldery to clayey gravel, locally including glacial outwash deposits. Soils observed in wells MW-26 and MW-27 drilled along Pioneer Trail include a sequence of sands to silty sands, overlying fine-grained silts with local sand beds or intervals. Samples could not be collected from a 40-foot interval in MW-27 due to very difficult drilling, which may represent a gravel or boulder unit. A boulder was also encountered during the drilling of well D-7 at approximately 238 feet bgs. The moraine is reported to have been deposited prior to the Tioga and Tahoe glacial periods. Flood plain deposits along Saxon Creek are identified on the geologic map on the eastern slope of the landfill. The CGS reports the flood plain deposits from the Holocene epoch are moderately to poorly sorted, gravelly to silty sand, and sandy to clayey silts (CGS, 2005).

3.3.2 Regional Hydrogeology

The Site is located in the Tahoe Valley South Subbasin, identified by the California Department of Water Resources (DWR) as Groundwater Basin Number: 6-5.01, and occupies 23 square miles. The subbasin is part of the Tahoe Valley Groundwater Basin and within the larger structural feature commonly referred to as the Lake Tahoe Basin. The Tahoe Valley Groundwater Basin consists of three alluvial areas surrounding the California side of the lake on the south, west, and north. The Tahoe Valley South Subbasin occupies a roughly triangular area and is bounded on the southwest and southeast by the Sierra Nevada mountain range, on the north by the southern shore of Lake Tahoe, and to the northeast by the California-Nevada state line. The southern boundary extends about 3 miles south of the town of Meyers and forms the apex of the triangle. Elevations within the subbasin range from 6,225 feet MSL at lake level to above 6,500 feet MSL in the south. The Upper Truckee River flows north along the entire length of the basin and drains into Lake Tahoe. The river is joined by Grass Lake Creek and Big Meadow Creek near the southern end of the basin, Angora Creek centrally, and Trout Creek near the northern extent of the basin. Average annual precipitation in the subbasin ranges from 23 to 49 inches, increasing from north to south (DWR, 2004).

Regional groundwater studies were performed in November 1996 by the USGS. The studies were documented in the Water-Resources Investigations Report 00-4001 entitled: *Surface and Ground-Water Characteristics in the Upper Truckee River and Trout Creek Watersheds, South Lake Tahoe, California and Nevada July-December 1996* (USGS, 2000). Comparison of water level measurements collected from environmental monitoring wells and STPUD wells show that strong downward (negative) vertical hydraulic gradients are present in the Basin. The USGS reported negative vertical gradients in the Meyers area, in the South Y area, and along the

Highway 50 corridor through the north-central portion of the Basin. Downward gradients typically occur in recharge areas within a groundwater basin (STPUD, 2004).

3.3.3 Water Bearing Formations

The principal source of groundwater in the Tahoe Valley South Subbasin is Tertiary and Quaternary age glacial, fluvial, and lacustrine sediments, collectively referred to as basin-fill deposits. Each of the three depositional processes gives rise to distinct sediment types with variable hydraulic properties. Specific-yield estimates in the basin range from 6 to 20% and average about 10%. Most water wells drilled in the basin are completed in basin-fill deposits, where groundwater occurs under confined, semi-confined, and unconfined conditions. Pre-Cretaceous granitic rocks form the base of the aquifer.

Basin Fill Deposits **—** Glacial outwash sediments, deposited on prograding deltas, are the predominant sediments within the basin, and are typically composed of soil ranging from fine silt to large boulders that have been sorted and stratified by the action of water flowing from the glacier. Thickness of the basin fill deposits may range from 1,600 to 1,900 feet in the South Lake Tahoe area, but typically range from 50 to 150 feet. Permeability of these deposits can be moderate to high. Glacial sediments consisting of moraine deposits also occur within the basin. These deposits are generally unsorted, have high clay content, and are produced by the grinding glacial action. They typically have moderate permeability.

Lacustrine Deposits **—** These deposits are widespread and discontinuous, and are a result of fluctuating lake levels. They occur as high as 600 feet above the current lake level. Deposits containing well-sorted beach sand have relatively high permeability, but those with high silt and clay content have lower permeability.

Holocene Fluvial Deposits **—** Holocene fluvial deposits are located along stream channels; however, their limited thickness and extent makes them relatively insignificant in the basin (DWR 2004).

3.4 SITE GEOLOGY AND HYDROGEOLOGY

As discussed in Section 2.3, subsurface investigations have been on-going at the Site since 1979. These investigations have provided a large data base for evaluating the subsurface geology and hydrogeology. Approximately 97 wells have been installed and monitored to some degree, and over 70 soil borings have been drilled, not including soil vapor probes and test trenches. The majority of these investigations have focused on the uppermost soils and water table, but deep borings have been drilled to over 280 feet. Wells and borings have been installed using variety of methods including direct push, CPT, hollow stem auger, mud rotary, air rotary casing hammer and sonic drilling methods. As of July 2015, there are 85 wells at the Site and 83 available for monitoring (wells T-7 and MW-26 are damaged).

The following section describes the geology and hydrogeology beneath the Site. An attempt is made to capture and incorporate information from all previous investigations. The subsurface conditions are complex, and correlation of lithologic logs installed by various contractors using a variety of drilling and sampling methods is challenging. The water bearing zones are not always contiguous and hydraulic heads between wells screened at the same elevations do not always correlate. Some interpretation regarding flow directions and water-bearing zones are biased based on contaminant distribution, and it appears that plume migration, particularly in the deeper zones, is likely controlled as much by preferential transport in coarser-grained units as by hydraulic gradients. Lithologic logs of the borings drilled as part of this investigation are provided in Appendix A. Table 3 provides well construction summary details for all monitoring wells installed at the Site.

3.4.1 Site Geology

The stratigraphic sequence beneath the Site is characterized by two major depositional systems; upper glacial and fluvial sediments underlain by interbedded fine-grained lake sediments. The upper glacial and fluvial deposits generally consist of light yellowish-brown to dark reddish-brown, fine to coarse sand and silty sand interbedded with thinner layers of silt and sandy silt. This upper sand unit is upwards of 150 feet thick on the top of the moraine along Pioneer Trail (well MW-26) and as thin as 30-40 feet near Saxon Creek (well MW-19A/B/C). First groundwater occurs in this unit and often causes heaving sands during drilling.

The lower lake deposits generally consist of fairly dense silt and clay with thin to fairly thick (10-20 feet) sand packages. The fine grained silt and clay is typically olive to dark gray, contains thin lamina or varves typical of lacustrine deposits, and yields water poorly. The sand units are generally fine to medium grained and yield water readily. The silt and clay units can be fairly thick (up to around 150 feet in MW-27) and can contain few to no significant sand intervals as in MW-26. At some locations, the contact between the upper fluvial and lower lacustrine deposits is fairly sharp, for example between 115-120 feet bgs in MW-24. In other locations, the contact is gradational, for example between 90-110 feet bgs in MW-17A/B.

Elks Club Well No. 2, owned by STPUD, is a water supply well located approximately 0.75 mile west of the Site. The well log indicates that silt (with varves) was encountered from 9 to 35 feet bgs, and clay that is underlain by sands and gravels was encountered from 35 to 88 feet bgs. Bedrock was encountered at 228 feet bgs (STPUD, 2006). The elevation of the silt and clay units encountered in this well correlate with similar strata observed in wells completed at the Site which suggests that the silt and clay layers may be somewhat contiguous throughout the study area.

3.4.2 Site Hydrogeology

Groundwater at the Site has been divided by WESTON into three major water bearing zones, the Upper Groundwater Zone (UGZ), the Middle Groundwater Zone (MGZ) and the Lower Groundwater Zone (LGZ). A localized Perched Groundwater Zone (PGZ) is found west and south of the landfill. Previous studies referred to two zones, the upper and lower aquifers. The delineation of a MGZ has been proposed to address contamination identified in some intermediate-screened wells in a particular well cluster. For instance, chemical data from the cluster wells OW-3/MW-18A/MW-18B show no vinyl chloride in the shallowest and deepest screened wells (OW-3 and MW-18B), but show vinyl chloride that has ranged up to 40-50 µg/L in the intermediate screened well MW-18A (Table 2).

As previously discussed, hydrogeology at the Site is complex making delineation of three distinct water bearing zones at a particular location difficult. Water level measurements from similarly impacted wells do not always show a hydraulic gradient consistent with the geometry of the plume. In order to try to delineate the zones, chemical data is used along with screen elevations, hydraulic head, and lithology to model the distribution of the three water bearing zones and the geometry of the contaminant plume. The distribution of VOCs in the groundwater indicates the UGZ and MGZ are hydrologically interconnected at the Site, but the data also suggest that transport of VOCs is affected by factors in addition to groundwater gradient, including dipping confining layers and occurrence of more permeable units at different stratigraphic intervals. For example, in the UGZ, the plume extends west of the landfill in the upgradient direction in the vicinity of well PU-2, which appears to be controlled in part by westward-dipping stratigraphy in that area. In the MGZ, the plume is found within sand units at varied elevations and the wells in which VOCs are detected exhibit varied hydraulic heads within an overall northerly gradient. For example, the MGZ well MW-24 is the furthest downgradient well containing VOCs but has higher groundwater elevation than well MW-18A which also contains VOCs and is closer to the landfill. In another example, VOCs in MW-21A appear to be showing a slight increasing trend in concentrations although this well is located cross-gradient from the main axis of the plume. The difference in overall gradients between the UGZ and MGZ results in divergence of plumes between the two units.

Perched Groundwater Zone (PGZ)

Gently westward-dipping, fine grained deposits of silts and clays have been found west and south of the landfill that support perched groundwater that surfaces in seeps and springs near the landfill. The presence of perched water along the west and southwest side of the landfill was visually observed during installation of the new french drain system in cuts into the adjoining slope along Garbage Dump Road, and from water level measurements recorded from former piezometers P-1 and P-2, and former well M-7. Perched groundwater was also observed during the remedial activities in 1975 to 1976 and waste consolidation activities south of the landfill in summer 2010. Perched groundwater was observed during drilling of wells T-9, MW-16, MW-29, MW-30, and MW-31A. The approximate area of perched groundwater is shown on Figures 8 and 9.

Effectiveness of the new french drain system was evaluated by installation of piezometers to monitor water levels in the PGZ. Initially, piezometers PP-1 and PP-3 were installed to the west of the french drain in September 2011, and PP-2 and PP-4 were installed between the french drain and the landfill in June 2012. Water level observations indicated that in general, groundwater in the PGZ was flowing east toward the french drain and being captured and discharged, although at times, water levels on the landfill side of the french drain were higher than those to the west, and perched groundwater was suspected to be periodically flowing from the landfill toward the french drain. In order to evaluate potential migration of contaminants to the french drain, sampling of the french drain outfall was initiated in December 2014, and vinyl chloride was initially detected at 5 µg/L, along with 0.86 µg/L of *cis*-1,2-DCE. Subsequent sampling rounds continued to show detectable vinyl chloride, although at lower concentrations through the most recent monitoring event in May 2017. Additional exploration of the PGZ was conducted to further evaluate groundwater flow and potential impacts to the PGZ. Piezometers PP-1 and PP-3 were sampled beginning in December 2014 and PP-2 and PP-4 were sampled beginning in March 2015. Vinyl chloride was detected in PP-4 at a concentration of 2.2 µg/L in that quarter. An additional nine piezometers (PP-5 through PP-13) were installed on either side and to the north of the french drain in July 2015 to further evaluate groundwater levels and potential impacts in the PGZ. Results of subsequent monitoring of the PGZ are discussed below in Section 5.1.2.

Groundwater flow in the PGZ differs with location along the french drain. In the area immediately west of the french drain, gradient is toward the east-southeast (toward the french drain) at steep gradients of 0.03 to 0.09 foot/foot. Farther west, gradients are toward the west, following the surface of the underlying westward-dipping aquitard. Between the french drain and the landfill, gradients generally have a significant component toward the french drain, with an overall decrease in groundwater elevations from north to south. During the summer and fall, perched groundwater is typically absent in the area east of the french drain. Groundwater gradients in the PGZ in the vicinity of the french drain for fall 2016 are shown in Figure 6, and for spring 2017 are shown in Figure 7.

Upper Groundwater Zone (UGZ)

The UGZ occurs in the upper glacial and fluvial deposits and is considered the water table aquifer. Groundwater wells installed in the UGZ range in depth from ground surface in lower portions of the site near Saxon Creek to 140 feet in wells located on the moraine ridge along

Pioneer Trail. The well screens bracket the UGZ from elevations of approximately 6,287 to 6,356 feet MSL (Table 3).

Groundwater flow at shallow depths is controlled by topography and flows eastward at a relatively steep gradient away from the Pioneer Trail ridge towards the intermittent stream valley where it flows north- to northeastward at a considerably shallower gradient (Figures 8 and 9). The groundwater beneath the landfill may be locally influenced by silty sediments beneath the ridge separating the intermittent stream valley from Saxon Creek, directing the groundwater flow towards the north.

Beneath and immediately downgradient (north) of the landfill, the upper portions of the UGZ have been impacted by VOCs as shown in wells LF-1, LF-2, LF-3, T-1, T-2, T-3, T-6 and T-9 (Table 2). As noted previously, the VOC plume extends a short distance west of the landfill as shown in wells PU-1 and PU-2 (in the upgradient direction). Farther downgradient, the upper UGZ is not impacted by VOCs outside a narrow, north-northeasterly trending band as shown in wells T-13, OW-3, and OW-5 (Figures 17 and 18).

Middle Groundwater Zone (MGZ)

The MGZ occurs near the contact between upper glacial and fluvial deposits and the underlying lacustrine deposits. In most cases, the MGZ occurs in an uppermost coarse unit within the lacustrine silt and clay, for example in MW-18A. In some locations, the MGZ appears to occur at the base of the upper glacial and fluvial deposits, for example in D-1. Groundwater wells installed in the MGZ range in depth from 60 feet in lower portions of the site near Saxon Creek to 284 feet in wells located on the moraine along Pioneer Trail. The well screens bracket the MGZ from an elevation of approximately 6,218 to 6,283 feet MSL (Table 3).

Groundwater flow in the MGZ is difficult to calculate from depth to water measurements collected from wells that are considered to be screened in this zone due to differing hydraulic heads in various horizons within the aquifer. However, using groundwater elevations from selected wells and the assumption that sand channels are in hydraulic communication with each other, a general flow direction has been calculated to be towards the north (Figures 10 and 11).

The variability of the thickness and location of the buried stream channels are most notable in the MGZ wells along Pioneer Trail where well MW-26 encountered only a few thin sand layers between the elevations of 6,199 to 6,209 feet MSL, while over 40 feet of sand was found in well MW-27 between the elevations of 6,233 to 6,273 feet MSL. (As discussed in the Lower Groundwater Zone below, MW-27 may be screened in the upper LGZ or in an interval where the two zones merge together, based on groundwater elevations). The VOC plume has been detected in MGZ wells D-1, D-2, MW-17A, MW-18A, MW-21A, MW-22A, MW-24, MW-29, and MW-31B (Table 2). Although vinyl chloride has not been detected in MGZ wells D-3 and MW-17B, low concentrations of PCE (approximately 1 µg/L or less) have been detected in D-3, and low concentrations of PCE and *cis*-1,2-DCE (approximately 2 µg/L or less and less than 1 µg/L, respectively) have been detected in MW-17B.

A generally low to moderate downward vertical gradient (up to 0.559 foot/foot with a vertical elevation difference of more than 30 feet in well pair MW-31A and MW-31B) is noted between the UGZ and MGZ in most clustered well pairs. One to five well pairs may exhibit slight upward gradients in a given quarter ranging to 0.097 foot/foot. Well pairs in which upward gradients have been observed include T-6/D-1, T-12/D-3, T-14/MW-21A, MW-15A/MW-15B, and MW-19A/MW-19B. Of these five well clusters, three are located near Saxon Creek. The close relationship between hydraulic heads in the UGZ and MGZ can be seen in water level hydrographs depicted in Figures 21 through 24.

As previously noted, the UGZ and MGZ are interconnected based on the presence of VOCs detected in MGZ wells. The VOC plume flow direction in the area where the zones are interconnected appears to be controlled by a preferential pathway in the lower portion of the UGZ, likely at the contact of MGZ where lacustrine silts appear to have been scoured in a northerly direction by either fluvial or glacial processes. The VOC plume flow direction deeper in the MGZ is likely controlled by other sand deposits trending in the same general direction.

Lower Groundwater Zone (LGZ)

Similar to the MGZ, groundwater in the LGZ occurs in sand units within or beneath lacustrine deposits. These sand units are considered to be deltaic sediments within the lake deposits since they appear to be thicker and more extensive than sand units typical of the MGZ. Groundwater wells installed in the LGZ range in depth from 90 feet bgs (MW-20C near Saxon Creek) to 255 feet bgs (D-7 northwest of the landfill). The well screens bracket the LGZ from an elevation of approximately 6,127 to 6,246 feet MSL (Table 3). The LGZ wells at the site are D-4, D-5, D-6, D-7, and D-9. Wells MW-18B, MW-19C, MW-20C and MW-27 may be installed in the uppermost portions of the LGZ as they are screened at similar elevations and exhibit groundwater elevations that correspond with other LGZ wells.

Groundwater flow in the LGZ is towards the east-northeast at a very shallow gradient away from the Carson Range towards the axis of Lake Valley (Figures 12 and 13). A strong downward vertical gradient up to 0.821 foot/foot is noted between the MGZ and LGZ in all clustered well pairs historically. Vinyl chloride has not been detected in any of the LGZ wells.

4. RECENT INVESTIGATIONS

WESTON conducted additional investigations between spring quarter (July) 2011 through the spring quarter 2017. In general, the work was conducted in accordance with the *Supplemental Remedial Investigation Work Plan, Operable Unit – 2, Meyers Landfill, El Dorado County, California (WESTON, March 2008)*. However, the final number of new well installations, and the scope and number of quarterly monitoring events increased as new data identified additional data gaps. The following scope of work was completed as part of the most recent investigation.

- Drill and install a total of seven (7) new monitoring wells to fill data gaps in the existing monitoring well network, including four UGZ wells (MW-30, MW-31A, PU-1, and PU-2) and three MGZ wells (MW-28, MW-29, and MW-31B). One additional boring (Boring MW-30) was drilled for installation of a background monitoring well southwest of the landfill, but groundwater was not encountered at the anticipated depth and the boring was grouted. Well MW-30 was installed closer to the landfill during a subsequent effort.
- Drill and install a total of 13 piezometers to monitor the PGZ in the vicinity of the new french drain. Two additional borings drilled in the PGZ north of the french drain did not encounter water and piezometers were not constructed.
- Conduct 24 quarters of monitoring well gaging and sampling, from spring quarter (July) 2011 through spring quarter 2017.
- Gage flow of Saxon Creek during 12 monitoring events and sample Saxon Creek during 12 of the 24 quarterly monitoring events to evaluate potential groundwater/surface water interaction.
- Collect water samples for analysis of VOCs and gage flow from french drain outfall during the past ten of the 24 monitoring events and sample the culverted intermittent stream outfall during the past nine of the 24 monitoring events.
- Collect additional samples for bioattenuation parameters from five wells during six of the 24 quarterly monitoring events to provide background and baseline data for evaluating the potential for natural attenuation.

- Refine site conceptual model, evaluate concentration trends, and conduct fate and transport modeling to simulate VOC plume behavior under assumed existing hydrological and biochemical conditions.

Results of each quarterly monitoring event have been summarized in quarterly reports.

4.1 MONITORING WELL INSTALLATION

Monitoring wells and piezometers all were installed using hollow stem auger drilling methods. A summary of the well construction details for all Site monitoring wells is provided in Table 3. Summary of the well installation rationale for this phase of investigation is provided in Table 4. Boring logs and well construction details for the most recent wells and piezometers are provided in Appendix A.

The drilling and well installation activities were overseen by a WESTON geologist. Lithologic logs were prepared for each well (or each well cluster). For locations where multiple (cluster) wells were installed, only the deepest boring at each location was logged as the wells were located within a few feet of each other. In general, soil samples for lithologic identification were collected approximately every five feet using a split spoon sampler, but continuous sampling was conducted in some instances, collecting the samples in a core barrel.

All wells were constructed of 2-inch diameter, polyvinyl chloride (PVC) screen and casing. The majority of the wells (17 of 20) were constructed of Schedule-40 PVC, but the three deeper wells (MW-28, MW-29, and MW-31B) were constructed of Schedule-80 PVC. Screen slot size was 0.01-inch as shown on the well logs. All wells were completed approximately three feet above grade and protected within a locking, steel stove pipe. Protective bollards were installed around wells in areas of higher off-road use (mainly snowmobiles).

Each well was developed after waiting a minimum of 48 hours after well completion in an attempt to remove fines and improve water clarity. Development consisted of swabbing with a surge block and bailing to remove fines. Most wells were not pumped due to the persistent accumulation of very fine sand and silt in the wells which would damage pumps, and several PGZ piezometers, because of insufficient water to pump.

During drilling, sampling, and well development, all downhole equipment was either steam-cleaned or hand washed using a soap solution and water rinse between sampling depths and between borings. The new wells were surveyed by a licensed professional land surveyor to locate the horizontal and vertical location of the top of each well casing.

4.1.1 Perched Groundwater Zone (PGZ) Wells

A total of 13 shallow piezometers were installed in the PGZ in three phases. Piezometers PP-1 and PP-3 were installed in September 2011, PP-2 and PP-4 were installed in June 2012 after the french drain was installed, and PP-5 through PP-13 were installed in July 2015. The purpose of the initial four piezometers was to evaluate the PGZ in the vicinity of the new french drain and effectiveness of the french drain. The nine additional piezometers were installed to asses VOCs in groundwater along the french drain after vinyl chloride was detected in the french drain outfall and PP-4. PGZ wells and borings are shown on Figure 2 as well as on Figures 15 and 16.

4.1.2 Upper Groundwater Zone (UGZ) Wells

A total of four wells were installed in the UGZ in three phases. Wells PU-1 and PU-2 were installed in September 2011, well MW-30 in June 2012, and well MW-31A in September 2014 (Figure 2). Wells PU-1, PU-2, and MW-31A were installed to assess the UGZ to the west of the landfill. Well MW-30 was installed to assess the UGZ upgradient (approximately 170 feet south) of the landfill with a screened interval at the same elevation as the landfill UGZ wells. An initial attempt to install well MW-30 at a location approximately 530 feet southwest of the landfill was not successful where fine grained silts were encountered at the anticipated depth of the UGZ.

4.1.3 Middle Groundwater Zone (MGZ) Wells

A total of three additional wells were installed in the MGZ in three phases. Well MW-28 was installed in September 2011, well MW-29 was installed in November 2011, and well MW-31B was installed in September 2014 (Figure 2). MW-28 was installed to delineate the west side of the main portion of the MGZ plume between the landfill and MW-24. Wells MW-29 and MW-31B were installed to delineate the MGZ plume to the west and northwest of the landfill.

4.1.4 Lower Groundwater Zone (LGZ) Wells

No additional wells were installed in the LGZ during the most recent phase of investigation between spring 2011 and spring 2017.

4.2 QUARTERLY MONITORING

Quarterly groundwater monitoring was conducted for 24 quarters from spring (July) 2011 through spring 2017. Quarterly monitoring activities included:

- Measure depth to groundwater in all accessible monitoring wells.
- Collect groundwater samples for VOC analyses from wells selected to provide optimal coverage of the plume and evaluate potential trends. Most wells that are part of the current monitoring program are sampled each quarter. Certain wells are sampled in the spring and fall quarters (Table 2). Monitoring was not conducted during the winter 2017 quarter (usually conducted in March).
- Evaluate groundwater conditions in newly installed wells.
- Gage the stream discharge during 12 quarterly events and sample Saxon Creek during 12 of the 24 quarterly monitoring events to evaluate potential groundwater/surface water interaction.
- Collect water samples for analysis of VOCs and gage flow from french drain outfall during the past ten monitoring events and sample the culverted intermittent stream outfall during the past nine monitoring events.
- Collect additional bioattenuation parameter samples from five wells during six of the 24 quarterly monitoring events to provide background/baseline data for evaluating the potential for natural attenuation.

The quarterly sampling was conducted to evaluate the extreme seasonal conditions from mid- to late winter, highest spring run-off, mid to late summer and late fall. Timing for the spring sampling was scheduled to try to monitor the peak run-off which varied from season to season.

Results of each quarterly sampling event were provided in separate quarterly reports.

4.2.1 Groundwater Monitoring

For each quarterly monitoring event, groundwater elevations were measured to the nearest 0.01 foot from every accessible well prior to collection of groundwater samples (some wells were buried by snow in the winter quarters and were inaccessible). A summary of groundwater elevation data is provided in Table 5.

Groundwater samples for VOC analyses were collected using passive diffusion bags (PDBs). The PDB technology is based on diffusion of VOCs through a semi-permeable membrane into reagent-grade organic-free water. The PDB is constructed of low-density polyethylene (LDPE), filled with laboratory-grade water free of detectable organic analytes, and suspended at desired depth(s) within the monitoring well for a minimum of two weeks (for vinyl chloride detection). The typical PDB immersion periods ranged from approximately seven weeks to 17 weeks, depending on the frequency of sampling and timing of events. The LDPE allows movement of VOCs into the sampling medium but prevents movement of water across the semi-permeable barrier resulting in an equilibration of the VOCs between the groundwater and the water inside the PDB.

In general, the number of wells sampled was highest in the spring and fall quarters to provide the most data at the extreme ends of the groundwater elevations (highest in spring and lowest in fall). Fewer samples were collected in the summer and winter months. In addition, more samples were collected after well installations as the new wells were included in the monitoring program. The number of primary groundwater samples (not including duplicates) collected over the 24 quarters is presented below:

Quarterly Sampling Summary		
Sampling Event	**Date**	**Sample Summary**
Spring 2011	July 18-20	40 PDBs from 39 wells
Summer 2011	October 17-19	16 PDBs from 15 wells
Fall 2011	December 5-7	42 PDBs from 41 wells
Winter 2012	February 28-29	17 PDBs from 16 wells
Spring 2012	May 14-16	47 PDBs from 45 wells
Summer 2012	September 10-11	20 PDBs from 19 wells
Fall 2012	December 3-5	48 PDBs from 46 wells
Winter 2013	March 19-20	20 PDBs from 19 wells
Spring 2013	May 28-30	47 PDBs from 45 wells
Summer 2013	September 23-24	20 PDBs from 19 wells

Fall 2013	December 16-18	48 PDBs from 46 wells
Winter 2014	March 11-12	21 PDBs from 20 wells
Spring 2014	June 24-26	48 PDBs from 46 wells
Summer 2014	September 24-25	20 PDBs from 19 wells
Fall 2014	December 8-10	51 PDBs from 49 wells
Winter 2015	March 23-24	20 PDBs from 19 wells
Spring 2015	May 18-20	54 PDBs from 52 wells
Summer 2015	September 14-15	29 PDBs from 28 wells
Fall 2015	December 12-15	54 PDBs from 52 wells
Winter 2016	March 28-29	32 PDBs from 31 wells
Spring 2016	May 21-23	60 PDBs from 58 wells
Summer 2016	September 17-18	28 PDBs from 27 wells
Fall 2016	December 4-6	57 PDBs from 55 wells
Winter 2017	Not Conducted	None
Spring 2017	May 22-24	61 PDBs from 59 wells

Two wells (MW-24 and MW-27) have two PDBs in each well installed at different elevations. A summary of VOC data from all quarterly events is provided in Table 2.

4.2.2 Stream Gaging and Sampling

Stream gaging of Saxon Creek was conducted during 12 quarterly events and sampling along Saxon Creek was conducted during 12 of the 24 quarterly events. Stream gaging consisted of stream flow measurement at two to four stations and collection of stream samples for VOC analyses at four stations upstream and downstream of the landfill. Stream monitoring activities completed during this investigation include:

Saxon Creek Gaging and Sampling Summary		
Sampling Event	**Gaging Summary**	**Sample Summary**
Summer 2011	Four stations	Four stations
Fall 2011	No gaging	Four stations
Spring 2012	Four stations	Four stations
Fall 2012	Four stations	Four stations
Spring 2013	Four stations	Four stations
Summer 2013	Four stations	No sampling
Fall 2013	No gaging	Two stations
Spring (April) 2014	Four stations	No sampling
Spring (June) 2014	Four stations	Four stations
Fall 2014	Four stations	Four stations
Spring 2015	Four stations	Four stations

Winter 2016	Four stations	Two locations*
Spring 2016	Four stations	Four stations
Spring 2017	Two stations	Four stations

* = Sampled at locations upstream and downstream from confluence of unnamed intermittent stream with Saxon Creek.

Gaging and sampling locations are shown on Figure 2. Analytical results of the stream sampling are included with the groundwater results in Table 2. Stream sampling was generally conducted during spring and fall quarters, but stream gaging was limited or omitted in two fall quarters due to snow and ice and one spring quarter due to high flows and water levels.

Surface water sampling locations were selected to monitor for potential discharge from the landfill leachate in groundwater to Saxon Creek (Figure 2). Surface water sample S-1 was collected southeast of well MW-8 and north of Fountain Place Road Bridge, above the point where the unnamed intermittent stream diversion culvert installed along the south end of the landfill enters Saxon Creek. Sample S-2 was collected east of the MW-15 well cluster below the point where the unnamed intermittent stream enters Saxon Creek. Sample S-3 was collected north of Power Line Road Bridge west of the MW-20 well cluster, downstream of the point where the unnamed intermittent stream enters Saxon Creek. Sample S-4 was collected approximately 50 feet upstream of the confluence with Trout Creek.

The surface water samples were collected by immersing an unpreserved VOA vial in the creek and decanting the water into a preserved VOA vial. The surface water samples were analyzed for VOCs by EPA Method 524.2. No VOCs were detected in any stream samples collected during this investigation.

The stream flow was gaged at two to four locations along Saxon Creek in general accordance with the USGS *Discharge Measurements at Gaging Stations (Buchanan and Somers, 1969)*. The stream gaging was performed above, adjacent to, and below Meyers Landfill at the following locations shown on Figure 2 and summarized below:

- Approximately 70 feet north of Fountain Place Road Bridge
- East of monitoring wells MW-15A/B
- Approximately 20 feet north of Power Line Road Bridge
- East of monitoring well OW-7

Each of the locations that were gaged were located in a straight section of Saxon Creek that was void of undercut banks, deadfall, significant changes in gradient, or significant amounts of obstructions (e.g. submerged rocks). The stream flow was gaged using a Marsh-McBirney Flo-Mate Model 2000 flow meter. Gaging was done at the same location each sampling event. However, in spring 2017, gaging could not be conducted at the usual measuring stations due to unusually high flows and water levels. Gaging was conducted at the locations of Fountain Place Road Bridge and Power Line Road Bridge, where the stream profile could be safely measured. The stream profile at each gaging station was prepared as follows:

- Stakes were placed on opposite banks at each gaging station. The stakes were not removed after monitoring and were found and re-used for each monitoring event.
- A measuring tape was tied between the stakes and used for horizontal measurement.
- Depth of water was measured at one foot intervals across the stream to represent the stream profile. The stream was shallow enough to wade across for the measurements.

Summary of stream flow calculations is provided in Table 6. Results of the gaging are discussed in Section 5.1.3.

Elevation points were surveyed at gaging stations in September 2011, and the stream elevations were measured at the time of gaging.

4.2.3 French Drain Outfall Sampling

Sampling of the french drain outfall (sample identified as FD-1) began in fall 2014, and sampling of the culvert outlet downstream of the french drain outfall (sample identified as CS-1) began in winter 2015. Sampling of the french drain outfall and culvert outlet was conducted on a quarterly basis through spring 2017.

The french drain outfall drains directly into the to the unnamed intermittent stream diversion culvert at the south end of the landfill. The culvert also receives surface water run-off from the two drop in drains above the culvert and an overflow standpipe in the lined run off pond south of the landfill. Surface water flowing from the culvert outlet typically infiltrates into the ground along the unlined open channel well before the confluence with Saxon Creek. However, in the winter 2016 and spring 2017 quarters, flows were high enough to reach Saxon Creek. During these two quarters, an additional sample (CS-2) was collected from the flowing intermittent stream between the culvert outlet and Saxon Creek. In the winter 2006 quarter, additional samples CS-3 and CS-4 were also collected from Saxon Creek immediately upstream and downstream, respectively, from the confluence with the intermittent stream.

Sampling was conducted in the same manner as for surface water sampling, using a clean, unpreserved vial to collect the water sample then decanting into preserved containers. These samples were analyzed for VOCs by EPA Method 524.2 with the exception of winter 2016, when samples were analyzed by EPA Method 8260B. Results of french drain outfall and culvert outlet samples are shown in Table 2 and discussed in Section 5.1.3.

In addition to sampling, flow from the french drain outfall was gaged by collecting the water in a graduated container and measuring the time taken to collect the measured volume. Results of french drain outfall gaging are discussed in Section 5.1.3.

4.2.4 Sampling for Natural Attenuation Parameters

Additional chemical analyses were conducted during six of 24 quarterly monitoring events to provide background/baseline data for evaluating the potential for natural attenuation. Five wells were selected for monitoring natural attenuation parameters to represent background and VOC impacted groundwater conditions in the UGZ and MGZ. Wells MW-30, T-6, and MW-20A were sampled to assess monitored natural attenuation (MNA) parameters in the UGZ and wells D-1 and MW-18A were sampled to assess MNA parameters in the MGZ. MW-30 was selected to represent conditions upgradient of the landfill.

Each well was purged using a low-flow purging technique at a steady flow rate generally in the range of approximately 175 to 500 milliliters per minute. During purging, field parameters (temperature, pH, specific conductance, dissolved oxygen (DO), oxidation-reduction potential (ORP), and turbidity) were measured using a flow-through cell. Polyethylene tubing dedicated to each specific well was used with a decontaminated pump. The field parameters of the stabilized well are provided along with the MNA results on Table 7.

Following well purging, groundwater samples were collected for laboratory analyses. Samples were collected directly from the pump and tubing after disconnecting the tubing from the flow-through cell. The pump was decontaminated between wells, and dedicated discharge tubing was used in each well.

The groundwater samples were submitted for analyses using the following analytical methods:

- Metals (ferric iron, ferrous iron, and divalent manganese) by Method 7199 modified.
- Inorganic Ions (chloride, nitrate, nitrite, and sulfate) by Method 9056.
- Dissolved Gases (methane, ethane, ethane, carbon dioxide, oxygen and nitrogen) by Method AM20GAX.
- Total Organic Carbon by Method 5310-2011.

Results of the MNA sampling from the most recent phase of investigation are included in Table 7. Table 7 also provides field parameters recorded during the more recent sampling and results for metals and other MNA parameters analyzed during prior investigations.

4.3 REGIONAL WELL SEARCH

As part of the previous phase of investigation, WESTON conducted a regional well search to identify groundwater wells in the vicinity of the Site in order to qualitatively evaluate potential receptors for groundwater contamination. Well records were requested from the California Department of Water Resources (DWR) and El Dorado County.

The well search identified one active well within one-mile radius of the Site, the Elks Club No. 2 well located approximately 0.75 mile west of the Site (Figure 1). This well is an active public supply well operated by STPUD. One other well, the Herms Brothers well, has not been located. Based on the DWR well designation, this well would be located north of the landfill. However, it is likely the location is mislabeled and the well probably is (or was) located at the Herms Brothers asphalt plant which was located southeast of the landfill near monitoring wells M-8/M-8A.

A buried water supply well located near the northwest side of the landfill was uncovered and abandoned during the landfill capping activities. The well was not identified in DWR or county records. It is likely this well is the one identified in a 1961 drawing titled “X - Section Lines, Lake Valley Garbage Site” provided by the USFS. No other information regarding the construction or use of the well has been found. Landfill facility drawings from 1961 illustrates that the well was located west of the limits of landfill waste near the site of several former structures associated with the County's landfill operations

5. INVESTIGATION RESULTS

The following section presents results of the recent investigations conducted by WESTON from July 2011 through May 2017. The scope of work completed for this phase of investigation is described in Section 4.

Results of each quarterly monitoring event have been summarized in quarterly reports. Results generated during the recent investigation build on those from prior investigations and are used develop the site hydrogeologic models discussed in the following section.

5.1 QUARTERLY MONITORING

Quarterly groundwater monitoring was conducted for 24 quarters from spring 2011 through spring 2017. The objectives of the monitoring were to:

- Evaluate plume stability and potential migration based on VOC concentrations in monitor wells.
- Evaluate seasonal effects on flow conditions and plume migration.
- Evaluate potential groundwater/waste interaction.
- Provide baseline data to evaluate the effect landfill capping may be having on the plume.
- Evaluate groundwater conditions in newly installed wells.
- Evaluate potential groundwater/surface water interaction in conjunction with surface water gaging and testing.
- Provide additional background/baseline data for evaluating the potential for natural attenuation.

The quarterly sampling was conducted to evaluate the extreme seasonal conditions from mid- to late winter, highest spring run-off, mid to late summer and late fall. Timing for the spring sampling was scheduled to try to monitor the peak run-off which varied from season to season.

The PGZ, UGZ, MGZ, and LGZ groundwater flow directions are illustrated in Figures 6 through 13 for the 2016 and spring 2017 sampling events. Figure 14 illustrates both the UGZ and MGZ overlapping plume boundaries for fall 2016. Figures 15 and 16 illustrate vinyl chloride concentrations in the PGZ. Figures 17 through 20 show the fall 2016 and spring 2017 UGZ and MGZ plumes that include interpretation of hydrogeologic data generated during prior and recent data collection activities, particularly regarding the vertical limits of the UGZ and MGZ. The interpretation included comparison of screen elevation data across the monitoring well network to groundwater elevation data, and using lithologic data and groundwater contaminant data to determine which wells are screened in similar horizons and therefore likely represent a similar groundwater piezometric surface. This interpretation is particularly difficult when attempting to plot groundwater gradients in the MGZ as discussed below.

5.1.1 Groundwater Elevation Monitoring

Depth to groundwater was collected from all accessible wells each quarter. These data were used in conjunction with survey data to calculate groundwater elevations at each well in order to prepare gradient plots. A summary of groundwater elevation data is provided in Table 5. Groundwater elevation contour maps for the PGZ, UGZ, MGZ and LGZ from the fall 2016 and spring 2017 sampling events are provided on Figures 6 through 13. Historical hydrographs for selected UGZ, MGZ and LGZ wells running roughly down the axis of the plume (south to north) are provided on Figure 21.

The depth to water in Site wells range from just below top of casing in wells near Saxon Creek (0.60 feet in MW-20A in May 2017), to over 200 feet in wells on the ridge along Pioneer Trail (as deep as 204.39 feet in MW-27 in September 2015). Hydrographs of selected wells from the UGZ, MGZ and LGZ show seasonal and year to year variation in groundwater elevations in all three zones (Figure 21). As expected, groundwater elevations are highest in the late spring following snowmelt and lowest in late fall/winter prior to spring thaw.

Vertical Gradients

A generally low to moderate downward vertical gradient (up to 0.559 foot/foot with a vertical elevation difference of more than 30 feet in well pair MW-31A and MW-31B) is noted between the UGZ and MGZ in most clustered well pairs. One to five well pairs may exhibit slight upward gradients in a given quarter ranging to 0.097 foot/foot. Well pairs in which upward gradients have been observed include T-6/D-1, T-12/D-3, T-14/MW-21A, MW-15A/MW-15B, and

MW-19A/MW-19B. Of these five well clusters, three are located near Saxon Creek. The close relationship between hydraulic heads in the UGZ and MGZ can be seen in water level hydrographs depicted in Figures 21 through 24.

Vertical groundwater elevation differences in UGZ/MGZ wells are generally in the range of one to five feet except for OW-3/MW-18A with an elevation difference around 13 feet, MW-31A/MW-31B with an elevation difference of more than 30 feet, and MW-19A/MW-19B in which the hydraulic head in the MGZ is typically slightly higher than in the UGZ.

A stronger downward vertical gradient up to 0.821 foot/foot is noted between the MGZ and LGZ cluster wells with differences in groundwater elevations of over 20 feet. Hydrographs for 12 UGZ, MGZ, and LGZ wells are shown on Figure 21. Hydrographs for three UGZ/MGZ/LGZ well cluster locations are provided on Figures 22 through 24. These plots (for well clusters T-6/D-1/D-4, T-11/D-2/D-5 and T-12/D-3/D-6) further illustrate the general vertical differences of around one to five feet between the UGZ and MGZ wells, and much larger vertical differences of around 30 to 40 feet to the LGZ. The hydrographs further show fairly similar patterns in seasonal response between the UGZ and MGZ indicating these two zones are interconnected or at least respond similarly to hydraulic changes.

PGZ Groundwater Elevations

A zone of groundwater is perched on a shallow aquitard along the west and southwest side of the landfill, and up the intermittent stream valley south of the landfill. The presence of perched groundwater on the west and southwest side of the landfill is inferred from historic water measurements recorded from the abandoned piezometers P-1 and P-2, historic data from the abandoned well M-7, and the presence of a spring emerging from the hillside near the southwest corner of the landfill (Figure 2). Perched groundwater and the underlying aquitard were observed during drilling of wells T-9, MW-16, MW-29, MW-30, and MW-31A. Figure 25 depicts interpreted elevations of the underlying aquitard based on observations from these borings and lines of a series of six cross sections through the PGZ presented in Figures 26 and 27.

Multiple cemented layers were observed in the cut-bank with seepage from fractures along Garbage Dump Road on the west side of the landfill during landfill cap construction. Perched water likely occurs in several joints and layers through the upper aquitard zone along the west and southwest side of the landfill as well as in overlying sediments. Figures 8 and 9 show the approximate extent of the perched zone based on available explorations. The french drain system installed during the recent cap construction is collecting and diverting perched water from the west and southwest of the landfill to a collection trench along the southern end of the landfill, and also appears to intercept perched groundwater periodically present between the french drain and the landfill. Periodic detection of vinyl chloride in piezometers PP-4 and PP-8, on the east side of the french drain, and in PP-9 west of the french drain indicates that perched groundwater periodically flows across the french drain in a westward flow direction, and the French drain may not efficiently be capturing the perched water near that location.

Perched groundwater is also present in the intermittent stream valley south of the landfill. Groundwater in well MW-16 south of the landfill exhibits dramatic seasonal water level fluctuations which have historically ranged from approximately 7 to more than 25 feet bgs and the PGZ and UGZ appear to be merged in this area. Remedial action work at the landfill encountered landfill debris near well MW-16 immediately above the perched groundwater. The extent of the perched groundwater north of well MW-16 (i.e., extending beneath the southern end of the landfill) has not been determined. The presence of groundwater in PTEM boring B-20 (Figure 3) as well as the relationship between perched water and the landfill depicted on the cross section in Figure 28 suggest some limited interaction between the perched zone and waste may be occurring in the southern-most part of the landfill. However, perched groundwater was not encountered during the drilling of landfill well LF-1 located approximately 250 north of MW-16, or in other landfill wells.

Groundwater elevations in the PGZ vary seasonally and with location relative to the french drain. West of the french drain, seasonal variation is generally less than one foot (although variation of up to seven feet was observed during the historic high groundwater event in spring 2017). East of the french drain, between the french drain and the landfill, seasonal differences may be as much as four feet or more, with some piezometers becoming dry in the summer to late fall, and water levels rebounding sharply following the spring snow melt.

Groundwater flow in the PGZ differs with location. In the area immediately west of the french drain, gradient is toward the east-southeast (toward the french drain) at steep gradients of 0.03 to 0.10 foot/foot. West of Garbage Dump Road, gradients are toward the west, based on following the surface of the underlying westward-dipping aquitard. Between the french drain and the landfill, gradients generally have a significant component toward the french drain, and an overall decrease in groundwater elevation from north to south. Groundwater gradients in the PGZ in the vicinity of the french drain in fall 2016 are shown in Figure 6, and gradients in spring 2017 are shown in Figure 7.

A series of six cross sections through the PGZ and landfill wells are presented in Figures 26 and 27 showing the relationships between the PGZ, UGZ, french drain, and landfill in this area. The sections depict groundwater levels in the PGZ during dry (December 2015) and wet (March 2016) seasons. During dry season conditions, flow near the french drain is toward the east, where it is intercepted by the french drain. During wet season conditions, groundwater accumulates in the area between the french drain and the landfill and flows west toward the french drain. Under these conditions, PGZ groundwater elevations along this area that are higher than those found in piezometers on the opposite side of the french drain suggest flow across the french drain may occur. The periodic presence of groundwater observed in piezometers located between the french drain and landfill indicates that some perched water is flowing across the french drain and/or that localized recharge is occurring from percolation of surface water between the french drain and the landfill.

UGZ Groundwater Elevations

In general, water elevations in the UGZ generally vary seasonally by as much as two to five feet between the highest levels during the spring snowmelt and the lowest levels in late fall/early winter at the end of the dry season, but have historically ranged to as much as 13 feet year to year in individual wells.

Overall, groundwater elevations in the UGZ have varied by as much as around 14 feet in T-1; from 6,330.90 feet MSL in May 2017 to 6,316.71 feet MSL in December 2015. The seasonal groundwater fluctuations are greater along the axis of the intermittent stream valley, and less pronounced in wells located on the ridges where seasonal variation is generally only around one to two feet. However, groundwater flow direction and gradient remain fairly consistent seasonally.

The groundwater in the UGZ flows eastward away from the ridge along Pioneer Trail at a steep gradient of approximately 0.03 foot/foot. The gradient becomes flatter toward Saxon Creek. Groundwater flow is northeastward beneath the substation and along the axis of the valley at a gradient of approximately 0.004 to 0.006 foot/foot (Figures 8 and 9). The groundwater beneath the landfill may be locally influenced by silty sediments beneath a ridge separating the intermittent stream valley from Saxon Creek, directing the groundwater flow towards the north.

MGZ Groundwater Elevations

Interpreting groundwater flow conditions in the MGZ is difficult. Groundwater in the MGZ is encountered in multiple sand layers separated by fine-grained clayey silt units. The sand layers were deposited by fluvial process where paleo-streams (Trout and Saxon Creek) once drained from the mountains and then flowed westward to northwestward towards the axis of the valley. During glacial periods of high lake levels (up to 600 feet above present Lake Tahoe levels), stream deposits formed deltas on the paleo-lake bottom near the mountain slopes. The course of the streams changed during the geologic history causing overlapping meanders with fine grained sediments deposited below and above the sand layers during various glacial lake levels. Additionally, the fluvial deposits in the MGZ are recharged with different hydraulic heads from the UGZ where the two water bearing zones interconnect. The different hydraulic heads may also be affected by local faulting creating discontinuities between fluvial deposits at similar elevations. Determining which wells to use to represent general groundwater flow conditions and develop groundwater contour maps for the MGZ utilizes a combination of water elevation data as well as VOC plume configuration. The wells used for preparing the groundwater contours to represent general flow conditions in the MGZ were selected by eliminating MGZ wells that would produce groundwater elevation contours in directions not trending from the landfill toward the furthest contaminated monitoring well (MW-24).

Groundwater flow in the MGZ is toward the north at a gradient of approximately 0.004 to 0.006 foot/foot to the north of the landfill (Figures 10 and 11). In general, water elevations in the MGZ vary by as much as two to six feet between the highest levels during the spring snowmelt and the lowest levels in late fall/early winter at the end of the dry season, similar to seasonal elevation variations exhibited in the UGZ (Figure 21). However, groundwater flow direction and gradient remains fairly consistent seasonally. The similarity between the water elevation variation in the UGZ and MGZ suggest that these zones, although distinct with respect to contaminant distribution, are interconnected or at least respond similarly to seasonal hydraulic changes. Overall, groundwater elevations in the MGZ have varied by as much as around 13 feet in D-1; from 6,317.37 feet MSL in December 2015 to 6,330.34 feet MSL in May 2017 (Figure 21).

Apparent groundwater flow and hydraulic heads are not always consistent with VOC plume data in the MGZ. For instance, MW-17A and MW-17B are both considered to be screened in the MGZ based on constructed depths and lithology, but only MW-17A shows the presence of vinyl chloride (although PCE and *cis*-1,2-DCE are detected in MW-17B). Further, the vertical distance between the bottom of the screened zone of the shallower well (MW-17A) is only 12 feet above the top of the screen zone of the deeper well (MW-17B), but water elevations in these wells are roughly seven to nine feet apart.

The disparity between groundwater elevation data (and expected flow direction and local gradient) and the VOC plume configuration in the MGZ are further exhibited by comparing groundwater elevations in the impacted MGZ wells to groundwater elevations across the area. The VOC data for the MGZ shown on Figures 19 and 20 show the plume migrating in a northerly direction from the landfill. However, review of the groundwater elevation data from impacted MGZ wells shown on Figures 10 and 11 would suggest a fairly steep northeasterly to easterly gradient from MW-17A to MW-20B with a trough at MW-18A, and an upslope plume migration between MW-18A and MW-24. This gradient does not match the VOC plume data indicating that the migration of VOCs is largely controlled by preferential pathways (buried stream channels) with differing hydraulic heads. As a result, the groundwater elevation contour maps presented for the MGZ are based on data from wells selected that exhibit a flow direction consistent with concentrations at intermediate wells as well as the furthest downgradient extent of the plume (MW-24).

LGZ Groundwater Elevations

Groundwater in the LGZ (similar to the MGZ as described above) is also encountered within sand layers separated by fine-grained clayey silt units. The hydraulic heads in LGZ wells are less pronounced than those found in the MGZ; however the elevation variance in wells MW-19C and MW-20C are enough to affect the very shallow gradient and are not used for contouring. Groundwater flow in the LGZ flows towards the west-northwest at a very shallow gradient of approximately 0.0004 to 0.0009 foot/foot (Figures 12 and 13). The westerly flow direction may reflect regional groundwater flow in the basal sediments above bedrock and away from the Carson Range. It is possible that the westerly flow direction in the LGZ may be influenced by pumping from Elks Club Well No. 2 located approximately 0.75 mile west of the site, but data has not been provided to evaluate this potential interaction. Water elevations in the LGZ also show subdued response to seasonal variation, but typically do not vary more than around one-half to two feet (Figure 21). Excluding the potentially anomalous data point from D-6 in August 1999, overall groundwater elevations in the LGZ have varied by as much as around 9 feet in D-4; from 6,284.69 feet MSL in September 2005 to 6,293.90 feet MSL in May 2017.

Groundwater/Waste Interaction

Three UGZ wells (LF-1, LF-2, and LF-3) were installed along the approximate axis of the buried intermittent stream channel beneath the landfill. The wells were screened beneath the landfill waste to evaluate potential groundwater/waste interaction by providing water level conditions and contaminant data before and after capping of the landfill. During drilling and well installation, the base of the waste and static water levels were identified at:

- LF-1 – base of waste (36 feet bgs), static water level (54 feet bgs)
- LF-2 - base of waste (43 feet bgs), static water level (62 feet bgs)
- LF-3 - base of waste (44 feet bgs), static water level (56 feet bgs).

Quarterly groundwater measurements were collected from the wells from October 2008 to present. The relationship of groundwater elevations to the base of the waste is depicted on Figure 28. This figure shows groundwater elevations beneath the majority of the landfill wells are approximately 2 to 15 feet below the base of the landfill waste in general and have not risen to the base of the landfill waste since these wells were installed in 2008. Water elevations from well T-2 located at the downgradient toe of the landfill rose to within approximately 2.5 feet of the base of the waste during the highest measured groundwater elevation in May 1998. Groundwater elevation in LF-1 was less than three feet below the lowest waste elevation (which occurs at the southern end of the landfill) in July 2011 and was above the lowest waste elevation in May 2017. Groundwater elevation in LF-2 matched the lowest waste elevation in May 2017. For the most part, the piezometric surface is typically around 10 to 15 feet below the waste elevation as shown on Figure 28. Based on presence of high perched groundwater levels to the south of the landfill as observed at MW-16, there is a possibility of perched water interaction with landfill waste. Further, projection of UGZ groundwater elevations between MW-30 and LF-1 suggests potential for interaction between water in the UGZ and waste near the southern end of the landfill during high groundwater events (Figure 28). However, the UGZ was encountered in well MW-30 at approximately 40 feet bgs and is confined beneath the PGZ aquitard at this location. Figure 28 illustrates potentiometric groundwater levels and soil (or waste) may not be saturated below the potentiometric surface in the southern portion of the landfill. In addition, due to the steep eastward gradient in the UGZ west of the landfill, it is possible that groundwater elevations beneath the western side of the landfill have approached the base of the landfill waste at times, as suggested by the cross section diagrams in Figure 20. The UGZ encountered in wells PU-1 and PU-2 is confined by the PGZ aquitard and potentiometric groundwater levels are often above the UGZ water bearing zone. However, the eastern extent of the PGZ aquitard beneath the landfill is unknown.

5.1.2 Groundwater Analytical Results

Historical VOC results from groundwater monitoring well samples are summarized in Table 2. The data has been was utilized to construct vinyl chloride concentration maps for the UGZ and MGZ for each quarter (Since VOCs are not detected in the LGZ, concentration maps were not prepared for the LGZ). Concentration maps for the PGZ have been included in the last six quarterly monitoring reports. Figures 15 through 20 show vinyl chloride concentrations maps for the fall 2016 and spring 2017 sampling events for the PGZ, the UGZ and the MGZ, respectively. Figure 14 shows the vinyl chloride plumes for the combined UGZ and MGZ, and also plots the wells used to construct cross-section transects A-A’ and B-B’ and the lines of the cross-sections (Figures 29 and 30). Discussion of the historical plume distribution is provided below.

The primary constituent identified in the groundwater is vinyl chloride, a common solvent degradation product. Vinyl chloride has been detected at a maximum concentration of 100 µg/L in the UGZ (well T-6 in December 1997), and at a maximum concentration of 69 µg/L in the MGZ (well MW-17A in October 2008).

Several other VOCs have been detected since monitoring began, most at trace concentrations (Table 2). The historical maximum concentration of other commonly detected VOCs include:

- 1,2-Dichlorobenzene (1,2-DCB) – 1.82 µg/L in T-1 (November 2000)
- 1,4-Dichlorobenzene (1,4-DCB) – 12 µg/L in LF-1 (December 2011).
- Benzene - 3.1 µg/L in D-2 (May 2008).
- Chlorobenzene - 27 µg/L in M-4 (May 2006).
- *cis*-1,2-Dichloroethene (*cis*-1,2-DCE) - 25 µg/L in T-9 (December 2015).
- Dichlorodifluoromethane (CFC-12) - 22.0 µg/L in M-4 (October 1994).
- Dichloromethane (methylene chloride) - 5.41 µg/L in T-6 (June 1998).
- Tetrachloroethene (PCE) - 12 µg/L in M-10 (November 2000).
- Trichloroethene (TCE) – 1.8 µg/L in MW-21A (May 2015).
- *trans*-1,2-Dichloroethene (*trans*-1,2-DCE) - 34 µg/L in M-5 (March 1992).
- Diethyl ether – 78 in T-6 (December 1997)

Vinyl Chloride Distribution

In the PGZ, vinyl chloride has been detected in piezometers PP-4 and PP-8 on the landfill side of the french drain, and in PP-9 west of the french drain. Vinyl chloride concentrations in perched groundwater range from 0.71 to 2.2 µg/L. The source of the vinyl chloride in the PGZ is either from contaminated soil above the perched aquitard or from rapid snow melt or precipitation resulting in the infiltration of water adjacent to the landfill and seepage beneath the landfill cap where groundwater contacts the western margins of landfill waste. Vinyl chloride was not detected in PP-9 during the September 2016 and December 2016 sampling. Piezometers PP-4 and PP-8 have had sufficient water to collect samples only sporadically, and have not been sampled since May 2017. Occurrence of vinyl chloride in PP-9 further indicates that perched groundwater flows across the french drain periodically. Vinyl chloride results for December 2016 and May 2017 are shown on Figures 15 and 16. Plots of historical vinyl chloride concentrations show both seasonal and long-term trends. Figure 31 show vinyl chloride concentrations versus time, and also groundwater elevations for PGZ piezometers PP-4, PP-8, and PP-9. Groundwater levels from these piezometers as well as PP-3 are also illustrated to provide historical elevations.

The vinyl chloride plume in the UGZ extends northward at the landfill, but curves toward the northeast between monitoring wells MW-17A, T-11, and T-13, following the drainage valley leading toward Saxon Creek (Figures 17 and 18). This diversion of the plume direction in the UGZ is likely a result of change from the steep eastward groundwater gradient coming off the ridge along Pioneer Trail to follow the unnamed intermittent stream valley leading to Saxon Creek. The UGZ plume extends a short distance in the upgradient direction from the landfill as observed by detections in wells PU-1 and PU-2. The cause of this may be that westward dipping strata including the overlying aquitard in this area may be controlling migration toward the west. The vertical and lateral extent of vinyl chloride in the UGZ is well defined; VOCs have not been detected in sentinel well MW-19A which bounds the down gradient extent of the VOC plume along Saxon Creek.

The vinyl chloride plume in the MGZ extends northward from the landfill and does not appear to be influenced by the steep eastward groundwater gradient observed in the UGZ coming off the Pioneer Trail ridge (Figures 19 and 20). The vertical and lateral extent of vinyl chloride in the MGZ is fairly well defined. In at least one instance, the upper surface of the MGZ appears fairly sharp as evidenced in well MW-24 where samples from the upper PDB hung at 144.5 feet contained no detectable vinyl chloride, but samples from the lower PDB hung at 159.5 feet consistently show detectable vinyl chloride (although concentrations have declined from approximately 20 µg/L to 8 µg/L). The thickness of the MGZ is approximated at approximately 20 to 60 feet at cluster group OW-3/MW-18A/MW-18B. Samples from UGZ well OW-3 screened from 40 to 60 feet bgs consistently contained no detectable vinyl chloride; samples from MW-18A screened from 80 to 100 feet bgs show detectable vinyl chloride (although concentrations have declined from approximately 40 to 60 µg/L to 25 to 30 µg/L; and samples from LGZ well MW-18B screened from 122 to 142 feet bgs consistently contain no detectable vinyl chloride.

The LGZ does not appear to be impacted by vinyl chloride or other VOCs.

Vinyl Chloride Cross-Sectional Distribution

The groundwater vinyl chloride data has been correlated to the hydrogeologic model to prepare cross-sections across the plume for spring 2017 (Figure 29 and 30). Figure 29 shows the vinyl chloride/groundwater zone relationship along a roughly south-to-north longitudinal profile down the length of the plume (A-A’). Figure 30 shows the relationship along a roughly west-to-east transect across the leading edge of the plume (B-B’). The cross-section locations are shown on Figure 14. The wells used in the transects were those which provided the best chemical and lithologic data sets to evaluate the interaction between the multiple groundwater zones, utilizing several of the cluster well groups. A time sequence of three dimensional (3D) two dimensional (2D) conceptual plume models going back to 2007 is included in Appendix B.

Cross-section A-A’ shows the interconnection of the UGZ and MGZ plume immediately north (downgradient) of the landfill around the T-3 and T-6 well clusters. The plume descends into the MGZ around the T-11 cluster, and is confined entirely within the MGZ by the time it reaches the OW-3/MW-18A/18B well cluster. The plume in the MGZ extends northward to MW-24, but was not identified further downgradient at MW-26. The lithologic log for MW-26 shows an approximately 115-foot thick sequence of poorly yielding laminated silt from approximately 6,194 to 6,309 feet MSL, with no significant sand layers. This indicates that the MGZ sand units impacted by VOCs present at well MW-24 pinch-out or veer off before reaching MW-26.

Figure 30 shows the control on the lateral edges of the plume along the leading edge. This figure shows that the plume in the MGZ is confined between MW-25 to the east and MW-27 to the west. The lithologic log for MW-27 also shows an approximately 35 feet thick sand unit near the base of the MGZ and top of the LGZ from around 6,236 to 6,270 feet MSL. Sample results from MW-27 showed an initial trace concentration of PCE (0.9 µg/L in June 2010), but have not showed the presence of any VOCs since that initial result. This well is an important sample point for monitoring the lateral edge of the plume particularly as this location may indicate a transition between the MGZ and LGZ. The vinyl chloride plume has not impacted well MW-25 where the MGZ may not be present.

Historical Vinyl Chloride Results

Plots of historical vinyl chloride concentrations show both seasonal and long-term trends. Figure 31 shows vinyl chloride concentrations for the PGZ wells versus groundwater elevations (PP-4, PP-8, and PP-9) and vinyl chloride concentrations for the french drain outfall. Figures 32 through 36 show vinyl chloride concentrations for UGZ wells aligned roughly down the axis of the UGZ plume (T-1, T-6, T-12, T-14, and MW-14/MW-20A). These plots indicate vinyl chloride concentrations are affected by seasonal groundwater variations, although the effects are not entirely consistent. Vinyl chloride concentrations appear to show an inverse relationship to groundwater elevation as best exhibited in wells T-6 and T-12 (Figures 33 and 34). Highest groundwater elevations commonly correspond with lowest vinyl chloride concentrations probably due to dilution caused by a large influx of fresh water into the groundwater system during spring snowmelt.

Figures 37 through 41 show vinyl chloride concentrations versus groundwater elevations for MGZ wells aligned roughly down the axis of the MGZ plume (D-1, D-2, MW-17A, MW-18A, and MW-24). Plots of the MGZ wells show a similar inverse relation between vinyl chloride concentrations and groundwater elevations as best exhibited in wells D-1 and D-2 (Figures 37 and 38).

The seasonal variations overlay apparent longer term concentration trends. Figure 41 shows historical vinyl chloride concentrations for UGZ wells T-1, T-6, T-12, T-14, and MW-14/MW-20A, and Figure 42 shows historical vinyl chloride concentrations for MGZ wells D-1, D-2, MW-17A, MW-18A, and MW-24. These wells are situated roughly along the axis of the UGZ and MGZ, and are the same wells used in the plots on Figures 321 through 41.

Second order polynomial regressions were performed for visual analysis of trends in UGZ and MGZ wells (Figures 42 and 43). The concentration plots and regression analyses are shown spatially on Figure 44 (UGZ) and 45 (MGZ). Newer well MW-29 is included on Figure 45 since it appears to exhibit a possible upward concentration trend, although Mann-Kendall trend analysis (see below) suggests there is not a statistically significant trend in this well at a 95% confidence level.

A series of wet season (spring) and dry season (fall) two dimensional and three dimensional UGZ and MGZ plume and groundwater contours figures (Appendix B) illustrate the migration of the plumes assuming homogenous aquifer conditions. The figures plan views and cross-sections were extracted from a 3D conceptual model. The plan views represent the maximum vinyl chloride concentrations within the stratigraphic unit in question. The cross-sections depict the model exactly along the transects whereas the data are projected onto the sections from up to 200-ft away. Although the 3D interpolations honor all data, the 2D depictions may not always appear to because of the projection and extraction extrapolation issues associated with presenting a 3D model in 2D. For example, the 3D conceptual model uses the concentrations from the UGZ well MW-20A and projects the UGZ plume deeper than what is observed in the MGZ well MW-20B and implies the MGZ is impacted at this location, which it is not. Additionally, plume configurations change vary based on the available data at the time. As more wells were are added over the duration of the monitoring or were not sampled, the modeled plume and groundwater contours may expand or contract in a given direction. This is apparent in the area of the landfill for the period before the UGZ wells (LF-1, LF-2 and LF-3) were installed and for events when UGZ landfill wells were not sampled, including during the spring 2011, fall 2011, and fall 2015 sampling events where UGZ landfill wells were not sampled.

The model depicts the UGZ plume fairly well as the UGZ aquifer is more homogenous as compared to the MGZ aquifer. The MGZ aquifer is comprised of preferential pathways (buried stream channels) that cannot be mapped across the Site with certainty; however, the homogenous model illustrates the extents and concentrations as those found in MGZ wells with the exception of MGZ area near well MW-20B as described above.

The MGZ aquifer is comprised of preferential pathways (buried stream channels) that cannot be mapped across the Site with certainty; however, the homogenous model illustrates the extents and concentrations as those found in MGZ wells.

The effect of installation of the impermeable landfill cap is not evident in UGZ vinyl chloride concentrations beneath and downgradient of the landfill. Since cap installation, most wells exhibit more stable concentration ranges, and most show decreasing trends, however, these effects are not clearly tied to installation of the cap. Cap installation decreases the leaching of contaminants from the primary source (landfill mass) to groundwater by inhibiting infiltration of meteoric water. Contaminated soil beneath the landfill and below the groundwater table will continue to be a secondary source of groundwater contamination for years to come. The impact of the landfill cap on the plume will require additional time before effects can be documented.

Visual review of the vinyl chloride concentration trends for the UGZ indicate the following:

- Concentrations at T-1 (nearest the landfill) were in the range of approximately 40 to 42 µg/L at initial monitoring in 1997 to around 2000, and dropped below 20 µg/L in 2009 where they stabilized, with one peak above 20 µg/L in December 2015. Concentrations dropped below detection limits in samples from May 2015 and May 2017.

- Approximately 350 feet downgradient from the landfill at T-6, concentrations generally decreased from approximately 60 µg/L at initial monitoring in 1997 to approximately 30 µg/L by the time of landfill cap liner installation in July 2011. Since then, concentrations have been fairly stable, mainly in the 30-40 µg/L range, and appear to continue a slight decreasing trend.

- Approximately 950 feet downgradient from the landfill at well T-12, concentrations dropped from approximately 90 µg/L at initial monitoring in 1997, to approximately 30 µg/L in 2006 with steadily decreasing concentrations especially since 2011, and are more recently in the 10-20 µg/L range.

- Approximately 1,200 feet downgradient from the landfill at well T-14, concentrations dropped from approximately 25 µg/L at initial monitoring in 1999, to less than 20 µg/L in 2008, then rose above 30 µg/L in 2010-2012. Since installation of the landfill cap liner in July 2011, concentrations have decreased considerably and have recently been below 10 µg/L until a slight increase in May 2017.

- Approximately 1,700 feet downgradient from the landfill at former well MW-20A (former MW-14), concentrations show an increase from initial monitoring in 2005, then a consistent slight decreasing trend, particularly since 2013. Recent concentrations have generally been below 5 µg/L.

Visually, vinyl chloride concentration trends suggest that concentrations in the UGZ are stable or declining. Review of the trends and plume configuration for the UGZ indicates there is a hot-spot that appears to have migrated downgradient along the plume axis, with concentrations peaks at T-6 approximately 350 feet downgradient of the landfill in March 2008, then at T-12 approximately 950 feet downgradient of the landfill in September 2010, and then at T-14 approximately 1,200 feet downgradient of the landfill in July 2011. This suggests a slug of vinyl chloride has migrated beyond the first set of monitoring wells located along the toe of the landfill, although vinyl chloride is still impacting groundwater from contaminated soil beneath the landfill as indicated by the continuing presence in wells T-1 and T-6 (although concentrations have decreased considerably from initial monitoring in 1997). Using the peak concentration to evaluate plume transport velocity is difficult since the monitoring well data only extends back to 1997 and likely missed the peak plume front at the toe of the landfill. In addition, the peaks between the wells are not easily correlated. Still, assuming the peak identified in T-1 at approximately 41 µg/L in 1999 is the same peak identified in T-14 at approximately 27 µg/L in 2008, the groundwater flow velocity is calculated at around 130 feet/year (0.35 foot/day). Utilizing the estimated progress of the concentration peaks cited above, a plume seepage velocity of approximately 265 feet per year (0.73 feet per day) was calculated. This is within the range of groundwater flow velocities calculated by Geomatrix. Using the hydraulic conductivity of 20 to 60 feet/day calculated by Geomatrix from the PTEM pump test assuming a 100 foot aquifer thickness (see Section 2.3.1) (Geomatrix, 2004), a hydraulic gradient of approximately 0.005 foot/foot between T-1 and T-14, and an assumed porosity of 30 percent for the UGZ, the groundwater flow velocity is calculated at around 120 to 360 feet/year (0.33 to 1.0 foot/day). Using the hydraulic conductivity of 40 to 90 feet per day and utilizing PTEM's assumed aquifer thickness of 65 feet and the assumptions above, groundwater flow velocity is calculated to range from approximately 240 feet to 550 feet/year (0.67 to 1.5 feet/day). The seepage velocity estimated above is within both of these ranges.

Vinyl chloride concentration trends in the MGZ suggest that concentrations are stable or declining in most wells. The overall plume is stable, although results from some recent quarters suggest the plume may be expanding laterally to a slight degree in the areas of wells MW-21A and MW-29. In the area of MW-29, the plume appears to be expanding slightly to the west (cross-gradient), possibly because migration may be in part controlled by westward dipping strata in that area, similar to the slight expansion toward the west observed in the UGZ at PU-2. The reason for the minor lateral expansion toward the east in the area of MW-21A is unclear, but may be due to other favorable hydraulic conditions in that location. Review of vinyl chloride trends and plume configuration for the MGZ indicates that a hot-spot extends from the area of D-1 to the vicinity of MW-18A. Since many of the MGZ wells were installed in 2008, insufficient data is available to evaluate longer-term trends in the MGZ or groundwater flow velocity. Review of the vinyl chloride trend analyses for six MGZ wells reveals the following:

- Concentrations at well D-1 (approximately 250 feet downgradient from the landfill) were fairly stable at approximately 30 µg/L at initial monitoring in 1998, exhibited a general decreasing trend with some variability to around 2011, and appear to continue a slow decline through 2016, with recent concentrations around 20 µg/L.

- Concentrations in well MW-29 (approximately 450 feet northwest of and somewhat cross gradient from the landfill) appear to be slowly increasing, ranging from just above 10 µg/L after monitoring began in 2010 to nearly 20 µg/L through 2016 (although decreasing to 5.2 µg/L in spring 2017). Minor expansion of the MGZ plume to the west around MW-29 may be similar to the expansion of the UGZ to the west at PU-2 in that westward-dipping strata may in part be controlling migration of the plume in this area.

- Concentrations in well MW-17A (approximately 650 feet downgradient from the landfill) have shown a steady decline from 70 µg/L when initially sampled in 2008 to approximately 25 µg/L recently. Although the trend appears to be leveling off from its initial sharp decline, concentrations still appear to be decreasing overall.

- Concentrations in well D-2 (approximately 800 feet downgradient from the landfill) have risen steadily from approximately 5 µg/L at initial monitoring in 1998 to between 20 to 30 µg/L through spring 2017. It appears that concentrations may be beginning to stabilize in D-2.

- Concentrations in MW-18A (approximately 1,350 feet downgradient of the landfill) are also showing steady decrease, with a more linear trend than for MW-17A. Concentrations in MW-18A have shown a decrease from approximately 40 to 50 µg/L in initial monitoring to approximately 25 to 35 µg/L recently.

- Concentrations in the lower PDB sample from well MW-24 (located approximately 1,750 feet downgradient of the landfill) have shown a slowly decreasing trend since monitoring began in 2009. Concentrations have decreased from approximately 20 µg/L in 2010-2011 to less than 10 µg/L through spring 2017.

Vinyl chloride concentrations were also analyzed using the Mann-Kendall trend test. The EPA ProUCL version 5.1.002 software package (EPA, 2015) was utilized to evaluate the vinyl chloride concentration trends at a level of significance (p-value) of 0.05 (i.e., a confidence coefficient of 95 percent). The effectiveness of this trend analysis decreases when greater than 50 percent of results are below detection limit (BDL), so wells with BDLs above 50 percent are not analyzed (censored). Results with p-values below 0.05 indicate a statistically significant trend (up or down); results with p-values above 0.05 indicate no statistically significant trend (i.e., concentrations are stable). Table 8 lists the results of the Mann-Kendall testing, including number of data points, number of detections, percent BDL, and trend (if any) for the UGZ and MGZ wells with detections of vinyl chloride. The Mann-Kendall trend test analysis outputs and graphs are included in Appendix C.

- Based on the Mann-Kendall analysis, 12 of 13 UGZ wells show statistically significant decreasing vinyl chloride trends at a 95 percent confidence factor. Well T-9 shows no trend (i.e., vinyl chloride concentrations are stable).

- Five of nine MGZ wells show statistically significant decreasing vinyl chloride trends at a 95 percent confidence factor. One MGZ well (D-2) exhibits a statistically significant increasing vinyl chloride trend. Wells MW-29 and MW-31B show no statistically significant trend at a 95 percent confidence factor (i.e., concentrations are considered stable).

5.1.3 French Drain Outfall Analytical Results

VOC results from sampling of the french drain outfall (FD-1) and the intermittent stream culvert outlet (CS-1) are provided in Table 2. The french drain outfall was initially sampled for ten quarters beginning in December 2014. Vinyl chloride has been detected at concentrations of 5 µg/L in the initial sampling, but concentrations have decreased to approximately 1 µg/L or less since then and vinyl chloride was not detected in December 2016. *Cis*-1,2-DCE was detected in the first four sampling rounds at concentrations ranging from 0.6 to 1.1 µg/L, but has not been detected since September 2015. Acetone was reported at a concentration of 6 µg/L in December 2016 but this result is suspected to be due to laboratory contamination. Detection of vinyl chloride in the french drain outfall indicates that perched water on the landfill side of the french drain is locally impacted by contact with landfill waste or contaminated soil and is flowing into the french drain. The reason for the apparent decline in vinyl chloride concentrations is not known but may reflect decrease in the source. Plots of historical vinyl chloride concentrations show both seasonal and long-term trends. Figure 31 show vinyl chloride concentrations for the french drain outfall.

The culvert outlet (CS-1) has been sampled for nine quarters and no VOCs have been detected except for acetone reported at 5.8 µg/L in December 2016 and methylene chloride at 0.50 µg/L in May 2017. These VOCs are suspected to be laboratory contaminants. In March 2016, surface water was flowing down the intermittent stream channel below the culvert outlet to Saxon Creek (normally flow from the outlet is low and infiltrates into the ground before reaching the creek. Samples were collected from the intermittent stream (CS-2), and from Saxon Creek immediately upstream and downstream of the confluence (CS-3 and CS-4, respectively). No VOCs were detected in these samples. Surface water flowed in the channel again during May 2017 and sample CS-2 was collected from the intermittent stream near the confluence with Saxon Creek. The only VOC detected was methylene chloride at 0.69 µg/L, which is suspected of being a laboratory contaminant and was not detected in the sample from the french drain outfall.

5.1.4 Monitored Natural Attenuation Parameters

Additional sampling for MNA parameters was conducted to provide background/baseline data for evaluating the potential for natural attenuation or enhanced in-situ groundwater bioremediation. The possibility of incorporating MNA in the overall site remediation strategy should be evaluated in a subsequent feasibility study.

There are non-biological and biological degradation processes capable of degrading vinyl chloride, a number of which can be enhanced as part of a full scale remedy. However, the three primary degradation processes that could be considered for this site are:

- Anaerobic reductive de-chlorination.
- Aerobic oxidation.
- Aerobic co-metabolic oxidation.

The bioremediation pathway most commonly used to remediate the vinyl chloride parent compounds PCE and TCE is anaerobic reductive de-chlorination; however, vinyl chloride can also be readily degraded by all three processes. Research has shown that vinyl chloride typically degrades faster under aerobic conditions than under anaerobic conditions. However, both approaches are capable of degrading the remaining vinyl chloride plume within a reasonable time frame. The most important thing to consider when selecting a remedial approach is to evaluate the existing natural conditions to determine which degradation process can most effectively accomplish the remediation goals.

During this phase of investigation, five wells were selected for MNA parameters to represent background and VOC impacted groundwater conditions in the UGZ and MGZ. Wells MW-30, T-6, and MW-20A were sampled to assess MNA parameters in the UGZ and wells D-1 and MW-18A were sampled to assess MNA parameters in the MGZ. A total of six sampling events were conducted in spring and fall quarters in December 2014, May 2015, December 2015, May 2016, December 2016, and May 2017. Results of the MNA sampling are provided in Table 7, which includes field parameters and results of MNA sampling from previous investigations.

The lack of significant concentrations of parent compounds such as PCE and TCE and, to a lesser extent, *cis*-1,2-DCE, indicates that anaerobic reductive de-chlorination biodegradation has previously occurred naturally to a large extent up-gradient of and within the current vinyl chloride plume. The persistence of the remaining vinyl chloride plume indicates that there is some resistance to complete degradation of the residual vinyl chloride plume under the natural conditions that exist at the site.

Vinyl chloride is the final daughter product of reductive de-chlorination from parent compounds with higher numbers of chlorine atoms in their structure. The sequence generally proceeds from PCE, to TCE, to *cis*-1,2-DCE, and then to vinyl chloride. *Trans*-1,2-dichloroethene is also a typical dechlorination daughter product but is not prevalent at the Site. Thus, the presence of vinyl chloride in the groundwater at the Site is considered to have resulted from degradation of parent compounds PCE and/or TCE. Because of low concentrations (approximately 2 to 3 µg/L) of PCE and/or TCE detected since the onset of monitoring in the mid-1990s, initial reductive de-chlorination is believed to have largely occurred prior to that time. PCE has been historically detected at concentrations ranging from 0.06 to 3 µg/L in up to 16 wells and TCE has been historically detected at concentrations ranging from 0.6 to 2 µg/L in up to 13 wells.

Recent detections of PCE are largely restricted to MGZ wells MW-17B, MW-21A, MW-24, and D-3 with concentrations generally varying within a range of 0.6 to 2 µg/L. PCE was not detected in well D-3 until 2007. PCE was initially detected in well MW-18A at approximately 0.6 to 0.8 µg/L, but has not been detected since 2013. Recent detections of TCE are largely restricted to wells MW-21A, MW-24, and D-2, all MGZ wells, with concentrations generally varying within a range of 0.3 to 1.8 µg/L. Similar to PCE, TCE was initially detected in well MW-18A at approximately 0.5 to 0.8 µg/L, but has not been detected since 2013. Remaining low concentrations of PCE and/or TCE in these wells suggest initial phase reductive de-chlorination is incomplete in these locations.

De-chlorination daughter product *cis*-1,2-DCE has been historically detected in up to 35 wells, and in 11 wells in spring 2017. Detections of *cis*-1,2-DCE are sporadic in many wells, but other wells show more consistent occurrences. Concentrations differ by location and most occurrences are in the range of approximately 1 to 6 µg/L, with a couple of locations showing greater concentrations in the range of approximately 10 to 20 µg/L, particularly at MGZ well D-2. Most locations show either no apparent trend or decreasing *cis*-1,2-DCE concentrations historically. Possible increasing *cis*-1,2-DCE concentrations trends have been observed at MW-21 (from less than 1 µg/L to approximately 10 µg/L over a period of six years) and at D-2 (from less than 10 µg/L to approximately 20 µg/L, although concentrations appear to be leveling off). Lingering, mostly low, concentrations of *cis*-1,2-DCE in several wells indicate reductive de-chlorination is incomplete.

Ratios between parent and daughter compounds (PCE/TCE/*cis*-1,2-DCE/vinyl chloride) provide an indication of degree of de-chlorination that has occurred. Ratios in a number of wells are 0/0/0/1, indicating complete degradation to vinyl chloride. Most wells containing *cis*-1,2-DCE show ratios in the approximate range of 0/0/1/5 to 0/0/1/20. Wells with detectable PCE and/or TCE exhibit varied ratios: MW-17B (1/0/0/0 to 2/0/1/0); MW-21A (1/0/2/0, changing to 1/1/8/3 as vinyl chloride became detectable); MW-24 (2/1/8/20, changing to 2/1/10/15 more recently); D-2 (0/1/20/25 to 0/1/30/40); and D-3 (1/0/0/0). At MW-18A, where both PCE and TCE have become non-detectable, ratios changed from approximately 1/1/10/70 to 0/0/1/5. Parent/daughter ratios indicate degradation to vinyl chloride is nearly complete in almost all UGZ wells, but less complete in several of the MGZ wells.

The half-life of vinyl chloride was estimated for the UGZ using concentrations at two distances along the plume centerline to calculate a first-order decay constant (EPA, 1998). Concentrations were normalized using chloride concentrations as a conservative tracer. Inputs regarding seepage velocity were taken from the calibrated groundwater flow model and those pertaining to retardation were developed from transport calibrations. A conservative organic carbon fraction of 0.005 was assumed. Based on the calculations (summarized in Table 2 of the Groundwater Flow and Contaminant Transport Modeling report in Appendix D), a half-life for vinyl chloride within the UGZ was estimated to be 2.9 years. A half-life estimate was not calculated for the MGZ due to insufficient data available for utilization of this method.

The MNA results suggest primarily aerobic conditions with localized anaerobic conditions exist across the site. Results for individual parameters are discussed below.

Dissolved Oxygen

Results for upgradient well MW-30 range from 2.1 milligrams per liter (mg/L) to 11 mg/L, with field measurements generally ranging from 1.21 to 5.93 mg/L (anomalous field measurement in December 2016 of 58.4 mg/L). These results indicate primarily aerobic conditions existing upgradient of the landfill. DO concentrations in wells T-6 and MW-20A in the UGZ range from 2.3 to 8.2 mg/L and field measurements ranging from 0.81 to 4.77 mg/L (anomalous field measurement in December 2016 of 23.4 mg/L in T-6). These results indicate generally aerobic conditions in the UGZ within the contaminant plume, although lower than in the upgradient location. DO concentrations in wells D-1 and MW-18A in the MGZ range from 1.8 to 7.1 mg/L and field measurements generally ranging from 0.31 to 9.38 mg/L (anomalous result in December 2016 of 36 mg/L in D-1). These results also indicate generally aerobic conditions in the MGZ within the contaminant plume, although lower oxygen concentrations occur at times, and are lower in general than in the UGZ. Past MNA results from wells located downgradient of the landfill suggest more anaerobic conditions may have occurred. Field DO concentrations in UGZ wells T-1 and T-6 have ranged from approximately <1 to 2.27 mg/L. Past MNA results further suggest more aerobic conditions return further downgradient in the UGZ when less aerobic conditions exist upgradient. Field DO concentrations in OW-3 have ranged from 5.24 to 6.06 mg/L indicating more aerobic conditions in this area (anomalous reading of 12.63 mg/L measured in March 2009).

Oxidation-Reduction Potential

Field measurements of ORP typically show positive readings ranging to +278.2 millivolts (mV), indicating a generally oxidizing chemical environment. Negative ORP values have been measured periodically, particularly in MGZ wells D-1 and MW-18A, ranging from -2.0 to – 369 mV, indicating that reducing conditions exist at times in the MGZ. Sporadic negative ORP readings recorded in UGZ wells may indicate periodic changes in the groundwater environment within that zone. Past ORP readings from T-1 were recorded as -13.1 to -133.6 mV.

Total Organic Carbon

Total organic carbon (TOC) in the aquifer may be utilized as a carbon source, electron acceptor, and/or co-metabolic inducer. TOC is also indicative of potential bacteriological nutrients and is a significant factor in potential sorption and retardation of migration of VOCs in groundwater. TOC in upgradient UGZ well MW-30 ranges from 0.52J mg/L to 0.82J mg/L when detected (the J flag indicates detection at an estimated value below the reporting limit). TOC in UGZ wells T-6 and MW-20A ranges from 1.2 to 16 mg/L. TOC in MGZ well D-1 ranges from 18 to 22.2 mg/L and in MW-18A ranges from 6.7 to 12 mg/L. TOC concentrations have been consistent during monitoring of these wells. Biodegradation of chlorinated VOCs may be enhanced by higher TOC, and may be occurring to a greater degree in the area of well D-1.

Nitrate and Nitrite

Nitrate may serve as an electron acceptor in anaerobic environments but can hinder microbial degradation if present at concentrations above 1 mg/L. Nitrate concentrations in the upgradient UGZ well range from 0.13J to 0.40J mg/L when detected. Nitrate concentrations in T-6 range from 0.12J to 8.0 mg/L and in MW-20A have not been detected above 0.50 mg/L with the exception of 0.13J mg/L reported in May 2017. Nitrate concentrations in MGZ wells D-1 and MW-18A range from 0.095J to 1.9 mg/L. In general, nitrite has not been detected above 0.5 to 2.5 mg/L, except for estimated detections in upgradient well MW-30 of 0.13J to 0.40J mg/L and a single detection at 0.16J mg/L in well T-6.

Dissolved Iron and Manganese

Reduction of ferric iron to ferrous iron occurs during anaerobic biodegradation of organic carbon including from VOCs, with ferric iron acting as an electron acceptor, and may foster anaerobic oxidation of vinyl chloride to carbon dioxide. In MW-30, ferric iron concentrations range from 0.10J to 1.2 mg/L (with outliers at 26 and 34 mg/L), and ferrous iron concentrations range from 0.19J to 0.55 mg/L. In UGZ wells T-6 and MW-20A, ferric iron is sporadically detected between 0.21J and 0.43J mg/L and ferrous iron is sporadically detected from 0.23J to 0.62J mg/L. In MGZ well D-1, ferric iron has been detected between 0.84 and 1.8 mg/L and ferrous iron has been detected from 2.3 to 10 mg/L. In MGZ well MW-18A, ferric iron has been detected at 0.16J to 0.46J mg/L and ferrous iron has been detected at 0.20J to 1.2 mg/L. These results suggest that anaerobic oxidation of vinyl chloride may occur locally in the MGZ.

After iron reduction has occurred, manganese may serve as an electron acceptor. Divalent manganese above 1 mg/L may foster anaerobic oxidation of *cis*-1,2-DCE. Divalent manganese concentrations in upgradient UGZ well MW-30 range from 0.19J to 2.9 mg/L. In downgradient UGZ wells T-6 and MW-20A, Mn+2 concentrations range from 0.23J to 9.5 mg/L. In MGZ well D-1, divalent manganese concentrations range from 7.6 to 10 mg/L and in MW-18A, concentrations range from 0.92 to 2.6 mg/L. *Cis*-1,2-DCE concentrations in these wells appear to be diminished in comparison to vinyl chloride concentrations.

Sulfate

Sulfate may serve as an electron acceptor in anaerobic environments in the absence of nitrate, iron, and manganese. Sulfate may compete with other electron acceptors and inhibit reductive dechlorination at higher concentrations above 20 mg/L. Sulfate concentrations in upgradient well MW-30 range from 1.0 to 1.8 mg/L. Sulfate concentrations in UGZ wells T-6 and MW-20A range from 3.3 to 65.6 mg/L. Sulfate concentrations in MGZ wells D-1 and MW-18A range from 40 to 74 mg/L. Relatively elevated sulfate concentrations in both UGZ and MGZ wells suggest that it may inhibit the degree of reductive de-chlorination.

Chloride

In addition to solubilization of salts, chloride can be introduced into the groundwater environment from degradation of chlorinated hydrocarbons. Presence of chloride at concentrations significantly above background levels may indicate decomposition of chlorinated hydrocarbons. Chloride concentrations in upgradient well MW-30 range from 1.1 to 2.9 mg/L. Chloride concentrations in UGZ wells T-6 and MW-20A generally range from 23 to 80.4 mg/L (outliers ranging from non-detect to 217 mg/L have been reported for T-6). Chloride concentrations in MGZ wells D-1 and MW-18A range from 72 to 208 mg/L. These results suggest elevated chloride concentrations may be due to liberation from degradation of chlorinated hydrocarbons.

Methane, Ethane, Ethene, and Carbon Dioxide

In addition to generation within the landfill mass, methane may also be an indicator of anaerobic bacterial activity in the groundwater as a later stage product of reductive de-chlorination.

Elevated methane concentrations are associated with accumulation of vinyl chloride. In upgradient well MW-30, methane concentrations range from 1.4 to 11 µg/L. In UGZ well T-6, methane concentrations range from 80 to 1,300 µg/L (with one outlier at 3.6 µg/L, possibly due to wet season dilution) and in MW-20A range from 26 to 200 µg/L. In MGZ well D-1 methane concentrations range from 1,100 to 2,000 µg/L (with one outlier at 2 µg/L, possibly due to wet season dilution), and in MW-18A range from 300 to 2,163 µg/L. The methane occurrences suggest that reductive de-chlorination has occurred to the point that vinyl chloride largely persists as the remaining chlorinated hydrocarbon.

Ethane and ethene represent the products of complete reductive de-chlorination of chlorinated hydrocarbons. Concentrations above 100 µg/L are indicative of completeness of the reductive de-chlorination process. Ethane and ethene concentrations in upgradient well MW-30 range from 0.0031J to 0.0088J µg/L and 0.0049J to 0.20 µg/L, respectively. Concentrations of ethane and ethene in UGZ wells T-6 and MW-20A range from 0.0094J to 0.14 µg/L and 0.036J to 3.0 µg/L, respectively. Concentrations of ethane and ethene in MGZ wells D-1 and MW-18 range from 0.023J to 0.073 µg/L and 0.040J to 1.6 µg/L, respectively. The low concentrations of ethane and ethene present in groundwater indicate reductive de-chlorination is incomplete.

Carbon dioxide represents the ultimate oxidative breakdown product of chlorinated hydrocarbons, produced by mineralization of organic compounds under anaerobic or aerobic conditions. Carbon dioxide concentrations in upgradient UGZ well MW-30 range from 23 to 47 mg/L. Carbon dioxide concentrations in UGZ wells T-6 and MW-20A within the plume are significantly higher, ranging from 100 to 360 mg/L (respective outliers of 30 and 89 mg/L were reported in May 2017, possibly due to wet season dilution). In the MGZ wells, carbon dioxide concentrations typically range from 150 to 350 mg/L (anomalous low concentrations were reported in May 2016 in D-1 [22 mg/L] and in December 2015 in MW-18A [78 mg/L]). The presence of carbon dioxide above background levels may indicate occurrence of microbial activity. Similarly, elevated alkalinity concentrations above 500 mg/L were detected in previous sampling of MW-18A, T-1, T-6, T-14, D-1 AND D-2 which may also be indicative of microbial activity, leading to increased carbon dioxide production.

It has been shown in the literature that it is possible for "micro-environments" to exist where anaerobic biodegradation processes can occur in a predominately aerobic aquifer and conversely aerobic biodegradation processes can exist in a predominately anaerobic environment. However, it is also commonly understood in the environmental industry that it would not be practical to design a groundwater remediation strategy that would depend upon both aerobic and anaerobic process to exist and flourish at the same time within the same groundwater plume area.

It has, however, been demonstrated to be an effective approach to use both aerobic and anaerobic biodegradation processes separated either spatially or temporally. An example of this would be to implement anaerobic reductive de-chlorination of TCE and/or *cis*-1,2-DCE in a source area and at the same time to implement aerobic biodegradation of vinyl chloride in a portion of the plume located farther down-gradient which is naturally more aerobic. Another example would be to implement anaerobic reductive de-chlorination across the entire plume area until only vinyl chloride remains. Over time, aquifer conditions will typically begin to return to more aerobic conditions at which time aerobic biodegradation efforts could subsequently be implemented across the entire plume in order to complete the vinyl chloride degradation, taking advantage of the aquifer naturally returning to aerobic conditions and the faster aerobic biodegradation rates. This can effectively complete the groundwater remediation process while also adding oxygen and raising the redox potential which can provide other ancillary benefits such as reduced dissolved metals concentrations and improved taste and odor of the water.

At the Site, PCE and TCE are largely absent or at very low concentrations, and *cis*-1,2-DCE concentrations are considerable lower than vinyl chloride in most locations (with the exception of MW-24 and D-2). Based on dissolved oxygen levels and positive ORP values, aquifer conditions are generally aerobic. This suggests that a strategy of enhancing aerobic biodegradation may be most effective to mitigate vinyl chloride impacts to the groundwater.

5.1.5 Stream Gaging and Sampling

Stream sampling was conducted in 12 of the 24 quarters and gaging was conducted during 12 of the 24 quarterly monitoring events. Gaging and sampling locations are shown on Figure 2. Results of the stream gaging are included in Table 6. Analytical results of the stream sampling are included with the groundwater results in Table 2. No VOCs were detected in any stream sample during this investigation.

Comparing flow variation over short distances in a single stream without significant influx between gaging stations is a challenge. Subtle discharge variation can be influenced by difficulties in measuring the streambed profile, obtaining accurate flow velocity readings in a shallow stream often only a few inches deep, hourly discharge changes that are common as a result of daily heating and melting of upstream snowpack, and diurnal variations due to changes in vegetation transpiration.

Review of the stream flow measurements in Table 6 show inconsistent results regarding the interaction of Saxon Creek and shallow groundwater. Stream gaging typically found steady state or losing stream conditions between Fountain Place Road and East of well OW-7 with occasional periods of slightly gaining conditions during spring gaging events. The expected model is Saxon Creek would generally be a gaining stream during high groundwater conditions in the spring and a losing stream as water levels drop after the spring surge in groundwater elevations. Results from the November 2007 and May 2008 gaging events exhibit flow patterns consistent with this model. In November 2007, the results show a fairly uniform drop in discharge volume from 1.11 cubic feet per second (cfs) at Fountain Place Road Bridge to 0.80 cfs downstream at Powerline Road Bridge indicating a losing stream condition, but the overall discharge was very low and may represent steady state conditions. In May 2008 during the spring run-off, the discharge increased slightly from 11.67 cfs at Fountain Place Road Bridge, to 12.20 cfs near MW-15 well cluster, to 12.38 cfs at Powerline Road Bridge before dropping off to 10.93 cfs downgradient near OW-7. These results indicate Saxon Creek at the time was a gaining stream east of the landfill and a losing stream downgradient of Powerline Road Bridge. However, the discharge differences are again very subtle and may reflect more steady state conditions then measurable trends. Results from the October 2009 gaging event are less consistent with the expected groundwater/stream interaction model, when gaging values indicated losing stream conditions between Fountain Place Road Bridge and the MW-15 well cluster, gaining stream between MW-15 cluster and Powerline Road Bridge, and losing again north of Powerline Road Bridge. In July 2011, following a season of above average precipitation and during above average groundwater levels, Saxon Creek was a losing stream, opposite of the expected model.

Flows decreased steadily downstream, from 41.16 cfs at Fountain Place Road Bridge, to 31.80 cfs near MW-15 well cluster, to 29.00 cfs at Powerline Road Bridge and 24.39 cfs downgradient near OW-7. The stream water elevations measured at gaging stations SG-2, SG-3 and SG-4 during the July 2011 gaging event were two to four feet lower than groundwater elevations in nearby wells MW-15, T-14, and OW-7, which should have impeded the seepage of surface water to the subsurface and apparent decrease in flow through that area. The variances of the relatively low stream flow rates could be the result of measuring error.

Gaging results since then have been inconsistent or represent generally steady-state conditions, but higher flows observed at times at the Powerline Road Bridge and Well OW-7 stations indicate periodic gaining stream conditions in that area, which may reflect interaction with shallow groundwater. During the exceptional high flows and groundwater levels observed in May 2017, streamflow could only be measured at Fountain Place Road Bridge and Powerline Road Bridge. At that time, Saxon Creek was a significantly losing stream between those two locations similar to July 2011, with flow decreasing sharply from 78.67 cfs at Fountain Place Road Bridge to 52.11 cfs at Powerline Road Bridge. Water levels in two wells close to Saxon Creek near Powerline Road Bridge (T-14 and MW-20A) were actually at or above ground surface, and localized minor ponding was also observed in that area.

In 1996, the USGS conducted a surface water monitoring program which included two gaging stations upstream and downstream of the landfill (USGS, 2000). Station #61 was located below the landfill at Powerline Road, and Station #62 was located above the landfill at USFS Road 12N01 (Fountain Place Road). The USGS reported discharges of 2.4 cubic feet per second at Station #61 and 2.5 cubic feet per second at Station #62, and concluded Saxon Creek was steady state through this stretch.

5.2 GROUNDWATER FLOW AND CONTAMINANT TRANSPORT MODELING

A steady state groundwater flow and transport model was prepared to evaluate the vinyl chloride plume within the UGZ and MGZ. Documentation of the model design and results are presented in the summary report included as Appendix D. Modeling utilized standard finite difference simulators by the USGS including MODFLOW NWT, MODPATH, and MT3DMS to simulate groundwater flow, particle tracking, and contaminant transport.

Geospacial data representing the UGZ, MGZ, and LGZ developed from site-specific and regional geologic and stratigraphic information were used as inputs to the model. Groundwater flow was developed and calibrated through an iterative process using 2015 hydraulic head data and residual values were assessed.

Initial plume conditions from 2010 data were used to develop the contaminant transport model to check calibration against 6 years of subsequent data and also because concentration data was lacking before then for many of the wells of interest. Additional inputs were developed including estimation of vinyl chloride half-life, groundwater seepage velocity, and retardation.

Comparison of observed and simulation concentrations over time indicated some variability in quality of fit. Several wells exhibited close correspondence (T-6, T-8, T-9, T-12, and T-14 in the UGZ and MW-22A in the MGZ), while others showed somewhat poor correlation (OW-5, PU-2, and T-2 in the UGZ and MW-29 in the MGZ).

Fate simulations suggest the UGZ plume will fully attenuate by 2030 assuming no ongoing source contribution, and stabilizing at lower concentrations by 2030 utilizing an assumed source term at the landfill. The plume would persist until the source is removed. For the MGZ, the model suggests northward migration of the center of mass will occur. However, observed concentration trends suggest that concentrations at the downgradient end of the plume are decreasing (at MW-24, e.g.), while concentrations farther upgradient (at D-2) appear to be increasing, possibly due to contribution from downward migration of contamination from the UGZ in this area. The model may underestimate the rate of mass transfer from the UGZ to the MGZ as well as the decay rate within the MGZ.

6. CONCLUSIONS AND RECOMMENDATIONS

Recent groundwater investigations have been completed by WESTON to address existing data gaps and further refine the site hydrogeologic model. The additional studies have provided data to:

1) Determine whether contaminant migration beyond the landfill waste boundary is still occurring and determine that waste does not appear to be in contact with the UGZ beneath the landfill. However, investigations suggest some limited interaction between the perched zone and waste may be occurring in the southwestern-most part of the landfill and a potential for limited contact between groundwater in the UGZ and landfill waste during highest seasonal groundwater levels, especially during years of increased precipitation.
2) Determine the effectiveness of the french drain in directing perched groundwater away from the landfill and source of vinyl chloride contamination entering the french drain.
3) Updates to the *Interim Groundwater Characterization Report, Meyers Landfill, El Dorado County, California* (July 2012) report with the following components:

- Further refine the site hydrogeologic model and further delineate VOCs in the aquifer.
- Evaluate hydraulic gradients laterally and vertically between water-bearing zones.
- Examine historical vinyl chloride trends and calculate potential transport velocity.
- Conclude that mostly aerobic conditions with localized anaerobic conditions exist across the site allowing incomplete reductive de-chlorination to vinyl chloride for most of the site. The possibility of incorporating MNA in the overall site remediation strategy is potentially feasible and should be evaluated as part of a subsequent feasibility study.
- Determine that stream gaging results are variable regarding the interaction between Saxon Creek and the groundwater. VOCs were not detected in any surface water samples collected from the creeks.
- Evaluate potential fate of the vinyl chloride plumes using groundwater flow and contaminant transport modeling.

6.1 CONCLUSIONS

Results of these investigations built upon previous studies to formulate the following conclusions.

6.1.1 Landfill Waste Interaction with Groundwater

Three UGZ wells (LF-1, LF-2, and LF-3) were installed along the estimated axis of the buried intermittent stream channel beneath the landfill. The wells were screened beneath the landfill waste to evaluate potential groundwater/waste interaction by providing water level conditions and contaminant data before and after capping of the landfill. During drilling and well installation, the base of the waste and static water levels were identified at:

- LF-1 – base of waste (36 feet bgs), static water level (54 feet bgs)
- LF-2 - base of waste (43 feet bgs), static water level (62 feet bgs)
- LF-3 - base of waste (44 feet bgs), static water level (56 feet bgs).

Quarterly groundwater elevation measurements were collected from the three landfill UGZ monitoring wells. These measurements show groundwater elevations in the three UGZ landfill wells are typically approximately 10 to 15 feet below the lowest recorded waste elevation and have not risen to the base of the landfill waste since these wells were installed in 2008; although, historically high UGZ groundwater elevations in wells LF-1, LF-2 and LF-3 were within 2 to 5 feet from the landfill waste in spring 2017. Water elevations from well T-2 located at the downgradient toe of the landfill rose to within approximately 2.5 feet of the base of the waste in that area during the highest measured groundwater elevation in May 1998, but is primarily 10 to 15 feet below the waste elevation.

Projection of the UGZ groundwater potentiometric surface between MW-30 and LF-1 suggests potential for interaction between water in the UGZ and waste near the southern end of the landfill during high groundwater events. However, the UGZ was encountered in well MW-30 at approximately 40 feet bgs and is confined beneath the PGZ aquitard at this location. The relationship of groundwater elevations to the base of the waste is depicted on Figure 28.

In addition, due to the steep eastward gradient in the UGZ west of the landfill, it is possible that groundwater elevations beneath the western side of the landfill have approached the base of the landfill waste at times, as suggested by the cross section diagrams in Figure 26. The UGZ encountered in wells PU-1 and PU-2 is confined by the PGZ aquitard and potentiometric groundwater levels are often above the UGZ water bearing zone. However, the eastern extent of the PGZ aquitard beneath the landfill is unknown. The aquitard appears to be eroded away in this area.

A zone of groundwater is perched on a shallow aquitard along the west and southwest side of the landfill, and up the intermittent stream valley south of the landfill. The french drain system was installed during the recent cap construction to collect and divert perched water from the west and southwest of the landfill to a collection trench along the southern end of the landfill. Detections of low concentrations of vinyl chloride in the french drain outfall indicate perched water contaminated by interaction with the contaminated soil or landfill waste is also entering the french drain. Sampling results to date suggest this is occurring in the area between piezometers PP-4 and PP-9.

Perched groundwater is also present beneath the intermittent stream valley south of the landfill. During landfill consolidation and capping landfill debris was found near well MW-16 immediately above the perched groundwater, at the south end of the landfill. If the shallow perched groundwater extends to the main body of the landfill, it could come in contact with landfill waste. Further, the presence of groundwater in PTEM boring B-20 (Figure 3) suggests some limited interaction between the perched zone and waste may be occurring in the southwestern-most part of the landfill; however, perched groundwater was not encountered while drilling the landfill well LF-1 located approximately 250 feet north of MW-16.

6.1.2 French Drain Assessment

Effectiveness of the new french drain system was evaluated by installation of piezometers to monitor water levels in the PGZ. Initially, piezometers PP-1 and PP-3 were installed to the west of the french drain in September 2011, and PP-2 and PP-4 were installed between the french drain and the landfill in June 2012. Water level observations indicated that in general, groundwater in the PGZ was flowing east toward the french drain and being captured and discharged, although at times, water levels on the landfill side of the french drain were higher than those to the west, and perched groundwater was suspected to be periodically flowing from the landfill toward the french drain. In order to evaluate potential migration of contaminants to the french drain, sampling of the french drain outfall was initiated in December 2014, and vinyl chloride was initially detected at 5µg/L, along with 0.86 µg/L of *cis*-1,2-DCE. Subsequent sampling rounds continued to show detectable vinyl chloride, although at lower concentrations through the most recent monitoring event in May 2017. Additional exploration of the PGZ was conducted to further evaluate groundwater flow and potential impacts to the PGZ. Piezometers PP-1 and PP-3 were sampled beginning in December 2014 and PP-2 and PP-4 were sampled beginning in March 2015. Vinyl chloride was detected in PP-4 at a concentration of 2.2 µg/L in that quarter. An additional nine piezometers (PP-5 through PP-13) were installed on either side and to the north of the french drain in July 2015 to further evaluate groundwater levels and potential impacts in the PGZ.

Groundwater flow in the PGZ differs with location along the french drain. In the area immediately west of the french drain, gradient is mostly toward the east-southeast (toward the french drain) at steep gradients of 0.03 to 0.09 foot/foot. Farther west, gradients are toward the west, following the surface of the underlying westward-dipping aquitard. Between the french drain and the landfill, gradients generally have a significant component toward the french drain, with an overall decrease in groundwater elevations from north to south. During the summer and fall, perched groundwater is typically absent in the area east of the french drain.

A series of six cross sections through the PGZ and landfill wells are presented in Figures 26 and 27 showing the relationships between the PGZ, UGZ, french drain, and landfill. The sections depict groundwater levels in the PGZ during dry (December 2015) and wet (March 2016) seasons. During dry season conditions, flow on the west side of the french drain is toward the east, where it is intercepted by the french drain. During wet season conditions, groundwater accumulates in the area between the french drain and the landfill and flows west toward the french drain. Under these conditions, PGZ groundwater elevations along this area that are higher than those found in piezometers on the opposite side of the french drain may flow across the french drain. The periodic presence of groundwater observed in piezometers located between the french drain and landfill suggests that perched water is flowing across the french drain after periods of snow melt and/or that localized recharge is occurring, resulting from percolation of surface water between the french drain and the landfill.

Vinyl chloride concentrations detected in perched groundwater range from 0.71 to 2.2 µg/L. The source of the vinyl chloride in the PGZ is either from contaminated soil above the perched aquitard or from rapid snow melt or precipitation resulting in the infiltration of water adjacent to the landfill and seepage from beneath the land fill cap where the perched groundwater contacts the western margins of landfill waste. Vinyl chloride concentrations in perched groundwater and the french drain outfall are decreasing based on the data to date. The decrease in the vinyl chloride concentrations are likely due to the groundwater flushing contaminants in areas where the landfill waste was disturbed during consolidation and capping activities and are expected to stabilize overtime.

6.1.3 Site Hydrogeologic Model

Hydrogeologic Model

The site hydrogeologic conceptual model has been defined into three primary water-bearing zones (UGZ, MGZ and LGZ), with a perched groundwater zone (PGZ) generally occurring in the area west of the landfill. The installation and sampling of wells PU-1 and PU-2 identified vinyl chloride in the UGZ west (upgradient) of the landfill while well MW-31A defines the VOC plume to the west. The installation of well MW-30 serves as a UGZ background well located south of the landfill. The VOC plume is well defined in the UGZ using the existing network of groundwater monitoring wells.

In the MGZ, well MW-28 constrains the plume to the west while wells MW-29 and MW-31B identified the plume northwest west of the landfill, respectively. Generally low vinyl chloride concentrations at MW-31B identify the southwestern edge of the plume in the MGZ in this direction. The extent of the VOC plume is fairly well defined in the MGZ, but is beyond well MW-24. The MGZ was not encountered during the drilling of well MW-26 located down gradient of MW-24. Continued monitoring of the LGZ wells indicates the LGZ is not impacted.

Four piezometers were initially installed in the PGZ to monitor effectiveness of the new french drain in diverting the eastward flow of groundwater away from the landfill. Erratic groundwater flow was found between the landfill and french drain which led to the sampling of the piezometers and french drain outfall to discover vinyl chloride impacts to the PGZ. Nine additional piezometers were installed in response to low detections of vinyl chloride in PGZ and french drain outfall and further assess groundwater flow along the french drain. Results of sampling and water level gaging suggest that perched groundwater accumulates on the landfill side of the french drain during the wet season, and that perched groundwater appears to flow across the french drain locally.

Groundwater Elevation Monitoring

The depth to water in Site wells range from above ground surface in wells near Saxon Creek, to over 200 feet deep in a well located on the ridge along Pioneer Trail. As expected, groundwater elevations are highest in the late spring following snowmelt and lowest in late fall/winter prior to winter precipitation and spring thaw.

A downward vertical gradient with elevation differences of around one to five feet is noted between the UGZ and MGZ in most cluster wells with the exception of well cluster OW-3/MW-18A with an elevation difference around 13 feet, MW-31A/MW-31B with an elevation difference of more than 30 feet, and MW-19A/MW-19B located near Saxon Creek in which the water level in the MGZ is typically higher than in the UGZ. Upward vertical gradients are observed in one to five well clusters in a given quarter. A stronger downward vertical gradient is noted between the MGZ and LGZ cluster wells with difference in groundwater elevations ranging from around 20 to 30 feet. Hydrographs show similar patterns between the UGZ and MGZ indicating these wells are interconnected or at least respond similarly to hydraulic changes.

In the PGZ, water levels west of the french drain generally vary a relatively small amount seasonally, typically less than one foot (although levels increased between approximately four to seven feet in May 2017). Water levels between the french drain and the landfill commonly differ as much as eight feet seasonally, these wells going dry at times during the fall and winter quarters. Review of perched water elevations further suggests that perched water may flow across the french drain locally.

In general, water elevations in the UGZ vary seasonally by as much as two to five feet between the highest levels during the spring snowmelt and the lowest levels in late fall/early winter at the end of the dry season, although seasonal differences may range to more than seven feet during exceptionally wet years. Groundwater in the UGZ flows eastward away from the ridge along Pioneer Trail at a steep gradient of approximately 0.03 to 0.05 foot/foot. The gradient becomes flatter toward Saxon Creek. Groundwater flow is northeastward north of the landfill and along the axis of the valley at a gradient of approximately 0.004 to 0.006 foot/foot. However, groundwater flow direction and gradient remain fairly consistent seasonally.

Groundwater levels in MGZ monitoring wells appear to differ based on the vertical distribution of sand layers screened by the well. Groundwater flow in the MGZ is toward the north at a gradient of approximately 0.004 to 0.006 foot/foot to the north of the landfill toward Pioneer Trail. In general, water elevations in the MGZ vary by as much as two to six feet between the highest levels during the spring snowmelt and the lowest levels in late fall/early winter at the end of the dry season, similar to seasonal elevation variations exhibited in the UGZ. However, groundwater flow direction and gradient remains fairly consistent seasonally. The similarity between the water elevation variation in the UGZ and MGZ suggest that these zones, although distinct with respect to contaminant distribution, are interconnected or at least respond similarly to seasonal hydraulic changes.

Groundwater flow in the LGZ flows towards the west at a very shallow gradient of 0.0004 to 0.0009 foot/foot. The westerly flow direction in the LGZ may be influenced by pumping from Elks Club Well No. 2 located approximately 0.75 mile west of the site, but data has not been provided to evaluate this potential interaction. Water elevations in the LGZ also show subdued and often delayed response to seasonal variation, but do not vary more than around 0.5 foot.

Vinyl Chloride Trends

Review of vinyl chloride concentration trends in UGZ wells indicates stable or decreasing concentrations in most wells, by visual analysis and Mann-Kendall test trend analysis. The extent of the plume and area of highest concentrations also appear to be slowly decreasing.

Review of vinyl chloride trends and plume configuration for the UGZ indicates a hot-spot had previously migrated down the axis of the plume from T-6 to T-14. This suggests that a slug of vinyl chloride had migrated beyond the first set of monitoring wells located along the toe of the landfill, although vinyl chloride is still impacting groundwater from contaminated soil beneath the landfill as indicated by the continuing presence in wells T-1 and T-6 (although concentrations have diminished considerably since monitoring began in 1997). Utilizing the estimated progress of the concentration peaks between T-6, T-12, and T-14, a plume seepage velocity of approximately 265 feet per year (0.73 feet per day) was calculated. Using a hydraulic conductivity of 20 to 60 feet/day assuming a 100 foot aquifer thickness, a hydraulic gradient of approximately 0.005 foot/foot between T-6 and T-14, and an assumed porosity of 30 percent for the UGZ, the groundwater flow velocity is calculated at around 120 to 360 feet/year (0.33 to 1.0 foot/day). Using the hydraulic conductivity of 40 to 90 feet per day assuming an aquifer thickness of 65 feet and the same assumptions above, groundwater flow velocity is calculated to range from approximately 240 feet to 550 feet/year (0.67 to 1.5 feet/day).

Review of vinyl chloride concentrations and plume configuration for the MGZ indicates the current hot-spot extends from the vicinity of D-1 to MW-18A. Since most MGZ wells were installed in 2008, and migration of a concentration peak could not be identified in this period, insufficient data is available to estimate groundwater flow or contaminant migration velocity by this method. Review of vinyl chloride concentration trends in MGZ wells indicates stable or decreasing concentrations in most wells. Slightly increasing trends are observed in well D-2, although concentrations may be reaching a peak. MW-21 appears to exhibit a slight increase, although results for this well were censored in Mann-Kendall test trend analysis due to more than 50 percent non-detect results during its monitoring period. Based on visual observations of time-series concentration graphs, vinyl chloride concentrations in MW-29 and MW-21B appear to have a potentially increasing trend, but Mann-Kendall analysis indicates no trend (i.e., stable concentrations) at a 95 percent confidence level.

Vinyl chloride has been detected at concentrations ranging up to 5 µg/L in the french drain outfall although concentrations have declined to approximately 1 µg/L in May 2017. Vinyl chloride has not been detected downstream at the culvert outlet to the intermittent stream channel. Vinyl chloride has been detected in three of the piezometers installed in the PGZ including PP-9 to the west of the french drain. Detected vinyl chloride concentrations in the PGZ piezometers range from 0.70 to 2.2 µg/L. Vinyl chloride was not detected in the french drain outfall or in piezometers sampled in December 2016 but was detected in May 2017.

The effect of installation of the impermeable landfill cap is not evident in vinyl chloride concentrations in groundwater beneath the landfill or downgradient. Since the time of the cap installation, most wells exhibit more stable concentration ranges, and most show decreasing trends, however, these effects are not clearly tied to installation of the cap as drought conditions had occurred during the same time period. Cap installation doubtless decreases leaching of contaminants from the primary source (the landfill) to groundwater due to decreased percolation of pore water through the landfill mass, although effects on the plume likely require additional time before effects can be documented.

MNA Monitoring

Sampling for MNA parameters was conducted to provide background/baseline data for evaluating the potential for natural attenuation or enhanced in-situ groundwater bioremediation. The lack of significant concentrations of parent compounds such as PCE and TCE and generally low concentrations of *cis*-1,2-DCE indicates that anaerobic reductive de-chlorination biodegradation has likely occurred up-gradient of the current vinyl chloride plume. The persistence of the remaining vinyl chloride plume indicates that there is some resistance to complete degradation of the residual vinyl chloride plume under the natural conditions that exist at the Site. The MNA results suggest mostly aerobic conditions with localized anaerobic conditions exist across the site leading to incomplete reductive dechlorination to vinyl chloride in most locations. Capping the landfill waste mass is anticipated to decrease the formation of additional leachate containing VOCs which should allow natural attenuation processes to reduce existing VOC concentrations. However, significant changes have not yet been observed since cap installation although concentrations appear to be trending lower overall at most locations. The possibility of incorporating MNA in the overall site remediation strategy is potentially feasible and should be evaluated in a subsequent feasibility study.

An estimation of half-life of vinyl chloride was made for the UGZ by calculation of first order decay based on concentrations at two distances along the plume centerline, with concentrations normalized employing chloride concentrations as a conservative tracer. Inputs to the calculations pertaining to particle flow rates and contaminant retardation were adopted from the calibrated groundwater flow model and transport calibrations. Based on the decay calculations a half-life for vinyl chloride within the UGZ was estimated to be 2.9 years. A half-life estimate was not calculated for the MGZ due to insufficient data available for utilization of this method.

Stream Sampling and Gaging

The USGS concluded that Saxon Creek was a steady state stream between Fountain Place Road and Powerline Road (USGS, 2000). Stream gaging conducted by WESTON typically found steady state or losing stream conditions between Fountain Place Road and East of well OW-7 with occasional periods of slightly gaining conditions during spring gaging events. Notwithstanding the difficulty in accurately gaging subtle flow variation across a short stretch of stream, the occurrence of shallow groundwater in monitoring wells located very near Saxon Creek suggests there are likely stretches of Saxon Creek that are gaining from groundwater seepage at various times of the year. However, surface water samples analyzed during this sampling program did not contain any VOCs indicating that VOCs are not seeping into Saxon Creek at detectable concentrations. Continued surface water sampling and stream gaging is recommended as part of the long-term monitoring program associated with the landfill capping.

Groundwater Modeling

A steady state groundwater flow and transport model was prepared to evaluate the vinyl chloride plume within the UGZ and MGZ that utilized standard finite difference simulators by the USGS to simulate groundwater flow, particle tracking, and contaminant transport.

Stratigraphic and litholgic information from site and regional investigations formed the basis for the physical model to represent the UGZ, MGZ, and LGZ. Groundwater flow was developed and calibrated through an iterative process using 2015 hydraulic head data. Initial plume conditions from 2010 data were used to develop the contaminant transport model to check calibration against 6 years of subsequent data and also because concentration data was lacking before then for many of the wells of interest. Additional inputs were developed including estimation of vinyl chloride half-life, groundwater seepage velocity, and retardation.

Comparison of observed and simulation concentrations over time indicated some variability in quality of fit. Several wells exhibited close correspondence while others showed somewhat poor correlation.

Fate simulations suggest the UGZ plume will fully attenuate by 2030 assuming no ongoing source contribution, and stabilizing by 2030 utilizing an assumed source term at the landfill but persisting until the source is removed. In the MGZ, the model suggests migration northward of the center of plume mass will occur. However, observed concentration trends suggest that concentrations at the downgradient end of the plume are decreasing (at MW-24, e.g.), while concentrations farther upgradient (at D-2) appear to be increasing, probably due to contaminant contribution from the UGZ. The model may underestimate the rate of mass transfer from the UGZ to the MGZ as well as the decay rate within the MGZ.

6.2 RECOMMENDATIONS

The following recommendations are provided to continue evaluating the groundwater for eventual completion of a supplemental RI/FS for OU-2:

- Continue routine groundwater monitoring to identify potential trends and further evaluate the effect that the remedial action on OU-1 (cap installation) may have on groundwater conditions.

- Continue routine monitoring of perched groundwater along the french drain and outfall sampling to verify declining trends in vinyl chloride are occurring. Exploratory trenching could be conducted in areas between the french drain and landfill to assess if landfill waste or contaminated soil are the source of perched groundwater contamination.

- Conduct additional investigation in the area south of the landfill to assess the extent of the perched groundwater and perched water aquitard and the potential of groundwater-landfill waste interaction.

- Continue evaluating potential groundwater/surface water interaction along Saxon Creek. Monitoring should consistent of gaging and sampling focusing in the area just upstream of Power Line Road Bridge to near well OW-7. Install shallow piezometers near the stream banks to provide additional data to determine whether the streams are losing, gaining, or steady state conditions are occurring in the area where the path of the UGZ plume and stream intersect.

- If further evaluation of naturally occurring in-situ bioremediation continues to be considered as a groundwater remediation strategy, incorporate pertinent natural attenuation monitoring parameters into the on-going quarterly groundwater sampling program and increase number of wells in MGZ sampled for MNA parameters so that a better estimation of degradation rate in the MGZ can be developed.

- Conduct additional evaluation of hydrogeological characteristics and downgradient plume extent within the MGZ to improve understanding of flow regimes and whether potential exists for further downgradient plume migration.

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